Short-term nutrient and sediment fluxes following dam removal

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To my wife Shanna, for her patience and support, to my daughter Gracen, and to the late Dr. Robert G. Wetzel for his guidance and inspiration.

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ABSTRACT

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Sediment and nutrient fluxes resulting from dam removal were investigated with a combination of field and laboratory studies. Impoundment-specific controls (i.e., regional, structural, biological and hydrogeomorphic) on loadings of dissolved organic carbon (DOC), inorganic and organic nitrogen and total suspended solids (TSS) were investigated. In particular, impoundment source areas (channel as well as floodplain wetlands) were compared to determine which represents a greater source of TSS, DOC and TDN to downstream environments. To determine if nutrient-rich sediments released from former impoundments continue to contribute C, N and P to the water column during downstream routing, a series of controlled laboratory experiments were performed. Sediment suspensions - at concentrations similar to those seen during dam removals – were exposed to simulated solar radiation, while DOC, total dissolved nitrogen (TDN), dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus (SRP) and CO_2 concentrations were measured before and after exposure. Additionally, the ability of successional plant community to sequester or otherwise immobilize interstitial N and P pools within formerly impounded sediment accumulations exposed by dam removal was

investigated. Finally, based on the experience and knowledge gained from this dissertation, a conceptual model of upstream and downstream disturbances resulting from dam removal was constructed. It is hoped that this dissertation will serve the shared interests among basic river researchers, river restoration practitioners, policy makers and aquatic resources regulators.

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CHAPTER I: INTRODUCTION

DAM REMOVAL: BASIC RESEARCH AND RIVER RESTORATION

Some of the most productive research efforts linking the historically independent fields of hydrology, geomorphology and ecology were accomplished by studying the effects of damming on river ecosystems. Likewise, the emergence of dam removal as a viable management strategy for river ecosystems has given basic researcher another chance to explore the interconnectivity among these disciplines. Since dam removal is increasingly used as a river restoration technique, research efforts generating data from actual removals are not only of interest to the river research community, but also regulators, policy makers, and restoration practitioners. There is much to learn about river responses to dam removal, and the reliable use of dam removal as river restoration is limited by the paucity of available scientific studies.

DAM REMOVAL BIOGEOCHEMISTRY

Among the many facets of dam removal research, geomorphic and biological responses are the most well documented (Stanley et al., 2002; Doyle et al., 2003a and b; Lenhart, 2003; Pollard and Reed, 2004; Sethi et al., 2004; Thomson et al., 2005; Wildman and MacBroom, 2005; Orr and Stanley, 2006). Water quality and/or aquatic biogeochemical responses to dam removal have received less attention, but have been investigated by some (Bushaw-Newton et al., 2002; Doyle et al., 2003b; Ahearn and Dahlgren, 2005; Ashley et al., 2006). In terms of river restoration, upstream responses to dam removal are typically favorable. Among the dam removal biogeochemical literature, one general trend has emerged; impoundments export accumulated sediments, nutrients and organic materials to downstream environments following dam removal. However, the magnitude of impoundment loading among dam removals exhibits considerable variability. For example, a couple of studies found that sediment and nutrient exports following dam removal were an order of magnitude higher than baseline levels (Doyle et al., 2003; Ahearn and Dahlgren, 2005). On the other hand, another study concluded that dam removal had no effect on suspended sediment or nutrient concentrations (Bushaw-Newton et al., 2002). Considering such variability, what then controls the magnitude of sediment and nutrient fluxes exiting former impoundments? Because the export of impounded materials may pose serious threats to downstream biota (Sethi et al., 2004), river restoration practitioners need to better understand and anticipate upstream and downstream responses to dam removal.

America's small impoundments exhibit considerable channel morphology variability. For example, Wisconsin systems typically exhibit wide impounded channels relative to natural channel dimensions (Stanley et al., 2002; Doyle et al., 2003). Conversely, in North Carolina, impounded channel widths are often similar to free-flowing channel dimensions (personal observation). This difference is important because it may affect the retentive capacity of a reservoir, and thus the quantities and magnitudes of sediment, nutrient and organic matter exports following dam removal. Therefore, removal studies must be conducted in many regions for the effective management of future dam removals.

As reservoirs shed materials accumulated over the course of impoundment, the routing of theses materials through downstream environments are not well known. To date, increased fluxes of materials exiting impoundments are infrequently investigated beyond the immediate downstream vicinity of the former dam site. An investigation that attempts to quantify the routing of various materials (particulate and dissolved) through downstream channels following dam removal could provide useful information for restoration practitioners as well river researchers. For the restoration industry, such information is important as the unintended downstream consequences of dam removal during restoration efforts may nullify upstream benefits. Further, it is likely that dissolved, suspended and bed loads released from former impoundments will not exhibit similar spatial ranges of influence within downstream channels because they are transported differently (i.e., advection and/or dispersion). Thus, restoration practitioners operating in a nutrient sensitive watershed may have different concerns that those operating in sediment sensitive waters. This is also of interest to the basic river research community. Dam removal often produces a flood wave carrying dissolved, suspended and bed loads through downstream channels. Such events can provide opportunities to gain considerable insight into the behavior of various materials routed through channel networks during floods.

Sediments released form former impoundments are often characterized by nutrient and organic matter-rich mineral surfaces (Stanley and Doyle, 2002). Thus, there is the potential that sediment suspensions routed through downstream environments can continue to contribute to already elevated water column nutrient and organic matter concentrations. Previous research has suggested that in the presence of light and turbulence, suspended sediments may be a measurable source of dissolved organic carbon to the water column (DOC) (Koelmans and Prevo, 2003; Tietjen et al., 2005; Mayer et al., 2006). In other words, it is possible that dissolved biogeochemical loads released to downstream environments may be underestimated if measured only at the former dam site.

Another area of dam removal biogeochemistry in need of investigation is the role of plant communities colonizing sediment accumulations exposed by dam removal on the fluxes of N and P to downstream environments. Plant communities rapidly colonize sediment accumulations following dam removal (Shafroth et al., 2002; Orr and Stanley, 2006). These accumulations represent potentially appreciable sources of N and P in particulate and dissolved forms. Thus, there is considerable interest regarding the ability of burgeoning plant communities to sequester or otherwise immobilize N and P, limiting downstream nutrient enrichment.

Finally, dam removal is often considered a disturbance to river ecosystems, generating considerable hydrogeomorphic and ecological consequences. There is currently a need for a conceptual framework from which biogeochemical and sediment responses can be viewed in both upstream and downstream directions from the former dam site. Such a model could be useful for both researchers and restoration practitioners. Practitioners wish to better anticipate the consequences of dam removal upstream and downstream, while basic river research may be able to appreciate the bidirectional nature of dam removal disturbances in river ecosystems.

PURPOSE OF DISSERTATION

The intention of this dissertation was to provide insight into both upstream and downstream spatial and temporal heterogeneity of sediment, nutrient and organic matter dynamics following dam removal. Further, it was my intention that all work presented within this dissertation would be useful to those interested in basic river research (biogeochemical, ecological and geomorphic), river and wetland restoration practitioners and policy makers. The broad questions addressed within are as follows:

1. What system-specific features control the magnitude of sediment, nutrient and organic matter fluxes from impoundments following dam removal?
2. How are dissolved, suspended and bed loads routed through river channels during floods and dam removals?
3. How do sediment suspensions affect water column and interstitial biogeochemistry in river ecosystems during floods and dam removals?
4. How do early successional plant communities affect fluxes of N and P from previously inundated, nutrient-rich sediment accumulations exposed by dam removal?

5. How do upstream and downstream disturbances caused by dam removal differ in their structure as well as their spatiotemporal extents?

The questions outlined above were answered using a combination of field and laboratory studies. To address questions 1, 2, 4 and 5, field studies were conducted predominately on the Little River, Johnston County, North Carolina where Lowell Mill Dam was removed in multiple stages from July 2004 to January 2006. Additional field studies were carried out on the Deep River, Chatham, Lee and Moore counties, North Carolina where Carbonton Dam was removed from October 2005 to February 2006. This system was used to address questions 3 and 5. Laboratory studies were conducted to provide controlled conditions to better determine how sediment suspensions could influence water column and interstitial biogeochemistry during floods and dam removals (question 3).

STRUCTURE OF DISSERTATION

This dissertation is written in the form of 5 chapters. This chapter is an introduction, while subsequent chapters were written for the purposes of journal submission. For this reason, there may be some repetition of introductory material. This was done so that each chapter could be submitted as an independent manuscript.

Chapter II documents the multiple stage removal of Lowell Mill Dam from the Little River. It explores structural and regional controls of impoundment sediment, nutrient and organic matter loading to downstream reaches as a result of dam removal.

Additionally, this chapter analyses the fate of dissolved, suspended and bed loads routed through channels during dam removals and floods. Finally, this chapter compares water quality impacts of dam removal to loads carried by low-magnitude floods within the same system.

Chapter III explores the biogeochemical role of suspended sediments in river ecosystems, which exhibit have high affinities for various forms of C, N and P. Sediments are frequently suspended during floods and dam removals, and since they represent significant pools of organic and inorganic forms of C, N and P, they are potentially important for water column biogeochemistry, particularly in the presence of light.

Chapter IV examines the role of plant communities and physical sediment properties on the fluxes of N and P from exposed sediments to adjacent formerly impounded channels following dam removal.

Chapter V explores the conceptual differences among upstream and downstream disturbances following dam removal. Specifically, this chapter demonstrates that there are considerable differences in the nature of upstream and downstream dam removal disturbances. Additionally, such upstream and downstream disturbances differ in their temporal and spatial extents.

REFERENCES

Ahearn DS and Dahlgren RA. 2005. Sediment and nutrient dynamics following a low-head dam removal at Murphy Creek, California. *Limnology and Oceanography* 50: 1752-1762.

Ashley JTF, Bushaw-Newton K, Wilhelm M, Boettner A, Drames G and Velinsky DJ. 2006. The effects of small dam removal on the distribution of sedimentary contaminants. *Environmental Monitoring and Assessment* 114: 287-312.

Bushaw-Newton KL, Hart DD, Pizzuto JE, et al. 2002. An integrative approach towards understanding ecological responses to dam removal: The Manatawny Creek study. *Journal of the American Water Resources Association* 38: 1581-1599.

Doyle MW, Stanley EH, and Harbor JM. 2003a. Channel adjustments following two dam removals in Wisconsin. *Water Resources Research* 36: 1147, doi: 10.1029/2003WR002038, 2003.

Doyle MW, Stanley EH, and Harbor JM. 2003b. Hydrogeomorphic controls on phosphorus retention in streams. *Water Resources Research* 39: 1147, doi:10.1029/2003WR002038, 2003.

Lenhart CF. 2003. A preliminary review of NOAA's community-based dam removal and fish passage projects. *Coastal Management* 31: 79-98.

Orr CH, and Stanley EH. 2006. Vegetation development and restoration potential of drained reservoirs following dam removal in Wisconsin. *Rivers Research and Applications* 22: 281-295.

Pollard AI and Reed T. 2004. Benthic invertebrate assemblage change following dam removal in a Wisconsin stream. *Hydrobiologia* 513: 51-58.

Sethi, SA, Selle AR, Doyle MW, Stanley EH, and Kitchel HE. 2004. Response of unionid mussels to dam removal in Koshkonong Creek, Wisconsin (USA). *Hydrobiologia* 525:157-165.

Stanley EH, Luebke MA, Doyle MW and Marshall DW. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of the North American Benthological Society* 21: 172-187.

Stanley EH and Doyle MW. 2002. A geomorphic perspective on nutrient retention following dam removal. *Bioscience* 52: 693-701.

Thomson JR, Hart DD, Charles DF, Nightengale TL and Winter DM. 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a

Pennsylvania stream. *Journal of the North American Benthological Society* 24: 192-207.

Wildman LAS and MacBroom JG. 2005. The evolution of gravel bed channels after dam removal: Case study of the Anaconda and Union City dam removals. *Geomorphology* 71: 245-262.

CHAPTER II: SEDIMENT, DISSOLVED ORGANIC CARBON AND NITROGEN FLUXES DURING THE DAM REMOVAL PROCESS

ABSTRACT

TSS, DOC and TDN loads were calculated for all stages of the dam removal process at various points upstream, within and downstream of Lowell Mill Impoundment on the Little River, North Carolina. Dewatering produced downstream loads of TSS, DOC and TDN, which were all one to two orders of magnitude less than loads associated with historic floods. Conversely, floods exiting the former impoundment following dam removal produced TSS, DOC and TDN loads comparable to, but slightly greater (1.2 to 1.75 times) than historic floods. Exported loads were greatest following the complete removal of the dam, most likely because of altered channel grade. Additionally, impounded floodplain wetlands were found to contribute the following percentages to total impoundment loads during the dewatering: 44% of stored water, 12.6 % of TSS, 49% of DOC and 33% of TDN. Moreover, the dewatering flood wave was sampled at various points along a 19.2 km reach below the dam to characterize the routing of TSS, DOC and TDN. Excess TSS released by the impoundment was retained within 10 km of the dam, while TDN and DOC loads increased slightly. We used these data to propose the concept of the advective-dispersive continuum, which explains the routing of different physical fractions of materials mobilized from former impoundments (e.g., bed load, suspended load and wash load). Finally, we used our data as well as those from

other removals to provide insight into regional and morphologic controls on exports of impounded materials following dam removal.

INTRODUCTION

Dam Removal and Emerging Science Needs

Dam removal has gained considerable scientific attention over the last several years as many states have begun to recognize the need for removing some of these structures from the nation's drainage network. As America's dams age beyond their intended design lives (Graf, 2005), dam owners are faced with the decision of repair, replacement or removal. Many of these aging structures are without clear titles of ownership, placing associated liabilities on federal, state and local governments. In response to assumed responsibilities, some states in the US have begun to provide incentives for private industry and government entities to remove obsolete, aged structures by accepting dam removal as a means of river restoration. Much of the foundation for these decisions is not based on scientific research, but rather the assumption that dam removal alleviates the well-documented negative impacts dams have on river ecosystems (e.g., impede fish migrations, low dissolved oxygen concentrations, population isolation).

While dam removal is expected to have many positive effects on rivers, the negative consequences are likely considerable, particularly increased sediment and nutrient loads delivered to downstream receiving waters (Stanley and Doyle, 2002, 2003; Doyle et al., 2003; Ahearn and Dahlgren, 2005). Increased sedimentation can result in the smothering of benthic organisms (e.g., Sethi et al., 2004) while concentrated loads of N and P can impair sensitive receiving waters (as determined by the EPA's TMDL standards). Since dam removal is likely to result in downstream disturbances, several questions regarding such perturbations must be resolved in order to use dam removal for river restoration. First, what is the role of floods on impounded material export, and is dam removal a larger disturbance than natural floods? Second, can a multiple stage demolition strategy reduce the export of sediment, organic matter, and inorganic nutrients? Third, what is the spatial extent of disturbance caused by the export of various impounded materials (dissolved vs. particulate) routed through downstream channel networks? Finally, are there regional and structural (i.e. impoundment channel morphology as controlled by dam dimensions) controls on the degree of downstream disturbance following dam removal?

Assessing Impacts of Dam Removal on Downstream Water Quality

Previous dam removal studies have documented the export of stored sediment from former reservoirs during impoundment adjustment processes (Bushaw-Newton et al., 2002; Stanley and Doyle, 2002, 2003; Doyle et al., 2003; Ahearn and Dahlgren, 2005). Other studies have detailed physical, chemical and biological implications of such transport events (Gray and Ward, 1982; Perrin et al., 2000 and Wohl and Cenderelli, 2000; Sethi et al., 2004). To date, most water quality data from removal studies used to document downstream disturbances are presented as time series of concentrations compared to upstream controls (Doyle et al. 2003a; Ahearn and Dahlgren 2005), or as mean concentrations of samples collected during unspecified flow conditions (Bushaw-

Newton et al., 2002). As stored materials within former impoundments are likely exported more effectively during episodic events (flood and stages of dam demolition) with varying water loads, the use of sediment, carbon and nitrogen budgets (i.e. loads) provide greater resolution for drawing inter-event comparisons. Further, impoundment dewatering, breaching or removal may release significant quantities of stored water, and reports of concentrations may mask the resulting water quality impacts, especially when compared to upstream input flows characterized by lower discharges.

Discharge is a master variable driving geomorphic and ecological processes in river ecosystems (Wolman and Miller, 1960; Doyle et al., 2005). Thus, floods should enhance export of impounded materials following dam removal (Stanley and Doyle, 2003; Ahearn and Dahlgren, 2005). In North Carolina, dam removal is typically performed in three stages (i.e., dewatering, breaching and complete removal) with the objective of minimizing downstream perturbations associated with impounded material export. North Carolina's removal strategy may serve as a simple and effective control step reducing downstream loading of sediment, organic matter and nutrients compared to the "blow-and-go" method of past removals. Quantifying impoundment import and export loads following the multiple dam demolition activities and any intermediate floods can provide a basis from which to gage removal-induced disturbances and the magnitude of flood intensification on downstream loading. This approach can enable the evaluation of dam removal as a river restoration mechanism by directly comparing post-removal loading to that of a system's natural disturbance regime.

Once impounded materials are released from a reservoir, concern shifts to the routing of the various materials, which will likely be routed differently through downstream channel networks. For instance, previous studies of sediment loads from impoundments have primarily documented or modeled the bed load transport following dam removal, and these studies have shown that bed load sediment fluxes are concentrated immediately downstream of the dam (Lisle et al., 2001; Wohl and Cenderelli, 2000). Conversely, it is expected that the release of dissolved impounded materials (dissolved C and N, in particular) could be transported well beyond a dam's immediate vicinity, possibly reaching sensitive downstream receiving water bodies such as coastal ecosystems or drinking water reservoirs.

Distinct flood waves are produced during dewatering events, which can be monitored at multiple points during the routing process to quantify the concentrations and/or loads of various transported materials. Such an investigation can provide insight regarding how different physical fractions of materials are conveyed through channel networks during floods. Additionally, potential spatial heterogeneity among sediment and nutrient disturbances caused by dam removal are important to consider when dam removal is used for river restoration.

Downstream disturbances following dam removal may also be influenced by regional and structural controls, which dictate reservoir channel morphology and the system's retentive capacity. Many small dams create reservoirs that are constrained to the river channel width, and thus there should be limited influence of riparian areas on the downstream impacts of dam removal. However, other reservoirs, particularly in low-gradient regions, inundate the channel as well as riparian zones and adjacent floodplains. In these cases, there is an increased chance that sediments and nutrients will be flushed out of these inundated riparian areas, potentially increasing loads released downstream. We lack an understanding of the role of such variable source areas of sediment and nutrients within impoundments recovering from dam removal, and how these areas contribute to loads transported downstream. Thus, it is important to isolate the relative contributions of channel and floodplain sources of water, sediment, and nutrients within a reservoir to downstream loads to better anticipate consequences of dam removal.

Purpose and Structure of Paper

Lowell Mill Dam on the Little River, Johnston County, North Carolina was removed in multiple stages from August 2004 to December 2005 (Figure 2.1). We generated multiple short-term hydrologic, suspended sediment and nutrient budgets at different times during the removal process and at different distances from the removal site to quantify fluxes of total suspended solids (TSS), total dissolved nitrogen (TDN) and dissolved organic carbon (DOC) with four purposes in mind: 1) compare the water quality impacts accompanying the various stages of dam removal. 2) Compare loads associated with dam removal to loads transported in the Little River during floods. 3) Examine the different spatial heterogeneity of disturbance associated with dissolved and particulate materials released from the former impoundment, and 4) compare data from the Little River removal with those of other removals to better understand regional and structural influences on downstream disturbances associated with dam removal.

STUDY AREA

The Little River is a 4th order tributary of the Neuse River located in the lower piedmont and upper coastal plain physiographic regions of North Carolina. Land use within its 600 km^2 drainage basin is comprised of 44% forest, 39% agriculture, 12% wetland and 5% developed. Impoundment bed sediment was a matrix-supported sand and gravel mixture with a thin veneer of fine sediments (<1%) which, during baseline data collection, exhibited fining in the downstream direction (Table 2.1). Bank and floodplain soils were composed of fine sands, silts, and clays supporting bottomland hardwood forest wetland ecosystems.

Lowell Mill Dam, constructed c.a. 1902, was a low-head, run-of-river structure that provided ~3 m of head storage to support grist mill operations. The majority of the 8 km impoundment was confined to ~175,500 m^2 of river channel (Figure 2.1). However, there were two prominent areas where the reservoir permanently inundated adjacent low-lying floodplain wetlands (Figure 1). This accounted for ~ 200,000 m^2 of inundated floodplain wetlands, with an average depth of ~1 m.

METHODS

Dam Removal Phases

In August of 2004 one of two water wheel housing cells was opened by removing a pair of metal flashboards (located on the upstream side of the cell) and breaking the downstream cell wall. The first flash board was removed on 08 August 2004 and the second was removed on 11 August 2004. Less than one meter of head was lost from the impoundment over the four day period. Water surface grade was controlled by the elevation of the downstream cell wall breach, which was ~ 2.25 m above the bed surface grade. The second gate removal event took place on 28 April 2005, at which time two flashboards were removed from the second wheel housing cell in ~ 2-2.5 hours. During this phase, the water surface grade was controlled by the draft hole present at the bottom of the cell, which induced more than 1m of head loss within the impoundment. This second event was the most punctuated head loss and the most intensely studied (referred to as dewatering). The dewatering generated a flood wave that passed through the downstream reach (19.2 km) over a course of ~ 30 hours (Figure 2.2). Shortly after the dewatering (6 days), precipitation produced a flood that filled the banks of the recently dewatered system. The resulting flood wave persisted from 06 May 2005 to 11 May 2005 (Figure 2.2).

The dam was breached-to-grade on 15 December 2005 by completely removing the wheel housing. Two separate precipitation events produced a flood with two distinct discharge peaks (Q_{peak}) with a duration of eight days (Figure 2.2). Shortly following the breaching, Lowell Mill Dam was completely removed from the channel on 28 December 2005. That same evening, precipitation produced another flood that was compounded by a second precipitation event on 02 January 2006. Together the precipitation events produced a flood which persisted for 13 days (Figure 2.2).

Sampling Design

Sampling locations

We generated multiple short-term hydrologic, suspended sediment and nutrient budgets at different times since removal and at different distances from the removal site (Figure 2.1). This approach treated the impoundment as a distinct spatial unit with one major channel input (-11.6 km upstream from dam; note, negative values indicate distance upstream from dam), 2 wetland inputs (-3 km and -2 km from dam) and one output (+0.4 km from dam). Within the impoundment, sampling stations were positioned upstream, within and downstream of both confluences with the river and wetland complex (referred to as wetland stations; Figure 2.1 inset). Downstream bridge stations (+1.6 km, +4.2 km, +9.6 km and +19.2 km; Figure 2.1) were used to monitor sediment and nutrient routing through the downstream reach. Wetland and bridge stations were sampled during the dewatering event only, while impoundment input (-11.6 km) and output (+0.4 km) stations were sampled during every event (Table 2.2). USGS stream gage station #02088500 is located +10 km downstream of the dam (Figure 2.1), and was used to generate reach-scale hydrologic budgets for all events.

Dewatering event

Sampling at river input and output stations (-11.6 km and +0.4 km) was accomplished using automated water samplers (Teledyne-ISCO 6712) which extended the temporal extent of data collection to cover the entire 30-hr event. River input and output samples were collected at least bihourly before, during and after the dewatering. Other stations were staffed with at least 2 people on 28 April 2005 who collected samples hourly for the first 8 hours of the dewatering. All stations produced a number of samples during each sample period (described in detail below) for TSS, DOC, TDN, and dissolved inorganic nitrogen (DIN), which includes NH_4-N, NO_3 -N and NO_2-N. Sampling efforts at bridge stations were maintained for the duration of the dewatering (30 hours), although the frequency decreased from hourly during the first 10 hours to every other hour thereafter. Wetland sampling efforts were initiated before gate removal, and continued until all surface water drained into the channel as determined by Q measurements (described in detail below).

Breaching, removal, flood events

TSS, DOC, DIN and TDN sample collection was limited to impoundment input and output stations only. Samples were collected from 05 May 2005 to 11 May 2005 at 2-hour intervals for the post-dewatering flood. Sampling during the breaching/flood was conducted from 15 December 2005 to 23 December 2005 at no more than 4-hour intervals. Likewise, the removal/flood was sampled at no more than 4-hour intervals from 28 December to 09 January 2006. Sampling covered the entire duration (beginning of rising limb to end of descending limb) of each event. Separate load calculations were not performed for breaching or removal and the floods which followed for two reasons: 1) stored water was not released by either the breaching or removal events because of previous dewatering, and 2) floods were seen immediately following both events (less than 24 hours).

Hydrology

The USGS gage (#02088500) located 10 km below Lowell Mill Dam was used to quantify discharge (*Q*) from 1 April 2005 to 31 January, 2006 (Figure 2.2). No perennial tributaries contribute significantly to river discharge for the 19.2 km study reach downstream of the dam. USGS 15-min interval data were used to produce hydrographs associated with dam demolition and flood events. During the dewatering, gage data were used to quantify total volume released from the impoundment, and the duration of the dewatering flood wave. We assumed the same hydrologic budget for all bridge stations during this event. During all other monitored events, gage data were used to quantify the volume of water that passed through the impoundment.

Hydrological budgets of the dewatering wetlands and upstream river inputs were produced using channel cross sectional area and velocity measurements. Upstream river input measurements were collected hourly at station -11.6 km. Dewatering wetland measurements were collected at stations located immediately upstream and downstream of the wetland confluences. Channel cross sectional surveys were completed before the dewatering from 25 April 2005 to 27 April 2005) during which time flows were near baseflow discharge values (~1.8 m^3s^{-1}). During the dewatering, area and velocity measurements were made at each wetland sampling station (Figure 2.1 inset). Velocity measurements were collected using either a *Marsh-McBirney* current meter or a *Sontek* acoustic Doppler velocimeter (ADV). Velocity measurements were taken at 60% of depth at 5 points across each channel cross section; time did not permit more detailed measurements. Differences between *Q* values measured upstream and downstream of each wetland confluence were assumed to be impounded wetland surface water draining into the river channel.

Total suspended solids

Impoundment input and output stations were sampled from 1 L composites at 1-2 hour intervals. During the dewatering event, a 250 mL HDPE bottle was filled from a 4 L composite of grab samples collected from 5 points across each bridge station cross section during each sample period. Wetland sampling teams collected 250 mL grab samples from the channel thalweg. TSS samples were shaken and known volumes were filtered through pre-weighed glass fiber filters (ProWeigh filters, Environmental Express), dried at 110^{o}C for at least 24 hours, desiccated and reweighed for TSS (APHA Standard Methods procedure 2540D).

Biogeochemistry

All biogeochemistry water samples were filtered using Whatman GF/F (0.7μm) glass fiber filters. Samples were filtered directly into acid washed 125 mL amber HDPE bottles for DIN analyses. Remaining filtrate was transferred to glass total organic carbon (TOC) vials pretreated with 600 μL of 2M HCl for DOC and TDN analyses. Samples were placed on ice during transport from the field to the laboratory where DIN samples were frozen at -20^{o}C and TOC vials were refrigerated at 4^{o}C until analyses were performed. DIN analyses were performed by Water Agricultural Laboratories in Camilla, GA using US EPA methods (250.1 for NH_4-N and 353.1 for NO_2-N and NO_3-N). DOC and TDN analyses were performed in-house using a Shimadzu TOC-V CPH analyzer coupled with a TNM-1 unit. Dissolved organic nitrogen (DON) fractions of each sample were determined by subtracting DIN concentrations from TDN concentrations.

Budget calculations

Budgets for TSS, DOC and TDN were calculated by multiplying concentrations by water load for each time interval. Fluxes for all sampling periods were summed across an event to get total mass at a given sampling station. Concentration data at river input and output stations were analyzed for significant differences using two-tailed, paired t-tests. Analyses were performed for each event for TSS, DOC and TDN.

RESULTS

Hydrology

The impoundment dewatering produced a small downstream flood wave that lasted ~ 30 hours with a Q_{peak} of 3.2 m^3s^{-1}, which is insignificant compared to common floods associated with the system's natural flow regime (Figure 2.2). Upstream values remained constant throughout the dewatering at 1.7 m^3s^{-1}. Integration of the dewatering hydrograph showed that ~ 40,100 m^3 of stored water was released from the impoundment, compared to 183,600 m^3 of surface water conveyed by incoming river flows (1.7 m^3s^{-1} over 30 hours). The wetland sites contributed a combined 17,700 m^3 of surface water to the channel during the dewatering, representing 44% of the total stored water released from the impoundment.

Six days after the dewatering, precipitation generated a near bankfull flood. The flood occurred from 06 May 2005 to 11 May 2005 with a Q_{peak} of 20.2 m^3s^{-1} on 07 May 2005, or 5.1 x 10^6 m^3 of water passing through the impoundment during the post-dewatering event. This flood was within the Q range of a one year flood (18-32 m^3s^{-1}; based on 72 years of gage records).

Immediately following the breaching of Lowell Mill Dam, two precipitation events produced high discharges. High Q levels occurred from 15 December 2005 to 25 December 2005 with two distinct Q_{peak} values on 17 December 2005 (11.4 m^3s^{-1}) and 19 December 2005 (11.7 m^3s^{-1}). Based on hydrograph integration, ~ 6.07 x 10^6 m^3 of water passed through the study reach over the 10 day post-breaching period.

Similar to the breaching, a precipitation event immediately followed the removal of the remaining structures of Lowell Mill Dam (28 December 2005). Initial precipitation was followed by additional rain on 02 January 2006. The resulting flood produced two distinct Q_{peak} values 30 December 2005 (13.37 m^3s^{-1}) and 04 January 2006 (11.92 m^3s^{-1}). This flood transported 8.67 x 10^6 m^3 of surface water through the reach during the post-removal period.

Total suspended solids

Before dam removal was initiated, mean reservoir retention of TSS was 50 $\pm$ 19% (input > output; $n = 10$, $t = 2.4368$, $p = 0.022$). The system continued to store TSS until the flood gates were opened for reservoir dewatering (28 April 2005). Immediately following gate removal, the system began to export significantly more TSS from the impoundment than upstream sources imported (Figure 2.3a). Mean TSS input concentrations following gate removal were 3 mg/L, while mean export concentrations were 8 mg/L ($n = 38$, $t = 10.2746$, $p < 0.001$; Figure 2.3e). Over the course of the dewatering, TSS loads entering the impoundment were 658 kg compared to 1735 kg exiting (Figure 2.3f), resulting in a TSS_{out}:TSS_{in} ratio of 2.43 (Table 2.3). The floodplain wetlands accounted for 136 kg (13%) of the 1077 kg of TSS transported from the impoundment whereas the wetlands contributed 44% of the water load. Thus, based on the wetland water load contribution, the wetlands diluted the TSS concentrations exiting the impoundment.

Excess TSS routed downstream of the dam was effectively attenuated within the 10 km (Figure 2.4). Of the 1735 kg of TSS exiting the reservoir, only 712 kg passed the bridge station +9.6 km. This load is comparable to the load entering the upstream end of the reservoir, 658 kg. Thus, excess TSS derived from the impoundment was deposited within 10 km of the dam.

TSS outputs were initially greater than TSS inputs during the post-dewatering flood (Figure 2.3b). This trend was reversed around the 24th hour of data collection as open flood gates were unable to conduct incoming discharge, creating backwater conditions (i.e., the dam was still in place). As a consequence, 24% of TSS inputs were stored within the impoundment. The input load of TSS was 127 metric tons (127×10^6 g) compared to an output load of 96 metric tons, and mean concentrations were 21 and 16 mg/L, respectively (input > output; $n = 94$, $t = -6.4523$, $p < 0.001$; Figure 2.3e and 2.3f). Effectively, the impoundment stored 31 metric tons of TSS during the post-dewatering flood. This is ~30 times the TSS load exported during the dewatering event; $TSS_{out}:TSS_{in}$ ratio equaled 0.76 (Table 2.3).

During the breaching/flood, output concentrations were significantly higher than input concentrations ($n = 62$, $t = 6.826812$, $p < 0.001$). The Little River carried 75 metric tons of TSS into Lowell Mill Impoundment with a mean concentration of 13 mg/L. Over the same time period, 104 metric tons of TSS were exported, with a mean concentration of 19 mg/L, a 29 metric ton enrichment (Figures 2.3c, 2.3e, and 2.3f); $TSS_{out}:TSS_{in}$ ratio equaled 1.37 (Table 2.3).

During the removal/flood, river inputs delivered 124 metric tons of TSS to the former impoundment with a mean concentration of 13 mg/L, while 218 metric tons with a mean concentration of 23 mg/L were exported from the impoundment. Output concentrations were significantly higher than input concentrations ($n = 68$, $t = 7.7507$, $p < 0.001$). The former impoundment provided an additional 96 metric tons of TSS to the Little River (Figures 3d, 3e and 3f); $TSS_{out}:TSS_{in}$ equaled 1.76 (Table 2.3). This represents the greatest concentration and load of TSS exported from Lowell Mill Impoundment during the course of this study.

Dissolved organic carbon

Baseline sampling for DOC was limited to 7 paired samples, and differences between input and output data sets were not significant ($n = 7$, $t = 0.87$, $p < 0.5$). The mean baseline concentration entering the impoundment was 7.8 mg/L, and 7.0 mg/L exiting the impoundment. During the dewatering, upstream inputs of DOC produced a total load of 1094 kg with a mean concentration of 6.4 mg/L, while downstream DOC exports totaled 1517 kg of DOC with a mean concentration of 7.2 mg/L (Figures 2.5a, 2.5e and 2.5f). DOC output concentrations were significantly higher than inputs ($n = 45$, $t = 7.58$, $p < 0.001$); DOC_{out}:DOC_{in} equaled 1.39 (Table 2.3). This represents a 39% increase in the DOC load. Wetland contributions of DOC accounted for 209 kg, or 49% of the total contributed impoundment DOC load compared to 44% of the total water load coming from the wetlands. Thus, impounded floodplain wetlands were slightly concentrated sources of DOC as these areas contributed 44% of the total water load. This DOC load increased from 1517 kg at the output station to 1576 kg at the +4.8 km bridge and 1533 kg at the +19.2 km bridge (Figure 2.6). That is, the DOC load exported from the impoundment did not decrease, but increased slightly during downstream routing.

Output DOC concentrations during the post-dewatering flood were significantly higher than input concentrations ($n = 96$, $t = 11.15$, $p < 0.001$; Figure 2.5b). Initially (during the early phase of the rising limb), upstream and downstream concentrations were nearly equal (~ 6.4 mg/L). The upstream DOC load during the six day flood was 38 metric tons with a mean concentration of 7.3 mg/L. The downstream DOC load was 44 metric tons with a mean concentration of 8.2 mg/L (Figures 2.5e and 2.5f). This represents an enrichment of 17% during the post-dewatering flood.

The breaching/flood produced another pulse of DOC with input and output loads totaling 41 and 48 metric tons, respectively (Figures 2.5c and 2.5f). Output concentrations were significantly higher than input concentrations ($n = 64$, $t = 6.83$, $p < 0.001$). Mean input concentrations were 7.3 mg/L, and mean output concentrations were 8.3 mg/L (Figure 2.5e). This event caused a DOC enrichment of 17%.

During the removal/flood, the input DOC load totaled 60 metric tons, while the output load totaled 72 metric tons (Figures 2.5d and 2.5f). Downstream concentrations (mean = 8.5 mg/L) were significantly higher than upstream input concentrations (mean = 7.1 mg/L; $n = 80$, $t = 18.29$, $p < 0.001$; Figure 2.5e). This represents the greatest load of DOC from the former impoundment, and an enrichment of 12 metric tons or 20%.

Dissolved nitrogen

Baseline data for TDN were limited; mean upstream concentrations were 0.72 mg/L while mean downstream concentrations were 0.54 mg/L ($n = 7$; $t = 0.96$, $p < 0.2$). During the dewatering, TDN concentrations exiting the impoundment were significantly higher than upstream inputs, as the mean upstream TDN concentration was 0.62 mg/L compared to 0.70 mg/L downstream ($n = 45$, $t = 2.52$, $p = 0.029$; Figures 2.7a and 2.7e). Upstream and downstream loads of TDN were 105 kg and 147 kg, respectively. The impoundment contributed 42 kg of TDN, which represents an enrichment of 40% (Figure

2.7f). Wetland inputs during the dewatering accounted for 14 kg, or 33%, of the impoundment's 42 kg contribution. While the dewatering wetlands were a considerable source of TDN, they were diluted compared to channel sources. TDN released from the impoundment during the baseflow dewatering event was comprised of 95% DIN, while wetland TDN entering the channel was 69% DIN. Downstream routing increased TDN loads from the dam to the +19.2 km bridge (Figure 2.8). TDN loads exiting the impoundment were 147 kg, while the load at the 19.2 km bridge was 160 kg. Similar to DOC routing, the TDN load did not decrease, but actually increased slightly with distance downstream.

Post-dewatering flood flows were characterized by a mean upstream TDN concentration of 0.59 mg/L, which was significantly less than the mean downstream concentration of 0.67 mg/L ($n=96$, $t = 8.78$, $p < 0.001$, Figures 2.7b and 2.7e). Input loads equaled 2882 kg while output loads were 3587 kg (Figure 2.7f). TDN exiting the impoundment was approximately 45% DIN. Impoundment derived N enriched channel waters by 24%.

Impoundment contributions of TDN during the breaching/flood loaded an additional 1.0 metric ton (Figure 2.7f) to channel surface waters; the TDN load delivered to the impoundment equaled 2.8 metric tons compared to 3.8 exiting the impoundment. The mean upstream TDN concentration was 0.51 mg/L, which was significantly less than the mean downstream concentration of 0.67 mg/L ($n = 64$, $t = 19.02$, $p < 0.001$; Figures

2.7c and 2.7e). 57% of the TDN entering the impoundment was comprised of DIN, while DIN accounted for 59% of the TDN exiting.

The removal/flood produced the greatest TDN loading in the series of monitored events. TDN loads entering and exiting the former impoundment equaled 3.7 and 5.0 metric tons, respectively (Figures 2.7d and 2.7f). Again, mean concentrations entering the former impoundment (0.45 mg/L) were significantly lower than those exiting the impoundment (0.61 mg/L; $n = 80$, $t = 21.02$, $p < 0.001$; Figure 2.7e). Impoundment contributions of TDN during this event enriched river water by 35%. DIN represented 29% of TDN exported from the impoundment.

DISCUSSION

Removal strategies and water quality impacts: dewatering to removal

Dam removal increased the transport capacity of the reach as the system's grade was progressively increased from the dewatering to the removal. Each flood routed comparable quantities of water (i.e. within the same order of magnitude) through the former impoundment, yet exported TSS, DOC and TDN loads following each event were progressively greater as dam removal progressed (Figures 2.3f, 2.5f and 2.7f). By far, the greatest export was seen during the flood following the complete removal of Lowell Mill Dam. It was at this point that the dam no longer offered grade control, so the transport capacity of the system was not influenced by backwater effects. Additionally, incoming flows were supply limited in reference to TSS, DOC and TDN, so the river carried greater loads as supplies were made available. Based on these observations, loads exported from the former impoundment appeared to be a function of the degree of grade change and *Q*. Grade adjustments caused by dam removal increase channel slope which in turn increases velocity and thus, transport capacity within these significantly aggraded reaches. Increased transport capacity erodes (and suspends) accumulated sediments, which can also lead to the release of TDN and DOC from the sediment matrix.

Previous dam removals have been conducted without the use of a dewatering step. Such removals involving a full reservoir may use either the “blow and go” method of complete demolition using explosives (e.g. Embrey Dam, VA; USACE, 2004), or the breach-to-grade approach (e.g., Rockdale Dam, WI; Doyle et al., 2003). If impoundments with significant retentive capacities are subjected to these removal strategies, there will be appreciable loads of stored materials released to downstream environments. Such systems impound significant quantities of water which will be immediately subjected to drastic grade alterations. Further, the materials contained within impounded channels are often completely saturated, and are therefore easily suspended and transported downstream.

Context for comparing water quality impacts of dam removal

Our results are similar to those of previous studies: dam removal results in increased concentrations and loads of sediment, organic matter, and inorganic nutrients (Doyle et al., 2003; Ahearn and Dahlgren, 2005). Decisions to use dam removal for river restoration should be made under the assumption that such increases are likely to follow most dam removals. There are some additional lessons provided by this study that are helpful to consider during the dam removal planning process. We assert that the disturbances (i.e. concentrations and loads) associated with dam removal should be compared to those during floods within the same system, rather than comparing the impacts of dam removal with baseflow conditions. For example, for the Little River during our study period, the maximum natural loads of TSS, DOC and TDN (i.e., the maximum input loads observed during any of the multiple floods reported here) were 127, 60, and 3.7 metric tons, respectively (Figures 2.3f, 2.5f and 2.7f). In comparison, the maximum loads generated by dam removal were 218 (TSS), 72 (DOC) and 5 (TDN) metric tons, all of which were released during the removal/flood event. This translates to ratios of removal export to maximum natural loads for similar storm events of 1.71 for TSS, 1.2 for DOC and 1.35 for TDN. Thus, floods intensified the export of TSS, DOC and TDN from the impoundment. This flood intensification, however, resulted in only modest levels of enrichment; this will not necessarily be the case for all dam removals (discussed below).

Routing of dissolved versus particulate loads through downstream channels

We conclude, based on the routing data set, that particulate and dissolved materials released from a reservoir are routed differently through downstream channel networks (Figure 2.4, 2.6 and 2.8). These data suggest that the mixture of materials transported by flood waves exhibit spatially heterogeneous patterns of transport and deposition (Figure 2.9). Different sized particles are transported by separate mechanisms, e.g., suspension of fine constituents, saltation of coarse constituents, and washload transport of dissolved constituents. When a dam is removed, materials stored within the reservoir will be delivered downstream as a function of these different mechanisms and their respective rates of transport, which will in turn affect the spatial impacts of dam removal on downstream reaches, particularly, the concentration or load of sediment or nutrients observed with distance downstream from a removed dam. How these different constituents are transported can be thought of along a continuum of dispersion to advection. If constituents are transported primarily through advection, then the concentrations will be large in the downstream direction, with some decrease in peak concentration, but not in total load with distance downstream from the removed dam, although the elevated concentration will also be brief. In contrast, if constituents are dispersed as they are translated (e.g., a dispersive sediment wave), then the peak concentration and total load following dam removal will decrease with distance downstream from the dam, although the duration of elevated concentrations will extend over a longer period of time.

Previous studies have shown that pulses of coarse sediment (bed load), like sediment introduced from a dam removal or a large land-slide, are transported through fluvial systems as dispersive waves, with very little advection of the sediment wave downstream (Lisle et al., 2001). In these conditions, the maximum impact of dam removal will be seen immediately downstream of the dam, and then limited impacts with increased distance downstream (Wohl and Cenderelli, 2000). Our data suggest that, on the opposite end of the continuum, dissolved constituents will experience much greater advection in comparison to dispersion, and thus, assuming limited-to-no biological uptake or in-channel storage, the total loads of dissolved sediment or nutrients would be expected to remain constant with distance downstream. If dissolved and bed load materials represent opposite ends of the dispersive-advective continuum, then suspended loads mobilized by dam removal should be subjected to both advection and dispersion, and thus experience some combination of advection and dispersion.

TSS load calculations at various points along the 19.2 km reach downstream of Lowell Mill Dam show a reduction by an order of magnitude within the first 10 km (Figures 2.4 and 2.9). Similarly, a reduction of TSS concentrations was reported over a 4 km distance following dam removal on the Koshkonong River (Doyle et al., 2003a). Therefore, the sedimentation associated with suspended material represent a diffusive pattern with greater impacts seen in close proximity to the former dam site, and limited impacts with distance downstream. Part of the reduction in suspended sediment concentration may be associated with dispersion, whereas part of it could be the result of retention of sediment within the reach (i.e., load reduction). Either way, the impact of suspended sediment mobilization following dam removal appeared to be greatest in the $\sim 10^0 - 10^1$ km downstream of removed dams.

At the Little River dam removal, dissolved constituents of mobilized impoundment materials (i.e. DOC, TDN) exhibited a distinctly advective behavior (Figures 2.6, 2.8 and 2.9). DOC and TDN loads in the dewatering flood wave remained fairly constant during downstream routing. Essentially, dissolved loads associated with the dewatering flood wave were transported well beyond the study reach, and perhaps to the Little River's receiving waters (the Neuse River). This can be of considerable importance as TMDLs for downstream environments could be violated by dam removal activities occurring far upstream if advection is dominant and if there is very little loss of constituents with distance through the channel network. For N in particular, this final point is expected for all dam removals taking place on larger rivers, but significant reductions in dissolved loads could be seen in smaller systems (Alexander et al., 2000).

Based on the available studies and our current understanding of solute and sediment transport in channel networks, it appears that dissolved constituents will be transported by floods and dam removals over channel distances of $\sim 10^1$ km with little change in the peak concentrations or total loads transported (i.e., limited retention). Over similar distances from a removed dam, there will be transport of suspended sediment, but an increasing degree of both retention (i.e., load removal via deposition in the channel) as well as dispersion of the sediment. This will result in reduced peak concentrations and loads of suspended sediment with distance downstream. Finally, bedload should be primarily dispersed and retained within the first few km of a removed dam, with drastic spatial changes in the peak bedload concentrations with distance downstream and limited impacts to further downstream reaches.

Regional and structural controls on downstream impacts of dam removal

One of the prevailing interests in dam removal involves the unintended disturbances to downstream environments (i.e. increased sedimentation and nutrient loads delivered to receiving waters), and what potentially influences their severity. Previous research has provided limited data suggesting large TSS effects on downstream environments (Doyle et al. 2003, Bushaw-Newton, 2001; Ahearn and Dahlgren, 2005). Doyle et al. (2003) found, TSS_{out}:TSS_{in} >14 during the 48 hours following dam breaching (Table 4). In addition, Ahearn and Dahlgren (2005) quantified annual TSS loads for Murphy Creek, California showing that annual TSS loads after removal were 27 to 35 times greater than annual pre-removal loads (Table 2.4). Conversely, TSS loads downstream of Lowell Mill Dam (this present study) were not as drastic as seen on Murphy Creek or Koshkonong River, and were more comparable to Manatawney Creek, Pennsylvania, in which sediment concentration changes were not detected (Table 2.4). In fact, the greatest TSS_{out}:TSS_{in} during the Little River removal was 1.76 (Table 2.3), an order of magnitude less than those load ratios seen for either Murphy Creek or the Koshkonong River. In the case of N, Murphy Creek post-removal TN loads were 7.75 to 7.82 times greater than pre-removal loads. While the maximum Little River TDN_{out}:TDN_{in} was 1.4. The Manatawny Creek dam removal reportedly had no effect on spatial variations of water chemistry (e.g. upstream and downstream measures for inorganic N and P were not significantly different before or after dam removal). Thus, based on TSS and available N data, there appears to be wide discrepancy in the amount of materials removed from reservoirs following dam removal, with the Koshkonong and Murphy Creek removals representing extremely high export loads, and the Manatawny Creek and Little River removals representing relatively limited export loads.

It is important to note some limitations in the comparisons of our results with those from previous studies. First, Ahearn and Dahlgren (2005) present annual loads, while we present event specific loads, which could ignore important seasonal relationships regarding sediment and N budgets. Second, TN and TDN are different physical fractions of N. However, while such comparisons are limited, they do provide valuable insights into regional controls on impoundment loading.

Differences in the exported loads of TSS and N presented above may be explained by both regional and structural controls. Sediment grain size distribution, watershed land use and the retentive capacity of reservoirs are important factors which could control downstream impacts following dam removal (Stanley and Doyle, 2002). The Koshkonong River is located within an agricultural watershed with a contribution area of 360 km^2. The sediments within the impounded Koshkonong reach were composed primarily of fine sand and silt. Because the reservoir was > 150 years old in an agricultural watershed and because its impoundment was much wider than the main river channel (impoundment width > 200 m compared to river width ~ 15 m), it was completely filled with a large quantity of very fine-grained, nutrient-rich sediment. When the dam was removed the reservoir became a substantial source of fine sediment to downstream reaches. Murphy Creek drains a 12 km^2 watershed dominated by cattle-grazing and viticulture, and the impoundment ws also much wider than the stream channel (Ahearn and Dahlgren, 2005).   Post-removal data presented for the site located closest to the dam reveal sediments were dominated by sand and silt, and thus were somewhat similar to the Koshkonong conditions. In contrast, Manatawny Creek watershed drains 238 km^2; land use was approximately 54% forest, 41% agriculture, and 3% urban, and the impoundment was not much wider than the river channel. Impoundment bed sediments on Manatawny Creek consisted primarily of sand and gravel. Thus, suspended sediment loads following dam removal on the Manatawny were limited compared to the Koshkonong River and Murphy Creek removals, but similar to our data.

While land use and geology should influence the magnitude of sediment transport during dam removal, our results also emphasize the effects of the reservoir itself in affecting these loads. Based on these studies and our own results, it appears that watershed land use and local geology dictate sediment budgets, grain size distributions and water chemistry following dam removal. Watershed land use can obviously result in large sediment loads, but sediment loads following dam removal on the Little River were only slightly greater than the loads entering the reservoir, despite the Little River being in a watershed of high sediment erosion (agricultural basin). Rather than being completely driven by land use and regional geology, the retentive capacity of each impoundment also exerts structural control on the volume of materials stored over the life of the reservoir, which may become mobile following dam removal. This was apparently the primary difference between the two sites with limited sediment export (Little River and Manatawny Creek), and those with large amounts of export (Koshkonong River and Murphy Creek). For the Little and Manatawny removals, both reservoirs were primarily contained within the widths of the main channel. We suggest that under these conditions, there will be a relatively limited amount of fine sediment that accumulates within the reservoir, particularly in comparison to the suspended sediment loads that would be delivered to the channel from upstream sources. In contrast, when a reservoir is much wider than the channel, greater amounts of suspended sediment can be stored laterally over time, and when the dam is removed, these sources of sediment are accessed, and result in substantially larger exports downstream. In all, we suggest that in addition to land use and regional geology, the width of the reservoir relative to the width of the river channel is a potential first indicator of the relative impact of removal in comparison to loads brought in from upstream.

Regional watershed conditions and reservoir channel morphology dictate whether impounded water is stored solely within the channel. As stated earlier, low-gradient impoundments may also store water within local flood plains. We were able to determine the degree to which variable source areas (channel versus floodplain environments) of sediments and nutrients within the impoundment contributed to the overall loads released to downstream environments during the dewatering process. Of the total loads released from the impoundment, the inundated wetlands contributed 44% of the stored water, 13% of the TSS, 49% of DOC and 33% of TDN. Thus, floodplain source areas represented concentrated sources of DOC, and diluted sources of TSS and TDN. Note that Lowell Mill Impoundment's morphology was unique as it was composed of distinct channel and floodplain segments (Figure 2.1). This distinction is important because other reservoir systems storing water on flood plains do so by producing a broad reservoir channel (i.e., Koshkonong River and Murphy Creek). These morphological differences affect the frequency and magnitude of shear stresses fine sediments are exposed to following dam removal. On the Little River, the floodplains are infrequently subjected to such erosive forces, but in the cases of the Koshkonong River and Murphy Creek, floodplain soils are continuously subjected to river shear stresses. Thus, these systems should be expected to contribute more TSS, TDN and DOC from floodplain sources, ultimately resulting in greater loads delivered to downstream environments.

CONCLUSIONS

Dam removal is considered a disturbance throughout the removal literature, but direct comparisons to other natural disturbances have not been drawn with actual data previous to this study. Such comparisons are needed to aid in decision making for the numerous dam removals likely to occur in coming years. We found that while floods on the Little River intensified the export of previously impounded materials, the resulting loads carried downstream were comparable to those of natural floods (Table 2.3). This is not the case on all rivers, but this comparative approach can provide context by which the downstream impacts of future removals can be judged.

When viewed as a disturbance, dam removal can initiate fundamental alterations within river ecosystems along various temporal and spatial scales. If a secondary disturbance follows an event such as dam removal, the resistance of the system is likely reduced, forcing changes in the physical structure of these former impoundments. Doyle et al. (2002) offer a conceptual model of geomorphic change induced by a flood following dam removal. The model simply states that the degree of geomorphic change caused by a flood event is a function of time since the removal. As the reservoir sediments stabilize, geomorphic parameters gain resistance. This model can also be used to explain potential biogeochemical loads following dam removal (Stanley and Doyle, 2002). Much of the biogeochemistry associated with dam removal is influenced, and to some extent controlled, by the size and quantity of sediments stored within reservoirs. If sediments are not stabilized, their reorganization or mobilization can release organic and inorganic forms of C, N and P to downstream environments.

The objectives of river restoration projects are important to consider when determining the timing of demolition activities (i.e., dewatering, breaching and removal). Because floods occurring during the sensitive recovery period following dam removal intensify the export of impounded materials, these episodic events are expected to have strongly divergent effects at different points along the river. In such situations, the same flood can accelerate the recovery of the former impoundment by excavating accumulated materials, while subjecting the downstream environment to a possibly catastrophic disturbance, forcing the downstream system to assume a new steady state. Thus, removal activities should be scheduled such that local seasonal hydrology does not interfere with restoration objectives. If, for example, a dam removal is used to restore channel habitat within an impoundment, then floods will accelerate the rehabilitation process. However, if removal activities are conducted during low flow seasons, recovering reaches may permanently retain significant fractions of impounded materials.

We encourage the consideration of structural, regional and seasonal controls on downstream disturbances when designing removal strategies. Considerations should include: impoundment retentive capacity, sediment budgets and size distributions (both upstream and downstream of the dam), watershed land use, seasonal hydrological patterns and general proximity to sensitive receiving waters. Further, the dispersive-advective continuum (Figure 2.9) presented above can be used to better determine how and which downstream communities will be affected by dam removal. Additionally, management strategies for dam removal should consider that the downstream impacts following dam removal can be reduced by performing removals in multiple stages.

REFERENCES

Ahearn DS and Dahlgren RA. 2005. Sediment and nutrient dynamics following a low-head dam removal at Murphy Creek, California. *Limnology and Oceanography* 50: 1752-1762.

Alexander RB. Smith RA. and Schwarz GE. 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. *Nature* 403: 758-761.

Bushaw-Newton KL, Hart DD, Pizzuto JE, et al. 2002. An integrative approach towards understanding ecological responses to dam removal: The Manatawny Creek study. *Journal of the American Water Resources Association* 38: 1581-1599.

Doyle MW, Stanley EH, and Harbor JM. 2003. Channel adjustments following two dam removals in Wisconsin. *Water Resources Research* 36: 1147, doi: 10.1029/2003WR002038, 2003.

Doyle MW, Stanley EH, Strayer DL, Jacobson RB, Schmidt, JC. 2005. Effective discharge analysis of ecological processes in streams. *Water Resources Research* 41: W11411, doi:10.1029/2005WR004222, 2005.

Graf WL. 2005. Geomorphology and American dams: The scientific, social, and economic context. *Geomorphology*, 71(1-2): 3-26.

Gray LJ, and Ward JV. 1982. Effects of sediment release on macroinvertebrates. *Hydrobiologia* 96: 177-184.

Lisle, TE, Cui Y, Parker G, Pizzuto JE, and Dodd AM. 2001. The dominance of dispersion in the evolution of bed material waves in gravel-bed rivers. Earth Surface Processes and Landforms 26: 1409-1422.

Perrin CJ, Ashley KI, and Larkin GA. 2000. Effects of drawdown on ammonium and iron concentrations in a coastal mountain reservoir. *Water Quality Research Journal of Canada* 35: 231-44.

Sethi, SA, Selle AR, Doyle MW, Stanley EH, and Kitchel HE. 2004. Response of unionid mussels to dam removal in Koshkonong Creek, Wisconsin (USA). *Hydrobiologia* 525:157-165.

Stanley EH, and Doyle MW. 2002. A geomorphic perspective on nutrient retention following dam removal. *BioScience* 52: 693-701.

Stanley EH, and Doyle MW. 2003. Trading off: the ecological effects of dam removal. *Frontiers in Ecology and the Environment* 1: 15-22.

United States Army Corps of Engineers. 2004. Embrey Dam removal media kit. http://www.nao.usace.army.mil/Projects/embrey/Docs.htm

Wohl EE, and Cenderelli DA. 2000. Sediment deposition and transport patterns following a reservoir sediment release. *Water Resources Research* 36: 319-333.

Wolman MG and Miller JP. 1960. Magnitude and frequency of forces in geomorphic processes. *Journal of Geology* 68: 54-74.

Table 1: Bed surface grain size analysis[a]

Distance from dam (km)[b]	D_{16} (mm)	D_{50} (mm)	D_{84} (mm)
-9.7	0.25	8	64
-6.4	0.5	2	16
-3.2	0.5	16	64
-1.6	0.25	0.5	2
-0.8	0.25	0.5	0.5
-0.4	0.125	0.25	0.25
0.4	0.5	16	64
0.8	0.25	0.5	8
1.6	0.5	64	64
3.2	2-4	32	64
6.4	0.5	8	32
9.7	0.5	0.5	8

[a] D16, D50 and D84 values were determined based on mass
[b] negative values denote site located upstream of dam

Table 2: Sampling station use summary

Station Location (km from dam)	Events Sampled	Frequency of sample collection	Parameters sampled
-11.6 (input)	1, 2, 3 and 4	1,2 or 4-hour intervals	TSS, DOC, TDN, Q
-3 (wetland)	1	1-hour intervals	TSS, DOC, TDN, Q
-2 (wetland)	1	1-hour intervals	TSS, DOC, TDN, Q
+0.4 (output)	1, 2, 3 and 4	1,2 or 4-intervals	TSS, DOC, TDN
+1.6 (bridge)	1	1 to 2- hour intervals	TSS, DOC, TDN
+4.8 (bridge)	1	1 to 2-hour intervals	TSS, DOC, TDN
+9.6 (bridge)	1	1 to 2-hour intervals	TSS, DOC, TDN
+19.2 (bridge)	1	1 to 2-hour intervals	TSS, DOC, TDN

1 dewatering
2 post-dewatering flood
3 breaching/flood
4 removal/flood

Table 3: Load ratios (downstream/upstream) for TSS, DOC, TDN, DIN and DON

Event	TSS	DOC	TDN	DIN	DON
Dewatering	2.43	1.39	1.40	1.55	1.06
Dewater-Flood	0.76	1.17	1.22	1.44	1.08
Breaching	1.37	1.17	1.36	1.04	1.58
Removal	1.76	1.20	1.35	1.13	1.44

Table 4: Regional controls on downstream loading following dam removal

System	Region	Watershed Area (km^2)	Land Use[a]	Sediment Size[b]	Degree of Impact (TSS/N)
Baraboo River	Southwest WI, unglaciated, high-relief	575	Agricultural	Fine sand and silt	Order of magnitude greater/NR
Koshkonong River	South central WI, glaciated, low-relief	360	Agricultural	Fine sand and silt	Order of magnitude greater/NR
Manatawny Creek	Piedmont in PA	238	54/41/0/3	45mm; sand and gravel	No impacts detected
Murphy Creek	Central CA	12	80% cattle grazing, 20% viticulture	35/45/20	Order of magnitude greater for TSS/N
Little River	Upper coastal plain NC	600	44/39/12/5	Matrix supported sand and gravel	Less than 2 times greater for TSS/N

[a] values reported as %forest/agriculture/wetland/developed
[b] values reported as D_{50} or %sand/silt/clay
NR – Not Reported

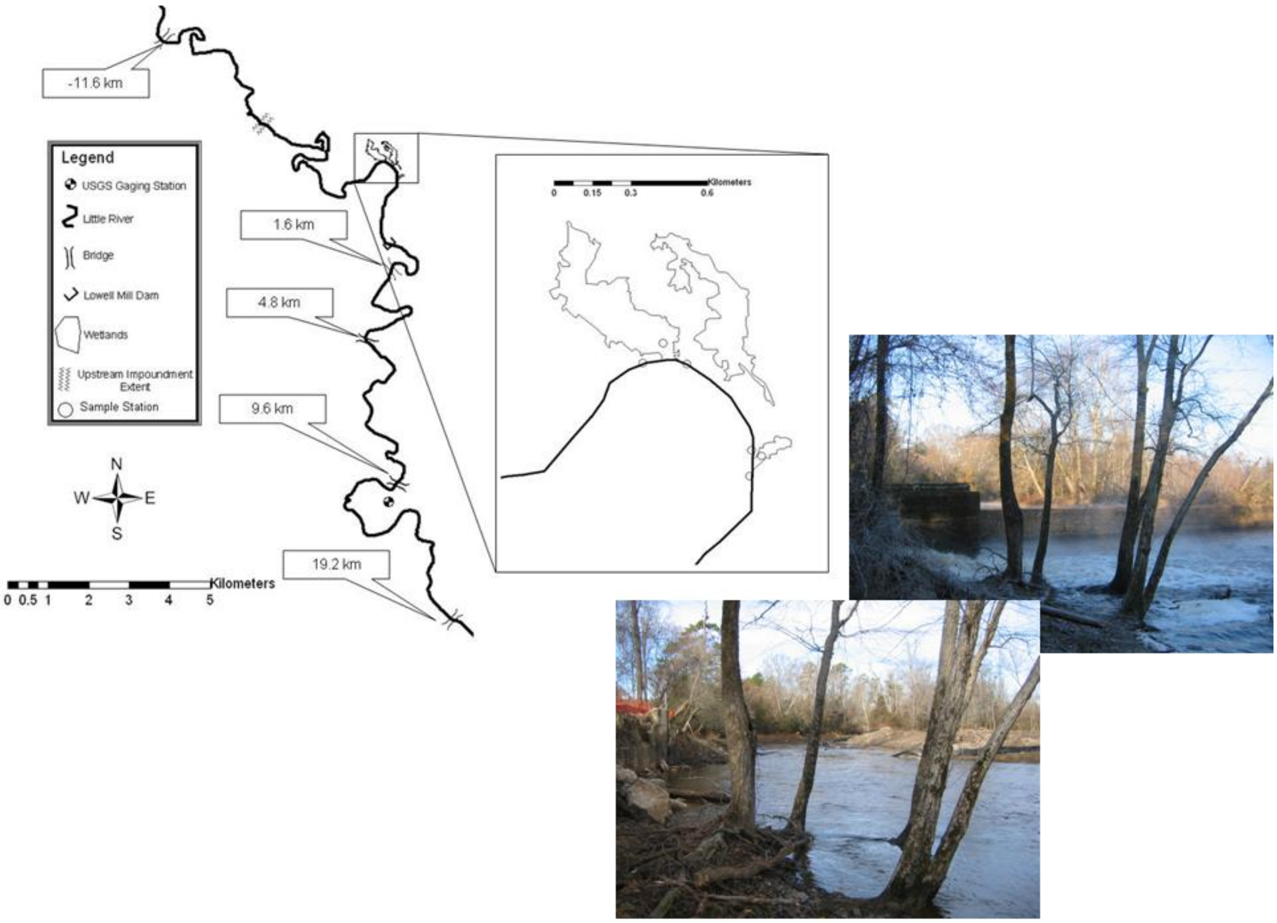


Figure 2.1: Study Reach: Lowell Mill Dam Impoundment located on the Little River in Johnston County, North Carolina. Arrows indicate the approximate locations of sampling stations.

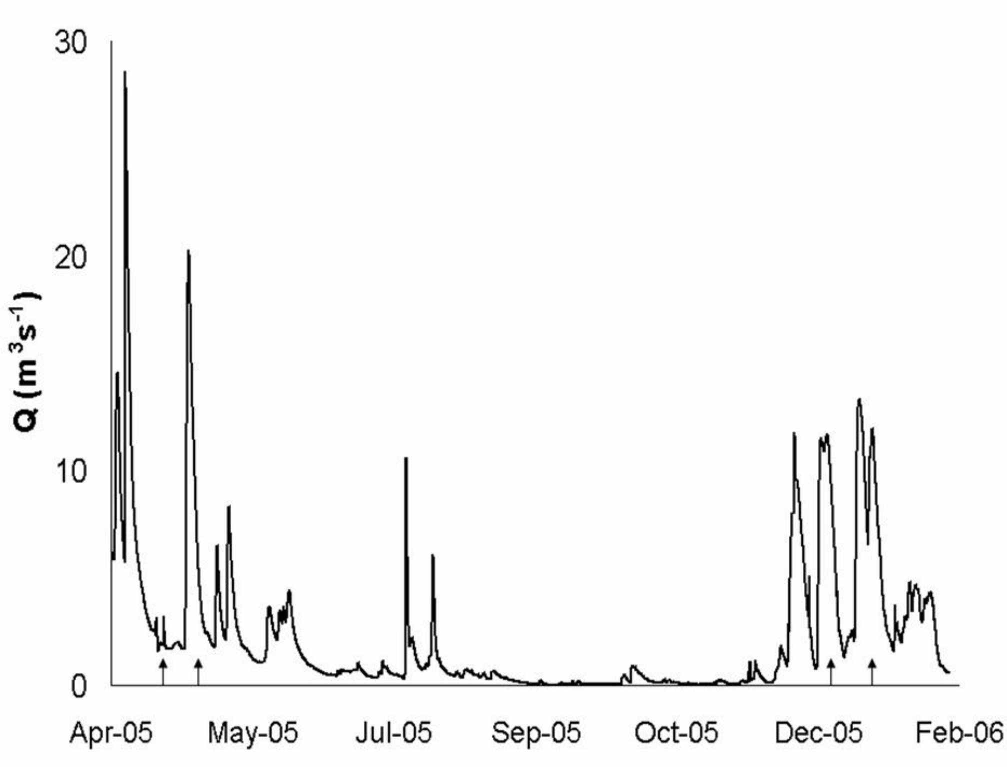


Figure 2.2: Little River hydrograph (April 2005 to January 2006). Arrows along x-axis indicate dam removal events.

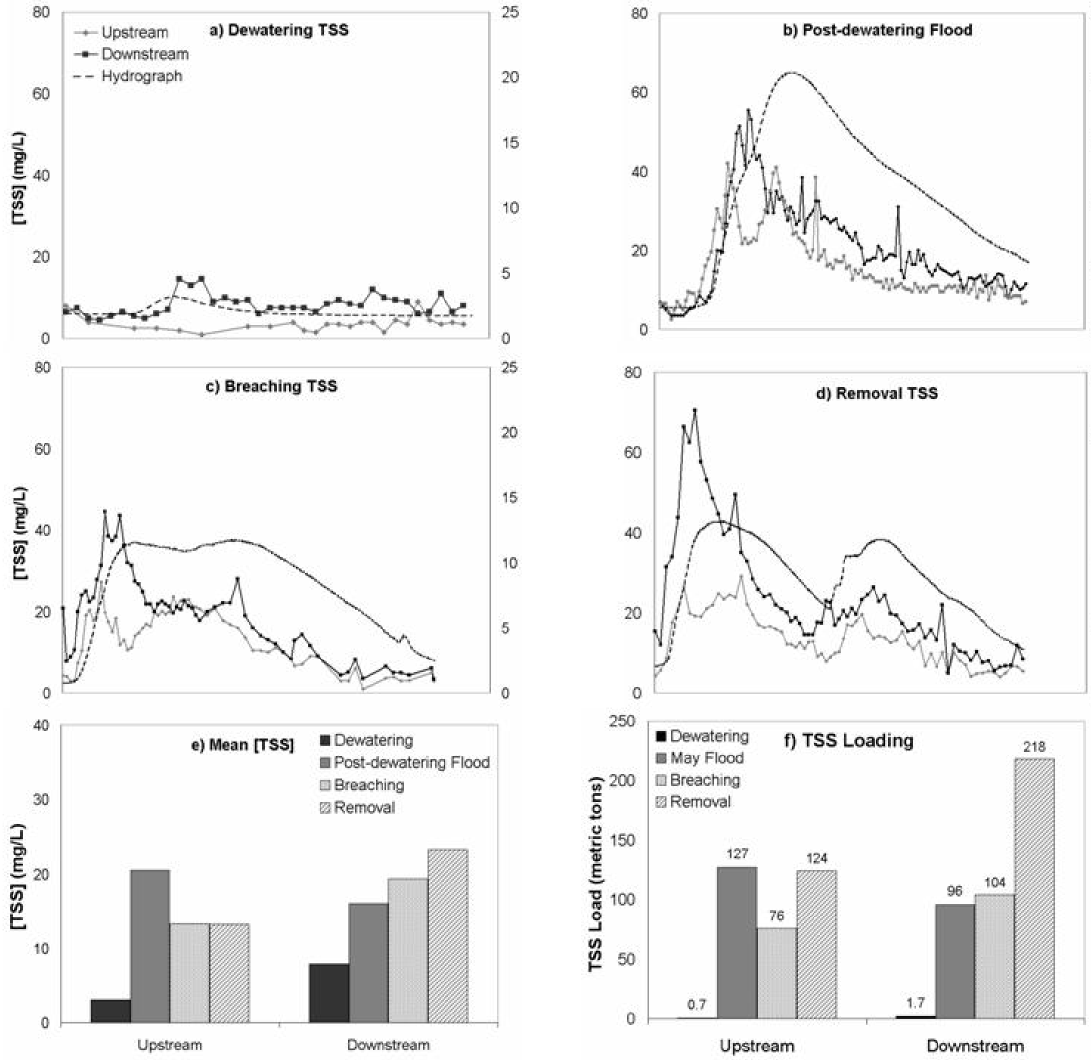


Figure 2.3: TSS Concentrations and loads during the removal of Lowell Mill Dam. Figures a-d: primary y-axes represent 0-80 mg/L TSS, secondary y-axes represent Q values 0-25 (cms), and x-axes represent time.

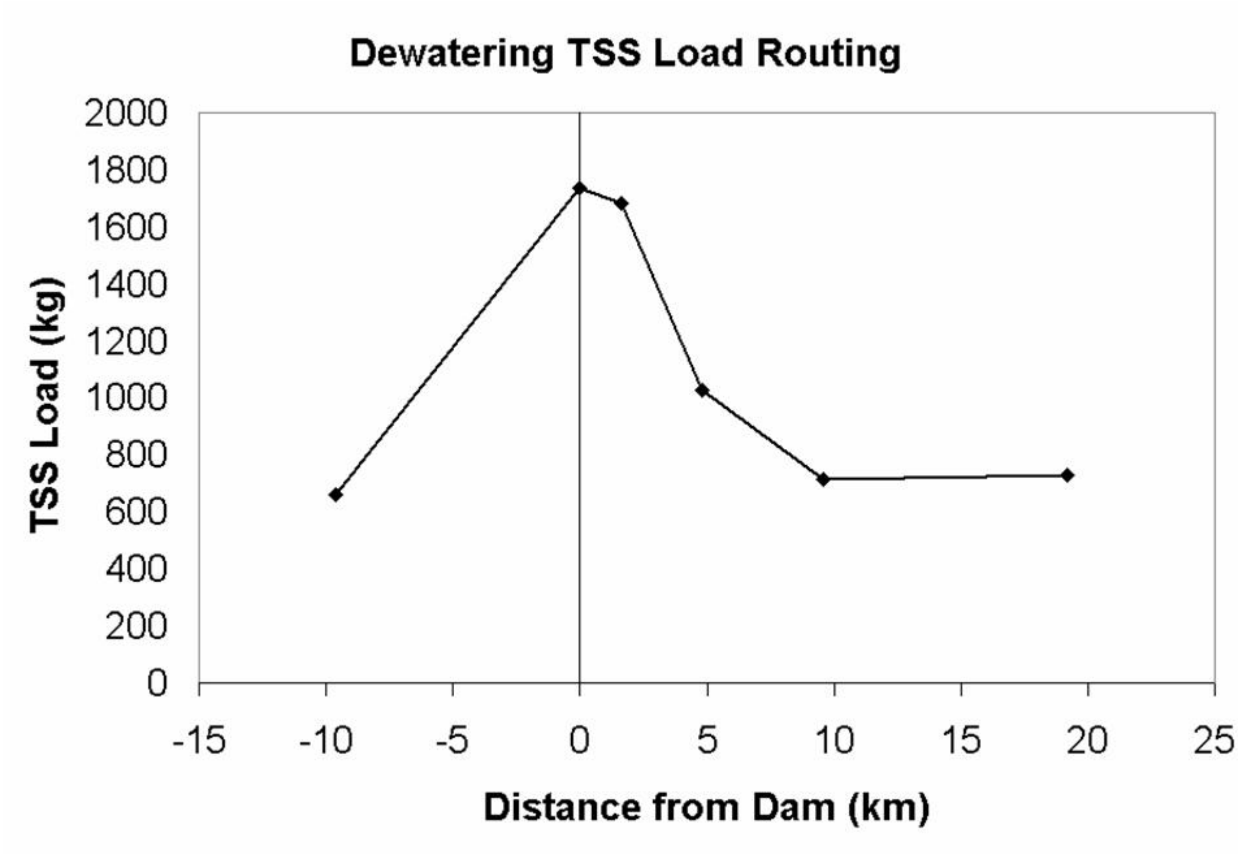


Figure 2.4: Downstream routing of TSS during the dewatering of Lowell Mill Impoundment. Vertical line indicates the location of the dam; negative values indicate distance upstream of the dam. Excess TSS released from the impoundment (relative to upstream inputs) was attenuated within 10 km of the dam.

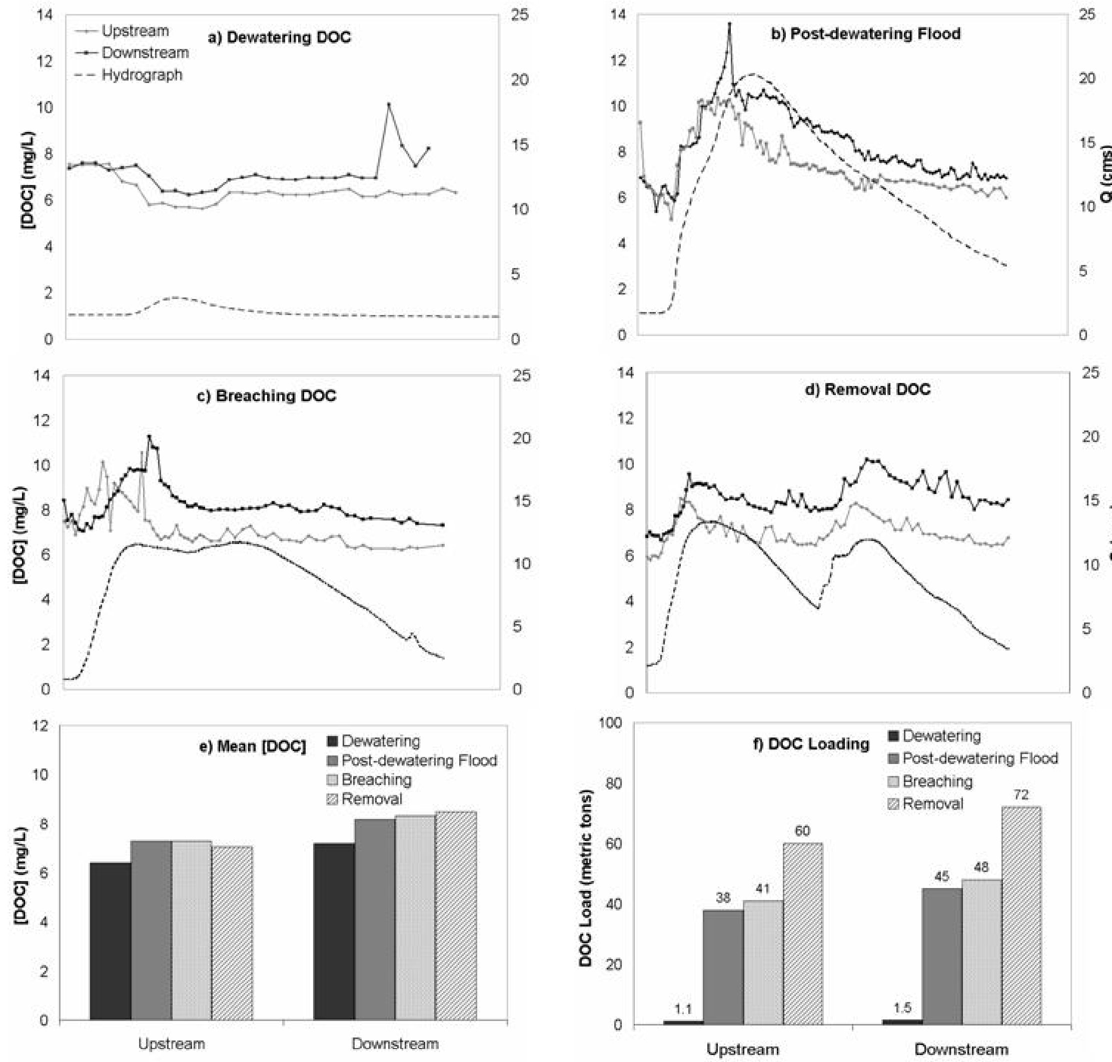


Figure 2.5: DOC concentrations and loads during the removal of Lowell Mill Dam. Figures a-d: primary y-axes represent 0-14 mg/L DOC, secondary y-axes represent Q values 0-25 (cms), x-axes represent time.

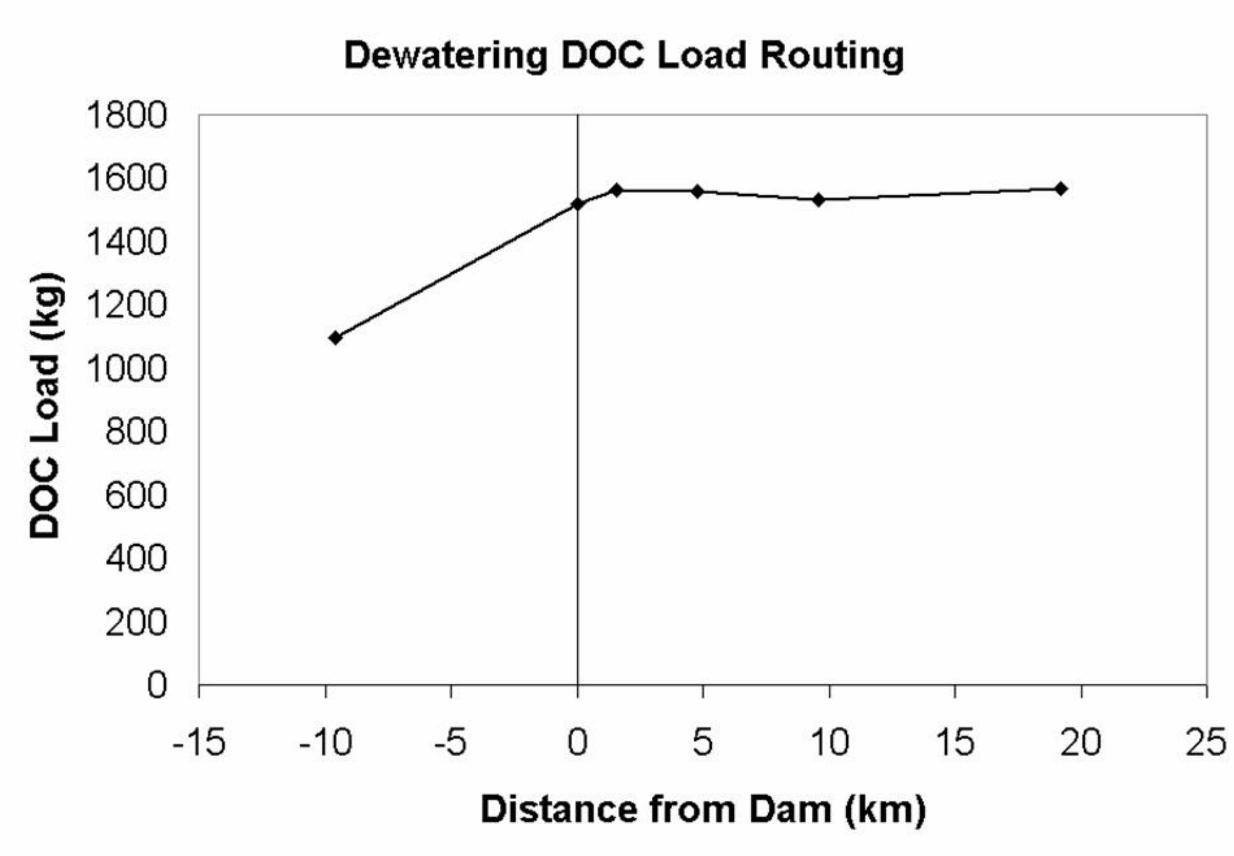


Figure 2.6: Downstream routing of DOC during the dewatering of Lowell Mill Impoundment. Vertical line indicates the location of the dam; negative values indicate distance upstream of the dam. The DOC load did not decrease with distance form the dam, instead DOC loads increased slightly.

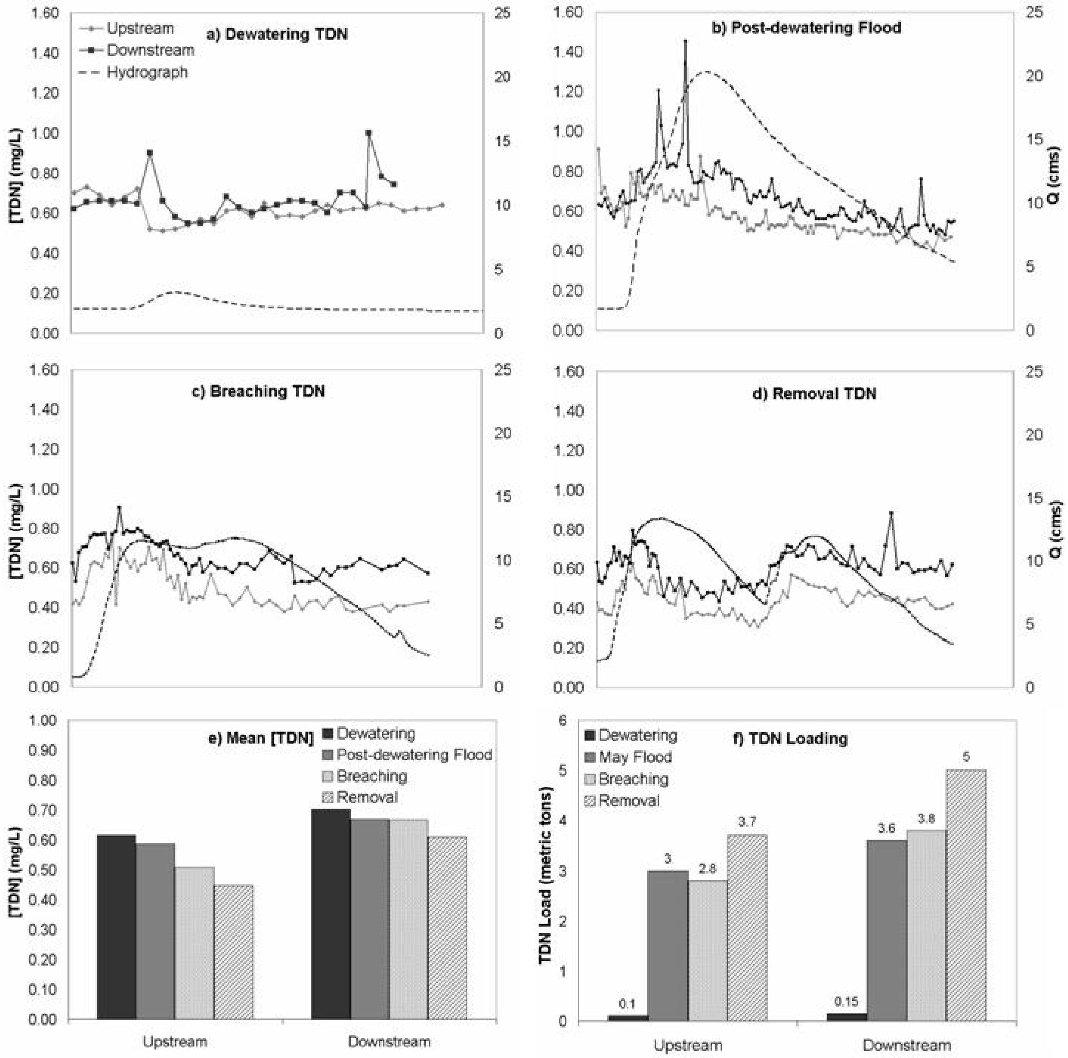


Figure 2.7: TDN concentrations and loads during the removal of Lowell Mill Dam. Figures a-d: primary y-axes represent 0-1.60 mg/L TDN, secondary y-axes represent Q values 0-25 (cms), x-axes represent time.

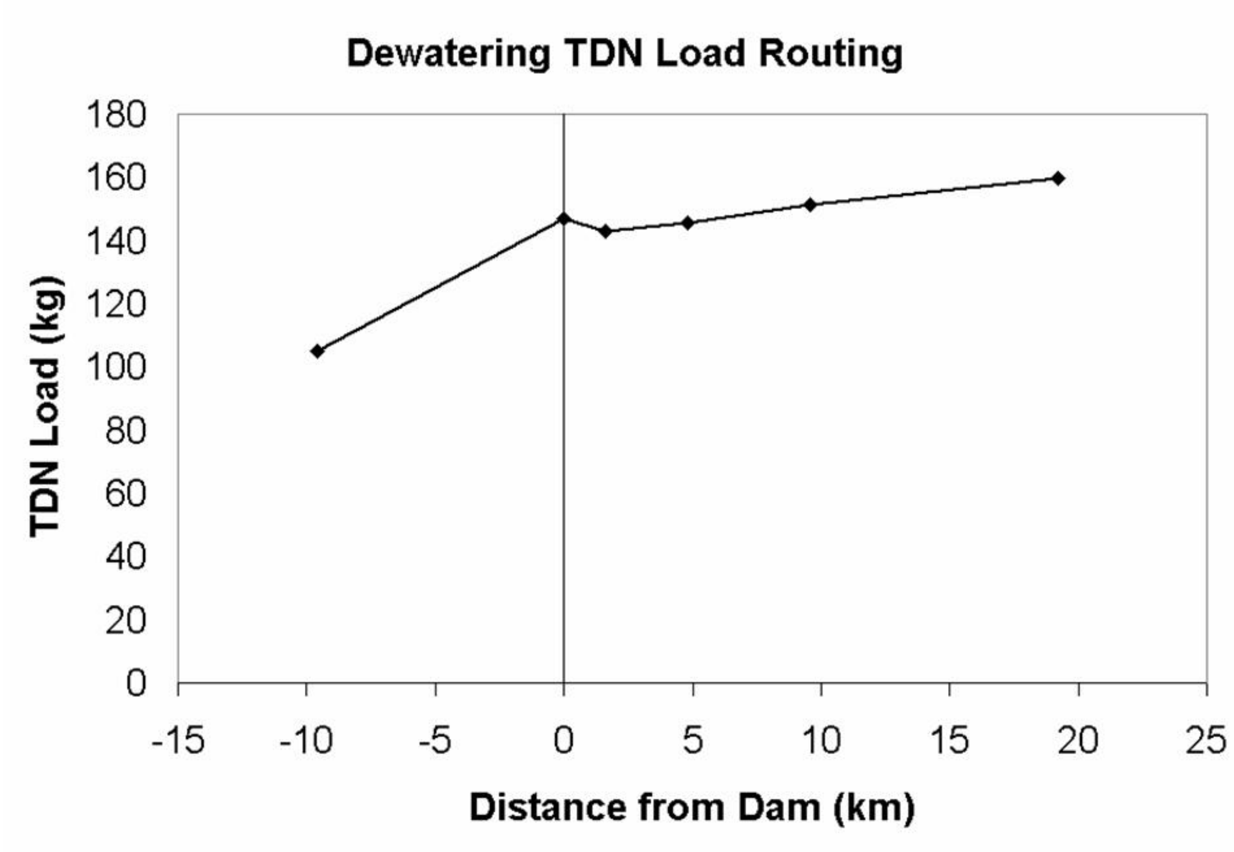


Figure 2.8: Downstream routing of TDN during the dewatering of Lowell Mill Impoundment. Vertical line indicates the location of the dam; negative distance values indicate distance upstream of the dam. The TDN load did not decrease with distance downstream, instead TDN increased slightly.

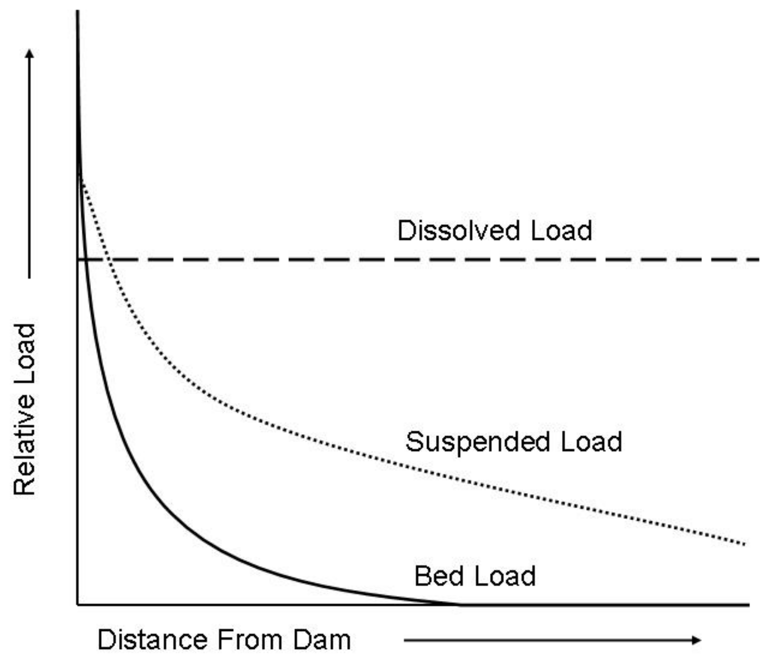


Figure 2.9: Transport of dissolved and particulate loads along the advective-dispersive continuum during floods and dam removals. Advective transport will have linear trends, while dispersive transport will have exponential trends.

CHAPTER III: SUSPENDED SEDIMENTS IN RIVER ECOSYSTEMS: PHOTOCHEMICAL SOURCES OF DISSOLVED ORGANIC CARBON AND ADSORPTIVE REMOVAL OF DISSOLVED IRON

> Under the action of the force of gravity the land surface is sculptured by water, wind, and ice. This sculpturing produces the landforms with which geomorphology is concerned. Some if these forms owe their origins purely to denudational processes; other forms may be depositional; still others owe their existence to combinations of both processes.
>
> - Leopold, Wolman and Miller (1964)

ABSTRACT

We generated suspended sediment solutions using river sediments and river water at concentrations similar to those observed during 1-1.5 year floods and a dam removal (~325 mg L^{-1}) on the Deep River, North Carolina. Suspended sediment solutions were exposed to simulated solar radiation, equivalent to one clear, summer day at the study site (36^{o} N). Concentrations of dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), total dissolved nitrogen (TDN), dissolved inorganic nitrogen (DIN), dissolved organic nitrogen (DON), soluble reactive phosphorus (SRP), and total dissolved iron (Fe_d) were measured before and after exposure. We found that sediment suspensions in the presence of simulated solar radiation were significant sources of C (1.2 $\pm$ 0.03 mmol C L^{-1} d^{-1}) and DON (1.2 $\pm$ 0.7 μmol N L^{-1} d^{-1}), but not DIN or SRP. Extrapolations through the Deep River water column suggest that suspended sediments, in the presence of light, represent fluxes of 3.9 mmol C m^{-2} d^{-1}, and 40 μmol N m^{-2} d^{-1}. Additionally, the sediment suspensions lowered river water Fe_d concentrations immediately (~ 24%) and

progressively (~40-90%) in both light and dark treatments. Thus, suspended sediments in river ecosystems are a C source and an Fe_d sink.

INTRODUCTION

On broad spatial and temporal scales, rivers owe their origins to erosional processes which create channels that transport hill slope materials to the world's oceans. However, on finer scales, rivers are actually mosaics of aggrading and degrading reaches routing materials through a series of erosional and depositional zones. Biogeochemical processes within river ecosystems are governed by these smaller-scale hydrogeomorphic conditions, making rivers important sites for global biogeochemical transport and transformation. Much scientific effort has been invested to explore links among hydrology, geomorphology and biogeochemistry in river ecosystems, including the River Continuum Concept (Vannote et al., 1980) as well as organic matter and nutrient spiraling (Webster and Patten, 1979; Minshall et al., 1992, 1993; Newbold, 1982, 1992). However, the biogeochemical role of suspended sediments, which are also controlled by hydrogeomorphic factors in watersheds, is less well understood.

Rivers are dynamic systems exhibiting considerable flow variability, resulting in periods of low and high transport capacity for various materials conveyed through channel networks. Increased concentrations of dissolved constituents during the rising limbs of flood hydrographs are well documented throughout the literature. Flushing from watershed soils has been offered as the most reasonable explanation for this trend. For example, dissolved organic carbon (DOC) flushing from hill slope soils was elicited to explain DOC hysteresis during snowmelt driven floods (Hornberger et al., 1994; Boyer et al., 1997, 2000). Watershed subsurface flow paths and near stream sources in particular are repeatedly cited as variable source areas, providing DOC enrichment to channels during floods (Meyer and Tate, 1983; Tate and Meyer, 1983; McDowell and Likens, 1988; Buffam et al., 2001). Most recently, DOC quality has been used as a hydrological tracer to infer hill slope source areas during various stages of flood hydrographs (Hood et al., 2006). Nitrogen flushing from watershed source areas has also been used to explain similar trends in dissolved organic nitrogen (DON) and dissolved inorganic nitrogen (DIN) flood dynamics (Creed et al. 1996; Creed and Band, 1998a and 1998b; Buffam, 2001).

Allochthonous sources of organic and inorganic materials are of obvious importance to river channel biogeochemistry and metabolism. While it is likely that watershed source areas contribute significant supplies of such materials, in-channel sources may also represent significant fluxes of organic and inorganic materials to the water column during episodic flood events. Once fine hill slope soils enter channel networks, the frequency and magnitude of movement is dictated by hydrogeomorphic controls which produce and maintain erosional and depositional features along the river continuum. These fine materials are subjected to multiple cycles of suspension and deposition as they are transported from hill slope to ocean. This cycle of suspension and deposition in rivers represents an important link among hydrology, geomorphology and biogeochemistry. We assert that this cycle, driven by flow conditions and channel geomorphology, results in the processing of watershed materials during downstream transport, and this processing represents an additional source of C, N and P to channel biogeochemistry.

Deposited fine sediments in aquatic ecosystems serve as benthic substrate with high denitrification potential (Pinay et al. 2000; Wetzel, 2001), and also as adsorptive sinks for dissolved organic matter (DOM) (McDowell and Wood, 1984; Nelson et al., 1993; Aufdenkampe et al., 2001), NH_4 (Triska et al., 1994; Schlesinger, 1997) and SRP (Meyer, 1979; Klotz, 1988; Mulholland, 1992). During inter-flood periods in which quiescent conditions dominate, benthic sediments may become anoxic, producing strong redox gradients which lead to the accumulation of DOM, inorganic N, P and various reduced terminal electron acceptors such as Fe^{2+} in interstitial waters (Wetzel, 2001). However, when river discharge increases these sediments are resuspended and may become an internal load (i.e., not from external hill slope variable source areas) of dissolved inorganic and organic forms of C, N and P to the water column via two pathways: interstitial water release, and desorption from sediment mineral surfaces. The concept of internal loading from anoxic hypolimnia and pore waters in lake ecosystems is well established and represents an appreciable source of P and N in mictic systems (Wetzel, 2001). In rivers, however, it is unlikely that pore water release could produce a measurable increase in water column N, P or C. This is because benthic sediment suspension, which leads to pore water release, typically only occurs during floods which dilute any effects of pore water. However, sediments may represent an important source of C, N, P and/or Fe to the water column.

There is some experimental evidence that suggests suspended sediments could be a considerable source of DOM to aquatic ecosystems. Reagent grade clay mineral surfaces sorbed appreciable quantities of DOM from leachate solutions, and simulated solar radiation facilitated the desorption of previously accumulated DOM (Tietjen et al., 2005). Additionally, recent experimental results, involving Mississippi River deltaic suspended solids in distilled water and artificial seawater solutions, demonstrated photodissolution of POC (Mayer et al., 2006). Finally, turbulence may control the degree of DOC mobilized from dried sediments during sediment resuspension (Koelmans and Prevo, 2003). Collectively, these studies suggest that suspended materials, in the presence of light and turbulence, can provide a measurable supplement of DOM to river ecosystems.

In rivers, the resuspension of fine sediments likely represents a source of DOM to the water column during transport events such as floods or dam removals. As has previously been shown, the desorption or dissolution of DOM from sediment surfaces is accelerated in the presence of light and turbulence. Further, the photochemical mineralization of desorbed DOM may release inorganic forms of C, N, P and Fe as has been demonstrated in rivers, lakes and estuaries (McKnight et al., 1988; White et al., 2003; Vahatalo and Wetzel, 2004; Vahatalo and Zepp, 2005). We propose that photoassisted desorption, coupled with photochemical mineralization, represent an internal load of organic and inorganic matter from sediment surfaces to the water column during floods. While it is likely that watershed contributions (i.e., external loads, watershed flushing) to flood biogeochemistry are more important from a total load or flux perspective, the concept of internal loading from sediment suspensions offers important insight into channel biogeochemical processing of hill slope materials routed to coastal ecosystems.

Laboratory experiments replicating the resuspension of river sediments were conducted in the presence of simulated solar radiation to determine if photoassisted sediment desorption of DOM could contribute measurable fractions of DOC to the water column during flood events. Additionally, organic and inorganic forms of N and P as well as total dissolved iron (Fe_d) were measured to determine if the photochemical mineralization of desorbed DOM would further enrich the water column. Thus, we tested whether the resuspension of fine sediments within river ecosystems represents an internal source of DOC, N, P and Fe.

METHODS

Overview of approach

We scaled laboratory experiments to replicate total suspended solids (TSS) concentrations observed during floods with recurrence intervals of 1.5 years ($Q_{1.5}$; Simon et al., 2004) on the Deep River, NC. This particular system was impounded by a run-of-river dam (Carbonton Dam), which was removed, following our study, in October of 2005. We chose this particular site because of the abundance of accumulated fine sediments, which would be mobilized during dam removal. DOC, TDN, DIN, SRP and Fe_d concentrations were measured before and after exposure to simulated solar radiation to determine whether photoassisted sediment surface desorption and photochemical mineralization of desorbed DOM contributes to water column nutrient enrichment during sediment suspension events (i.e., floods). Two series of experimental treatments with Deep River sediments were used: one treatment series used filtered river water, while the second used Milli-Q deionized water. All other experimental conditions were held constant for both treatment series. To accommodate for potentially active microbial communities and the complexity of coupled photoassisted desorption and photochemical mineralization, C fluxes were measured using a series of DOC and CO_2 measurements in closed systems. Measured CO_2 concentrations were used to calculate DIC within the water column following exposure to simulated solar radiation. Thus, DOC, DIC, and CO_2 measurements were used to calculate the total photochemically mediated C flux from suspended sediment surfaces.

Hydrogeomorphic scaling

Sediment and water were collected from the Carbonton Impoundment on the Deep River in Chatham, Moore and Lee counties, NC. Sediment concentrations used in our experiments were scaled to common transport events using USGS data from two gages which envelope the reach where water and sediments were collected. The upstream gage (Ramseur, NC; USGS # 02100500) is located approximately 35km from the impoundment. The downstream gage (Moncure, NC; USGS # 02102000) is located approximately 35 km downstream of the dam. Gage data were analyzed using standard recurrence interval (RI) analyses (Knighton, 1998), and all available TSS measurements were plotted against Q (mean daily flow) for the TSS collection dates (Figure 3.1). The upstream gage has an 82 year record with 46 TSS measurements, and the downstream gage has a 74 year record with 126 TSS measurements. Plots of TSS vs. Q were generated to determine the appropriate TSS concentrations at 1.5 year recurrence intervals. We determined that the system exhibited TSS concentrations of 200-400 mg L^{-1} during $Q_{1.5}$ events.

Solar radiation exposure was delivered to each treatment using an Atlas Suntest XLS+ solar simulator equipped with an arc xenon lamp. The lamp was calibrated to deliver radiation equivalent to the amount received by the study reach (Latitude 35°31'N, Longitude 79°21'W) during one clear summer day. The solar simulator supplies 14 KJ m^{-2} (equivalent to 650 W m^{-2}) of radiation over a course of 6 hours. A forced air cooling system kept water solutions at 25°C during the exposure process. Quartz tubes were used for all treatments as quartz transmits full spectrum sunlight.

Experimental sediment and water collections

Sediment cores and river water were collected on four separate occasions during September and October 2005 from the impounded reach of the Deep River. Sample collections occurred before the reach was impacted by dam removal (20 October 2005). This reach was selected because significant sediment accumulations are often associated with run-of-river impoundments. Such sediment accumulations are typically undisturbed for long periods of time, allowing for the development of strong redox gradients. Particle size distribution was determined to be 9% sand, 49% silt and 42% clay (using methods from Dane and Topp, 2002).

During all sampling trips, one core was collected from each of six sites within the impoundment. Cores were collected by inserting polycarbonate sleeves (without a coring device) measuring 30 cm in length and 5 cm in diameter into the soft, submerged sediment deposits along channel margins. Sleeves were pushed into the sediments until the top was flush with the sediment surface. The top opening of each sleeve was capped, and the sleeves were removed from the sediment accumulations. A second cap was used to cover the bottom sleeve opening to hold core contents in place before the core was removed from the water column. Tape was used to seal the caps to the core sleeves, and the cores were immediately transferred to a light-proof cooler packed with ice, and transported to the laboratory. Upon arrival, cores were stored in a light-proof container at $4^{o}C$ overnight.

On each sampling date, approximately 15 L of river water were collected in a location central to the six coring sites in acid washed HDPE containers. Water samples were packed on ice for transport to the laboratory. Collected river water was stored at $4^{o}C$, and filtered using 0.7 µm glass fiber filters (all filtration, unless otherwise mentioned, was accomplished with Whatman GF/F) within 12 hours of collection.

Experimental procedures

In the lab, one sediment core from each site was carefully pushed out of its sleeve, added to a plastic bag purged with N_2 gas after overlying water was poured off. The bag was then further purged with N_2 gas and sealed. Sediment homogenization was accomplished with vigorous hand kneading for several minutes. A grab sample of the homogenized sediments was then collected for addition to each sediment treatment. Remaining homogenized sediments were again purged with N_2 gas, sealed and stored for future experiments in a light-proof container and kept at 4^oC for no more than 48 hours. Experiments were run for three consecutive days using the same sediment stock.

To deliver the appropriate mass of sediment to experimental reliably and repeatedly we developed a sediment volume to dry mass curve. Two modified syringes (BD 20 ml and 3 ml) were used as small coring devices which allowed for reliable volumetric delivery to pre-dried and pre-weighed crucibles. Crucibles were dried at 100^oC for 24 hours and reweighed. It was determined that each 1.0 ml of wet, homogenized sediment was approximately equivalent to 600 mg of dried mass (R^2 = 0.99; n = 6). Thus, selection of 0.5 ml of wet sediment was determined as the appropriate volume to add 300 mg of TSS to each treatment (mean = 325 $\pm$ 0.06 mg of TSS, n = 83).

Homogenized sediment subsamples of 0.5ml were added to acid washed quartz tubes using a modified 3ml syringe followed by the addition of 950 ml of either river water (RW) or deionized water (DI). The resulting solution completely filled the volume of the quartz tubes so that no headspace remained.

Biogeochemical sampling procedures

Solutions within the quartz tubes were gently mixed at the beginning of the experiment to suspend all sediment and incubated under the appropriate light conditions for 6 hours. Full spectrum light treatments were placed directly in the solar simulator (referred to hereafter as Light). Because the solar simulator could only accommodate 4 quartz tubes at one time, the remaining quartz tubes were wrapped in aluminum foil and incubated on the lab bench directly adjacent to the solar simulator (referred to hereafter as Dark). A total of 8 quartz tubes were prepared for each experiment: 4 were subjected to the appropriate treatments (DI Light, DI Dark, RW Light, and RW Dark), 2 were duplicates which were rotated among treatments during each experiment, and 2 were sampled to represent initial concentrations (Initial). The initial quartz tubes (DI and RW) were sampled for DIN, TDN, SRP and DOC to represent the condition of sediment resuspension before exposure. This approach was chosen over destructively sampling each treatment to maintain controlled volume conditions for mass balance calculations and to avoid possible experimental error associated with altered diffusive gradients. Following 6 hour exposure, the solutions were sampled for DIN, TDN, SRP and DOC to represent the final condition (Final). These experiments produced a total of 9 replicates for each light and dark treatment. Data were analyzed by subtracting Final measurements from Initial measurements to determine change in concentrations following exposure.

One additional experiment was performed to determine the rate of DOC desorption from suspended sediments. Five quartz tubes were mixed with river water and sediments as detailed above. The rate experiment consisted of five measurements throughout the 6 hour exposure process: 0, 0.5, 1, 2 and 6 hours. Water samples were filtered and analyzed for DOC to establish a relationship between DOC concentration and time.

CO_2 accumulation in the headspace of the gas tight quartz tubes was measured at the end of 6-hour incubations for two experiments, and at 2-hour intervals for a third experiment (referred to as the CO_2 rate experiment). These experiments were prepared as detailed above with two exceptions: 1) sediment and water volumes were reduced, and 2) an additional control was used to account for photochemical CO_2 production from RW without sediments. The volume of sediment was reduced by one half (0.25 ml) as was the water volume (475 ml). This maintained the same concentration of sediment (300 mg L^{-1}), but allowed enough head space for gas sample collection. Gas samples were collected to determine differences in photochemical mineralization rates and mass loadings of sediment-based C among treatments. These experiments produced at least 3 replicates for all initial and final conditions.

Fe_d dynamics were also quantified during the exposure process. Fe_d was sampled from RW treatments only, as RW was the source of Fe_d in our experiments. Fe_d was determined before and after light exposure in two experiments using four treatments: light and dark with and without sediments. An additional rate experiment was also performed using the same design as the DOC rate experiments described above. Fe_d data are presented as percent difference because of considerable variability in Fe_d concentrations within stock RW between experiments. These experiments produced at least 3 replicates for all initial and final conditions.

Biogeochemical analyses

DOC and TDN samples were filtered into glass TOC vials and acidified to pH 2 with 2M ultrapure HCl. Samples were stored at 4^{o}C until analyzed using a Shimadzu TOC-V CPH total organic carbon analyzer coupled with a TNM-1 total nitrogen measuring unit. Samples were analyzed within 2 days of collection.

DIN and SRP samples were filtered into acid washed 125 ml amber HDPE bottles. Samples were frozen at -20^{o}C until analyzed for NH_4-N, NO_3-NO_2-N and SRP by the Analytical Services Laboratory at North Carolina State University in Raleigh, NC using an FIA autoanalyzer. DIN measurements from each treatment were subtracted from TDN measurements to determine DON content.

Fe_d was determined using the ferrozine colormetric method with hydroxylamine hydrochloride as a reducing agent (Stockley, 1970). Samples were analyzed within 3 hours of collection using a Beckman DU 650 UV/Vis spectrophotometer.

Headspace gas samples were taken from each treatment using 1 ml plastic syringes and measured within 1 hour of sampling for CO_2 concentration on a Shimadzu GC-14A gas chromatograph equipped with a Supelco 80/100 parapet Q column (6 ft x 1/8 in), a methanizer (set at 500^{o}C), and a flame ignition detector. Oven temperature was set at 35^{o}C and the runtime was 7 minutes. Helium was used as the carrier gas.

Corrections for CO_2 concentrations were needed to determine the photochemical mineralization of C from sediment sources only. In the case of DI treatments, the only source of CO_2 would be the sediments themselves. To account for respiration, dark control CO_2 (ppm) measurements (sediments and water) were subtracted from light treatments. For RW treatments, sediments are a CO_2 source, but RW DOC also contributed to CO_2 production via photomineralization or photobleaching. Thus, CO_2 production from sediments in RW was calculated by subtracting RW light/without sediments and RW dark treatments from RW light treatments.

CO_2 concentrations in the headspace of each treatment were used to calculate DIC concentrations. Measurements of CO_2 (g) were converted to mmol C using the Ideal Gas Law. DIC was then calculated as H_2CO_3 and HCO_3 using an assumed solubility constant (K_{CO2}) of 3.38×10^{-2} mol L^{-1} atm^{-1} (Pankow, 1991; Stumm and Morgan, 1996). For both RW and DI treatments, calculations were based on an assumed pressure of 1 atm, a measured temperature of 25°C, and a measured pH of 7.

Statistical analyses

Statistical analyses were conducted using analysis of variance adjusted for multiple observations within days. Major effects analyzed included: light treatment (light vs. dark) and water treatment (DI vs. RW). All analyses were performed using the SAS 9.1.3 statistical software package.

RESULTS

Carbon

DOC desorption from suspended sediment surfaces was significantly higher in both DI and RW light treatments compared to dark controls ($p < 0.001$; $n = 56$; Figure 3.2). There were no statistically significant differences between water treatments ($p = 0.6652$; $n = 56$). DOC increased in DI light treatments by 0.5 ± 0.32 mg L^{-1}, while DOC increased in RW light treatments by 0.45 ± 0.20 mg L^{-1} (Figure 3.2).

CO_2 concentrations in DI light treatments were significantly higher than those of RW light-sediment treatments ($p < 0.001$; $n = 9$; Figure 3.3). Both light-sediment treatments resulted in significantly higher CO_2 production than dark controls ($p < 0.001$; $n = 9$; Figure 3.3). Additionally, RW without sediments accounted for 0.65 ± 0.15 mg L^{-1} d^{-1} of CO2 production (Figure 3.3). Sediments surface derived C, however, increased CO_2 production by 70% in RW light treatments.

The greatest pool of C produced during light exposure in RW and DI treatments was IC (CO_2-C (*g*) + DIC; 18% CO_2-C and 82% DIC for both water treatments), as calculated from CO_2 production measurements (Figure 3.4). Based on mean DIC concentrations in sediment treatments, these pools accounted for 1.40 ± 0.18 mg L^{-1} for DI treatments, and 0.98 ± 0.46 mg L^{-1} for RW treatments. Additionally, based on DOC and CO_2 rate experiments using RW/sediment mixtures, an equilibrium condition was approached for DOC desorption and CO_2 production during the 6 hour exposure (Figures 3.5 and 3.6).

To quantify the total amount of C (TC) derived from sediment surfaces, IC pools were added to DOC pools (Figure 3.4). Thus, DI and sediments in light desorbed 1.90 $\pm$ 0.50 mg C L^{-1} d^{-1}, while RW and sediments in light desorbed 1.43 $\pm$ 0.66 mg C L^{-1} d^{-1} (Figure 3.4).

Dissolved nitrogen, phosphorus and iron

Sediments in both water-light treatments were a small, yet statistically significant, source of TDN enrichment compared to dark controls ($p < 0.001$; $n = 52$; Figure 3.7). As was the case with DOC, water type did not generate significant differences in TDN desorption from sediment surfaces ($p = 0.15$; $n = 52$). Mean water column enrichment of TDN in DI light treatments was 0.04 $\pm$ 0.03 mg N L^{-1} d^{-1}, while mean enrichment in RW light treatments was 0.02 $\pm$ 0.04 mg N L^{-1} d^{-1}. In both water treatments, nearly all TDN enrichment was identified as DON (99%).

Sediments in both water treatments in the presence of simulated solar radiation resulted in significantly higher concentrations of SRP ($p < 0.001$; $n = 47$). Mean SRP enrichment was minimal, however, at 0.01 $\pm$ 0.01 mg L^{-1}. Thus, the increase in SRP concentrations was negligible.

Sediments in both light and dark treatments were effective at removing Fe_d from river water ($p < 0.005$, $n = 9$; $p < 0.0250$, $n = 9$, respectively) relative to dark controls without sediments (Figure 3.8). There were no significant differences between light and dark sediment treatments ($p = 0.21$; $n = 9$), nor were there significant differences between dark and light treatments without sediments ($p = 0.08$, $n = 6$). The Fe rate experiments showed that once sediments were suspended in RW, there was an immediate 24% removal on Fe_d in both light and dark treatments (Figure 3.9).

DISCUSSION

Experimental Discussion

Our experimental results provide insight into the biogeochemical roles of suspended sediments during floods and dam removals. TSS concentrations representative of $Q_{1.5}$ events, exposed to simulated solar radiation produced measurable fluxes of organic matter, as nitrogen and carbon (Figures 3.2 and 3.7), while effectively removing Fe_d from the water column (Figure 3.8).

Total DON and TC fluxes in RW treatments equaled 0.02 ± 0.04 mg N L^{-1} d^{-1} and 1.43 ± 0.66 mg L^{-1} d^{-1}, respectively (Figures 3.7 and 3.4). In the following subsection, extrapolations were calculated based on the assumption that all C pools were first desorbed from sediment surfaces.

Removal of Fe_d in sediment treatments (Figure 3.8) can be explained by sediment surface adsorption, as fine sediments have an affinity for Fe^{3+} oxides. Previous research has suggested this relationship in an Eastern Shore aquifer in Virginia, where extractable Fe from oxic aquifer sediment surfaces was an order of magnitude greater than anoxic sediments from the same system (Knapp et al., 2002). Additionally, Fe^{3+} has been demonstrated to alter surface charges on clay particles inducing coagulation and sedimentation of Fe-clay complexes (Pierre, 1997; Ma and Pierre, 1999).

Extrapolation and comparison

While the processes described here are not as significant to flood biogeochemistry as has been previously demonstrated for watershed source areas (i.e., external loading), they do represent pathways of internal OM loading. For similar results to be expected in natural systems there are certain conditions which must be met, particularly a combination of adequate solar radiation and elevated concentrations of fine suspended sediments. Such conditions are often observed during floods as flood waves are routed through reaches hours to days following precipitation events. Floods may also originate in headwater reaches and travel through midreaches with reduced riparian and cloud cover. Dam removals and some ecosystem engineers such as carp, cows and hippos may create sediment suspensions as well.

There are challenges in applying our results directly to natural systems because they simulate processes occurring within the top 5 cm of the river water column only (depth of quartz tubes used for experimentation). Under the assumption that all IC pools presented in our results were first desorbed as DOC, and based on mean desorption values in RW treatments, suspended sediment loads within the upper 5 cm of the water column could contribute 119 mmol DOC m^{-3} d^{-1}, 1.2 mmol DON m^{-3} d^{-1} (SRP was not calculated because of its negligible increase during our experiments). It is important to note that in our experiments most C (65% or 77 mmol C m^{-3} d^{-1}) was mineralized during the course of exposure, and was therefore DIC (Figure 3.4). Such photochemical mineralization rates are comparable to previous studies in humic lake ecosystems, which have shown C photochemical mineralization rates in the surface layer of the water column to be 19 to 57 mmol C m^{-3} d^{-1} (Vahatalo et al., 2000; Anesio and Granéli, 2003). These systems are located in northern latitudes of Europe, and thus photochemical mineralization was limited by latitudinal controls.

Since our data represent photochemical reactions with suspended sediment in the top 5 cm of the water column only, other data are required to calculate this contribution throughout the water column including extinction coefficients and degree of riparian shading. Since these data are not currently available for the Deep River, similar data from the nearby Neuse River, NC we used (from Vahatalo et al., 2005). We assumed that UV-A (320-400 nm) was responsible for the reactions seen in our experiments based on previous research which showed that 68% of the photochemical mineralization of DOM in a humic lake was accomplished by UV-A (Vahatalo, 2000). For the purposes of this analysis, we used 360 nm to represent the wavelength of light responsible for the photoassisted desorption and photochemical mineralization of DOM from suspended sediments. The extinction coefficient for 360 nm on the Neuse River at Goldsboro, NC is 17.6 m^{-1}, and riparian shading, at this same site, reduces light availability to the channel by 58% (Vahatalo et al., 2005, personal communication). The average channel width of the Neuse River at this site is 36m, while the average channel width of the Deep River at Carbonton is 40 m. Thus, we assume that riparian cover effects on light availability are similar. Thus, incorporating these variables, suspended sediments account for 3.92 mmol C m^{-2} d^{-1}, and 40 µmol N m^{-2} d^{-1}.

Values allowing direct comparisons of our DON data were not readily available in the literature because most efforts involving photochemistry are focused on mineralization, specifically. We were unable to detect N mineralization in our experiments (i.e., DIN concentrations did not increase during exposure). However, our N enrichment (as DON) is comparable to the photochemical ammonification of DON in the Baltic Sea, which was found to generate 53 µmol of NH_4 m^{-2} d^{-1} (Vahatalo and Zepp, 2005).

Photoassisted desorption of DOC from suspended sediments, as reported here, is comparable to phytoplankton productivity at the Goldsboro, NC site on the Neuse River, 5.33 mmol C m^{-2} d^{-1} (Vahatalo et al., 2005). It is not comparable, however, to the mean daily load of DOC in the Deep River. At a mean DOC concentration of 7.15 mg L^{-1} (calculated during these experiments, n = 9), and a mean annual Q of 11.3 m^3 s^{-1} (USGS gage # 02100500), the Deep River transports 6.98 x 10^9 mg C d^{-1}. Once this is normalized to mean channel width (40 m), assuming flow across the channel is evenly distributed, this equals 1.75 x 10^8 mg C m^{-2} d^{-1} or 1.45 x 10^7 mmol C m^{-2} d^{-1}. Therefore, photoassisted sediment surface desorption of DOC during flood events is minimal compared to system inputs. However, this mechanism of DOC enrichment may supplement organic C losses caused by reduced phytoplankton productivity during floods.

Dams are effective sediment traps within channel networks, and therefore likely suppress sediment processing in rivers. However, their removals may present the proper conditions for photochemically mediated sediment desorption of DOM. Accumulated sediments behind Carbonton Dam were composed of predominately silt (49%) and clay (42%) fractions. Following dam removal, fine sediments were observed exiting the impoundment during flood events in concentrations greater than 350 mg L^{-1} (unpublished data). During reservoir dewatering (21 October 2005), impounded water was released from the structure's flood gates and fine sediments were exported from the reservoir in concentrations greater than 200 mg L^{-1} (unpublished data). Our results suggest that dam removal activities may produce environments capable of photoassisted desorption of DOM from suspended sediment surfaces. This mechanism may provide additional DOM enrichment as sediments released from impoundments are routed through downstream environments.

Suspended sediment controls on dissolved iron dynamics in river ecosystems

Iron oxides in the presence of light can mediate the oxidation of DOM in natural aquatic ecosystems through metal-ligand surface complexation reactions (McKnight et al. 1992). Fulvic, humic and hydrophilic acids are all expected to support such metal-ligand complexations. In acidic surface waters, the photochemical mineralization of DOM is coupled with the photoreduction of Fe^{3+} to the more soluble Fe^{2+}, which has been shown to increase Fe_d (McKnight et al. 1988). However, in less acidic systems, such as the Deep River (pH 7), the photochemical mineralization of DOM can lead to the formation of H_2O_2, which will increase the oxidation rates of Fe^{2+} (Cooper et al., 1988; Zuo and Hoigné, 1993). This suite of reactions is sometimes referred to as the photo-Fenton reactions, and supports the oxidation of DOM and the repeated reduction/oxidation of dissolved Fe. This cycle of Fe reduction and oxidation maintains the persistence of metal-ligand complexes which has been shown to positively influence rates of photochemical mineralization (Brinkmann et al., 2003). We have shown in our experiments that fine river sediments suspended during floods can rapidly and effectively scour the water column of Fe_d (Figures 3.8 and 3.9). Essentially, sediment suspensions reduce the importance of metal-ligand complexation reactions during photochemical mineralization.

There are interesting implications for Fe_d removal from the water column during flood events beyond that of photochemical mineralization. The ability of clay minerals to adsorb Fe_d from the water column during suspension events offers a source of Fe^{3+} to benthic microbial communities responsible for Fe reduction reactions. Our research, as well as that of others (Knapp et al., 2002), suggests that Fe reduction can be fueled by Fe^{3+} found on mineral surfaces in aquatic ecosystems. Ultimately, this process can be represented as a cycle driven by the suspension and deposition of fine sediments in river corridors. Following deposition, fresh sediment deposits are rich with Fe^{3+} oxides that will supply oxidative power necessary for the continued decomposition of OM trapped within the sediment matrix (Figure 3.10).

Photochemical processing of terrestrial C

Clay soils are responsible for halting the movement of DOC from soil pools to streams. In particular, polysaccharides, fulvic and humic materials are associated with clay soil surfaces (Tisdall and Oades, 1982; Oades, 1988; Tiessen and Stewart, 1988). Based on these relationships, it is apparent that clay minerals exert strong controls on stream biogeochemistry by limiting DOC transport from hill slopes to channels. For example, because of the absence of clay minerals in blackwater systems, DOC moves freely from watershed to channel (Beck et al., 1974). Thus, the movement of clay minerals from hill slopes to river channels represents an appreciable source of OM to river networks during transport events. As our data show, the interaction of suspended sediment and solar radiation is an additional flux of terrestrial C to aquatic ecosystems. Therefore, the photochemical mineralization of OM from mineral surfaces represents a sink of terrestrial C reducing the particulate C load delivered to coastal ecosystems.

Mineralization and desorption of terrestrial C sorbed to sediment mineral surfaces can be viewed as a gradual process driven by hydrogeomorphic features within fluvial systems. Photochemical mineralization is an important mechanism driving this flux of C during sediment suspension events (Figure 3.11). Hill slope soils are introduced to channels by denudational forces such as erosion produced by overland flow. In flows conditions which are capable of maintaining these soils/sediments in suspension, light can accelerate desorption and mineralization of some C from suspended mineral surfaces. Upon flood recession, suspended fine sediments are deposited within channel networks, where continued desorption and mineralization of sorbed terrestrial C can occur via microbial decomposition. This process represents the progressive, longitudinal processing of terrestrial C as fine sediments are slowly routed through channel networks.

CONCLUSIONS

Hydrogeomorphic controls on the suspension and deposition of fine sediments within river corridors are important in the regulation of OM processing in rivers via two mechanisms: 1) photoassisted desorption of DOM which transfers and oxidizes C from suspended hill slope soils, and 2) Fe_d adsorption, which delivers Fe^{3+} (necessary for Fe reducing bacteria) from the water column to the anoxic benthos following deposition. This second pathway supplies essential terminal electron acceptors for the continued decomposition of OM sorbed to sediment surfaces.

It is apparent that altered land use has increased both erosion rates and the frequency and magnitude of events that are capable of suspending fine sediments in rivers (Wolman, 1967). Thus, altered hydrology has increased the flux of C containing materials from watershed to channel. It is unclear whether this hydrological control would increase or decrease the efficacy of photochemical reactions on C processing during suspension events. However, it is reasonable that with increased frequency of suspension and increased fine sediment loads, appreciable quantities of riverine dissolved Fe may be transformed into particulate Fe. This relationship may influence global C cycling because fluxes of particulate Fe are not thought to reach pelagic marine phytoplankton communities, which are considered to be Fe limited (Jickells et al., 2005).

REFERENCES

Alexander RB, Smith RA and Schwarz GE. 2000. Effects of stream channel size on the delivery of nitrogen to the Gulf of Mexico. *Nature* 403: 758-761.

Anesio AM and Granéli W. 2003. Increased photoreactivity by acidification: Implications for the carbon cycle in humic lakes. *Limnology and Oceanography* 48: 735-744.

Aufdenkampe AK, Heges JI, Richey JE, Krusche AV and Llerena CA. 2001. Sorptive fractionation of dissolved organic nitrogen and aminao acids onto fine sediments with the Amazon Basin. *Limnology and Oceanography* 46: 1921-1935.

Beck KC, Reuter JH and Perdue EM. 1974. Organic and inorganic geochemistry of some coastal plain rivers in the southeastern United States. *Geochimica et Cosmochimica Acta* 38: 341-364.

Boyer EW, Hornberger GM, Bencala KE, McKnight DM. 1997. Response characteristic of DOC flushing in an alpine catchment. *Hydrological Processes* 11: 1635-1647.

Boyer EW, Hornberger GM, Bencala KE, McKnight DM. 2000. Effects of asynchronous snowmelt on flushing of dissolved organic carbon: a mixing model approach. *Hydrological Processes* 14: 3291-3308.

Brinkmann T, Horsch P, Sartorius D and Frimmel FH. 2003. Photoformation of low-molecular-weight organic acids from brown water organic matter. *Environmental Science and Technology* 37: 4190-4198.

Buffam I, Galloway JN, Blum LK and McGlathery KJ. 2001. A stromflow/baseflow comparison of dissolved organic matter concentrations and bioavailability in an Appalachian stream. *Biogeochemistry* 53: 269-306.

Cooper WJ, Zika RG, Petasne RG and Plane JMC. 1988. Photochemical formation of H_2O_2 in natural waters exposed to sunlight. *Environmental Science and Technology* 22: 1156-1160.

Creed IF, Band LE, Foster NW, Morrison IK, Nicholson JA, Semkin RS and Jefferies DS. 1996. Regulation of nitrate-N release from temperate forests: A test of the N-flushing hypothesis. *Water Resources Research* 32: 3337-3354.

Creed IF and Band LE. 1998a. Exploring functional similarity in the export of nitrate-N from forested catchments: A mechanistic modeling approach. *Water Resources Research* 34:3079-3093.

Creed IF and Band LE. 1998b. Export of nitrogen from catchments within a temperate forest: Evidence of a unifying mechanism regulated by variable source area dynamics. *Water Resources Research* 34: 3105-3120.

Dane JH and Topp GC. 2002. *Methods of Soil Analysis: Part 4 – Physical Methods*. Soil Science Society of America, Madison. 1692 pp.

Hood E, Gooseff MN and Johnson SL. 2006. Changes in stream water dissolved organic carbon during flushing in three small watersheds, Oregon. *Journal of Geophysical Research* VOL. 111, G01007, doi:10.1029/2005JG000082, 2006

Hornberger GM, Bencala KE and McKnight DM. 1994. Hydrological controls on dissolved organic carbon during snowmelt in the Snake River near Montezuma, Colorado. *Biogeochemistry* 25: 147-165.

Jickells TD, An ZS, Andersen KK, Baker AR, Bergametti G, Brooks N, Cao JJ, Boyd PW, Duce RA, Hunter KA, Kawahata H, Kubilay N, LaRoche J, Liss PS, Mahowald N, Prospero JM, Ridgwell AJ, Tegen I and Torres R. 2005. Global iron connections between desert dust, ocean biogeochemistry and climate. *Science* 308: 67-71.

Klotz RL. 1988. Sediment control on soluble reactive phosphorus in Hoxie Gorge Creek, New York. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 2026-2034.

Koelmans AA and Prevo L. 2003. Production of dissolved organic carbon in sediment suspensions. *Water Research* 37: 2217-2222.

Knapp EP, Herman JS, Mills AL and Hornberger GM. 2002. Changes in the sorption capacity of Coastal Plain sediments due to redox alteration of mineral surfaces. *Applied Geochemistry* 17: 387-398.

Knighton D. 1998. *Fluvial Forms and Processes: A New Perspective*. Arnold, London. 383 pp.

Leopold LB, Wolman MG and Miller JP. 1992. *Fluvial Processes in Geomorphology*. 2nd Edition. Dover Publications, Inc., New York. pp 522.

Ma KS and Pierre AC. 1999. Colloidal behavior of montmorillonite in the presence of Fe^{3+} ions. *Colloids and Surfaces: Physiochemical and Engineering Aspects* 155: 359-372.

Mayer LM, Schick LL, Skorko K. 2006. Photodissolution of particulate organic matter from sediments. *Limnology and Oceanography* 51: 1064-1071.

McDowell WH and Wood T. 1984. Podzolization: Soil processes control dissolved organic carbon concentrations in stream water. *Soil Science* 137: 23-32.

McDowell WH and Liken GE. 1988. Origin, composition and flux of dissolved organic carbon in the Hubbard Brook Valley. *Ecological Monographs* 58: 177-195.

McKnight DM, Kimball BA, Bencala KE. 1988. Iron photoreduction and oxidation in an acidic mountain stream. *Science* 240: 637-640.

McKnight DM, Bencala KE, Zellweger GW, Aiken GR, Feder GL and Thorn KA. 1992. Sorption of dissolved organic carbon by hydrous aluminum and iron-oxides occurring at the confluence of Deer Creek with the Snake River, Summit County, Colorado. *Environmental Science and Technology* 26: 1388-1396.

Meyer J. 1979. The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. *Limnology and Oceanography* 24: 365-375.

Meyer JL and Tate CM. 1983. The effects of watershed disturbance on dissolved organic carbon dynamics of a stream. *Ecology* 64: 33-44.

Minshall GW, Petersen RC, Cummins KW, Bott TL, Sedell JR, Cushing CE and Vannote RL. 1983. Interbiome comparison of stream ecosystem dynamics. *Ecological Monographs* 53: 1-25.

Minshall GW, Petersen RC, Bott TL, Cushing CE, Cummins KW, Vannote RL, Sedell JR. 1992. Stream ecosystem dynamics of the Salmon River, Idaho – An 8th order system. *Journal of North American Benthological Society* 11: 111-137.

Mulholland PJ. 1992. Regulation of nutrient concentrations in a temperate forest stream: Roles of upland, riparian and instream processes. *Limnology and Oceanography* 37: 1512-1526.

Newbold JD, Mulholland PJ, Elwood JW and O'Neill RV. 1982. Organic carbon spiraling in stream ecosystems. *Oikos* 38: 266-272.

Newbold JD. 1992. Cycles and spirals of nutrients. *In* P. Calow and GE Petts, eds. The rivers handbook. I. Hydrological and ecological principles. Blackwell Science Publishers, Oxford. pp. 379-408.

Oades JM. 1988. The retention of organic matter in soils. *Biogeochemistry* 5: 35-70.

Pankow JF. 1991. *Aquatic Chemistry Concepts*. Lewis Publishers, Chelsea, MI. 683 pp.

Pierre AC. 1997. Sedimentation behavior of kaolinite and montmorillonite mixed with iron additives, as a function of their zeta potential. *Journal of Materials Sciences* 32: 2937-2947.

Pinay G, Black VJ, Planty-Tabacchi AM, Gumiero B and Décamps H. 2000. Geomorphic control on denitrification in large river floodplain soils. *Biogeochemistry* 50: 163-182.

Schlesinger WH. 1997. *Biogeochemistry: An analysis of global change*. 2nd Edition. Academic Press, San Diego. 588 pp.

Simon A, Dickerson W and Heins A. 2004. Suspended-sediment transport rates at the 1.5-year recurrence interval for ecoregions of the United States: transport conditions at the bankfull and effective discharge? *Geomorphology* 58: 243-262.

Stockley LL. 1970. Ferrozine-A new spectrophotometric reagent for iron. *Analytical Chemistry* 42: 779-781.

Stumm W and Morgan JJ. 1996. *Aquatic Chemistry*. 3rd Edition. John Wiley and Sons, New York. 1022 pp.

Tate CM and Meyer JL. 1983. The influences of hydrologic conditions and successional state on dissolved organic carbon export from forested watersheds. *Ecology* 64: 25-32.

Tiessen H and Stewart JWB. 1988. Light and electron microscopy of stained microaggregates: The role of organic matter and microbes in soil aggregation. *Biogeochemistry* 5: 312-322.Wetzel RG. 2001. *Limnology: Lake and River Ecosystems*. 3rd Edition. Academic Press, San Diego. 1066 pp.

Tietjen T, Vahatalo AV and Wetzel RG. 2005. Effects of clay mineral turbidity on dissolved organic carbon and bacterial production. *Aquatic Sciences* 67: 51-60.

Tisdall JM and Oades JM. 1982. Organic matter and water-stable aggregates in soils. *Journal of Soil Science* 33: 141-163.

Triska FJ, Jackman AJ, Duff JH and Avanzino RJ. 1994. Ammonium sorption to channel and riparian sediments: A transient storage pool for dissolved inorganic nitrogen. *Biogeochemistry* 26: 67-83.

Vahatalo AV, Salkinoja-Salonen M, Taalas P and Salonen K. 2000. Spectrum of the quantum yield for photochemical mineralization of dissolved organic carbon in a humic lake. *Limnology and Oceanography* 45: 664-676.

Vahatalo AV and Wetzel RG. 2004. Photochemical and microbial decomposition of chromophoric dissolved organic matter during long (months-years) exposures. *Marine Chemistry* 89: 313-326.

Vahatalo AV and Zepp RG. 2005. Photochemical mineralization of dissolved organic nitrogen to ammonium in the Baltic Sea. *Environmental Science and Technology* 39: 6985-6992.

Vahatalo AV, Wetzel RG and Paerl HW. 2005. Light absorption by phytoplankton and chromophoric dissolved organic matter in the drainage basin and estuary of the Neuse River, North Carolina (U.S.A). *Freshwater Biology* 50: 477-493.

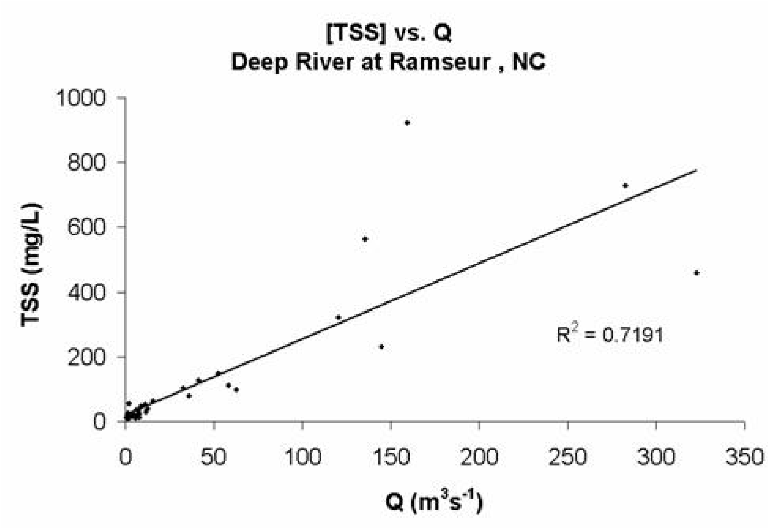
Vannote RL, Minshall GW, Cummins KW, Sedell JR and Cushing CE. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.

Webster JR and Patten BC. 1979. Effects of watershed perturbation on stream potassium and calcium dynamics. *Ecological Monographs* 49: 51-72.

White EM, Vaughn PP, Zepp RG. 2003. Role of photo-Fenton reaction in the production hydroxyl radicals and photobleaching of colored dissolved organic matter in a coastal river of southeastern United States. *Aquatic Sciences* 65: 402-414.

Wolman, M.G., 1967. A cycle of sedimentation and erosion in urban river channels. Geografiska Annaler, 49A: 385-395.

Zuo Y and Hoigné J. 1993. Evidence of photochemical formation of H_2O_2 and oxidation SO_2 in authentic fog water. *Science* 260: 71-73.



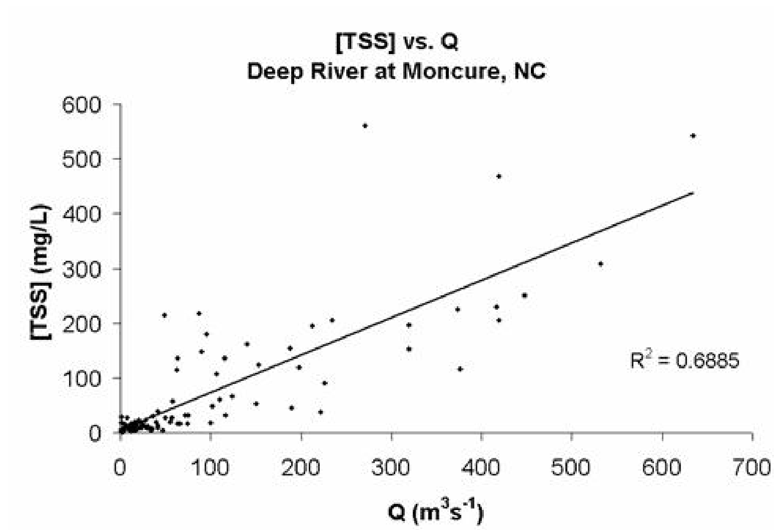


Figure 3.1: TSS vs. Q for the Deep River at Ramseur and Moncure, NC.

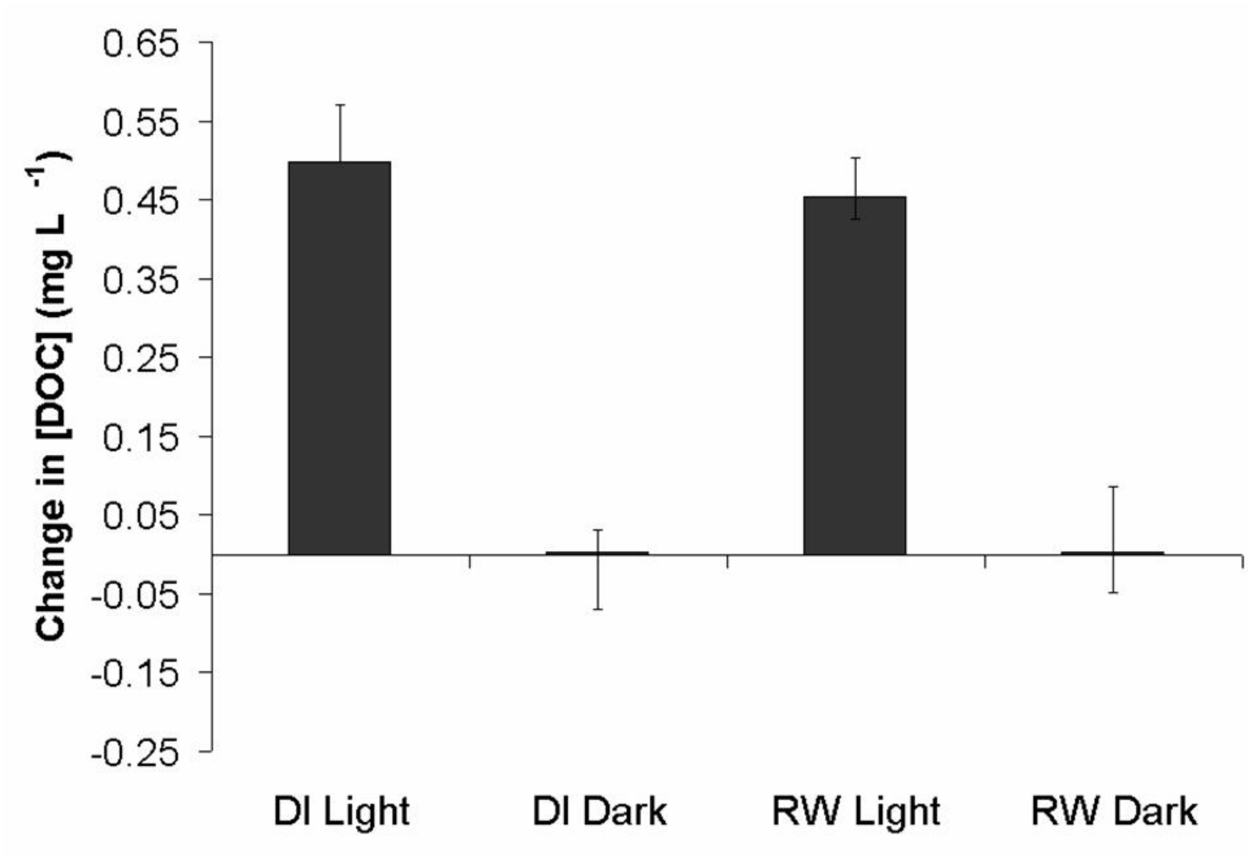


Figure 3.2: DOC desorption from suspended sediments in light and dark treatments. Error bars represent $\underline{+}1$ SE, n = 9 per treatment.

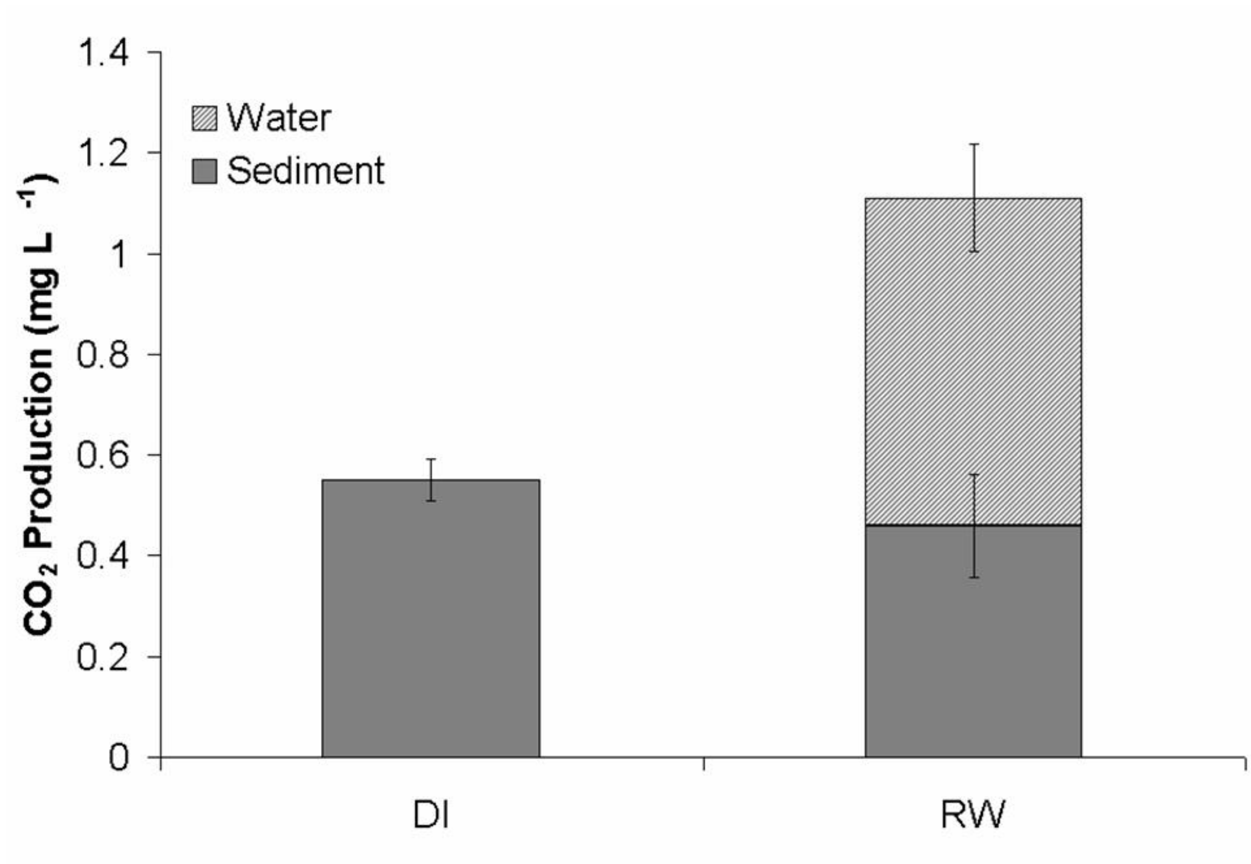


Figure 3.3: CO_2 production for DI and RW treatments. DI with sediments treatments were corrected by subtracting CO_2 production of dark controls. River water with sediment CO_2 production was corrected by subtracting both river water/ no sediments in light and river water/sediment dark controls. River water/no sediment light controls are also shown for comparison purposes. Error bars represent ± 1 SE, n = 3 per treatment.

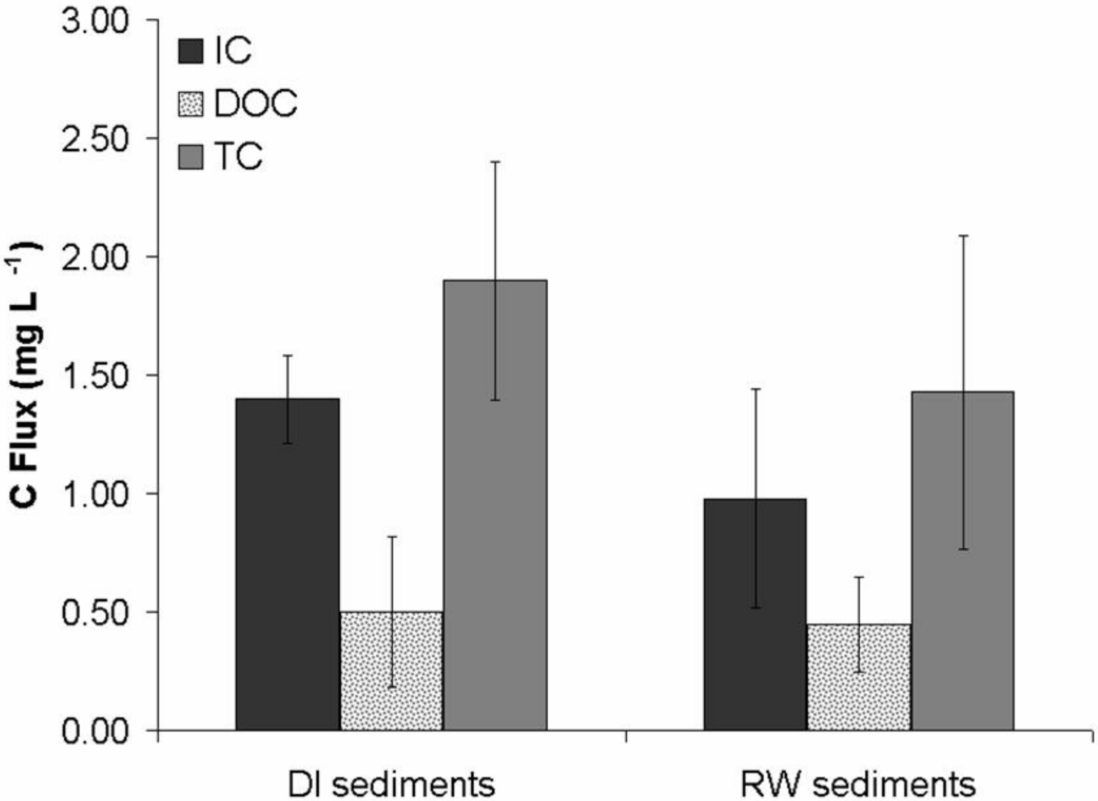


Figure 3.4: Total C (TC) desorption in DI and RW treatments in the presence of light. TC equals sum of IC (DIC + CO_2) and DOC. Error bars represent $\pm$1 SD.

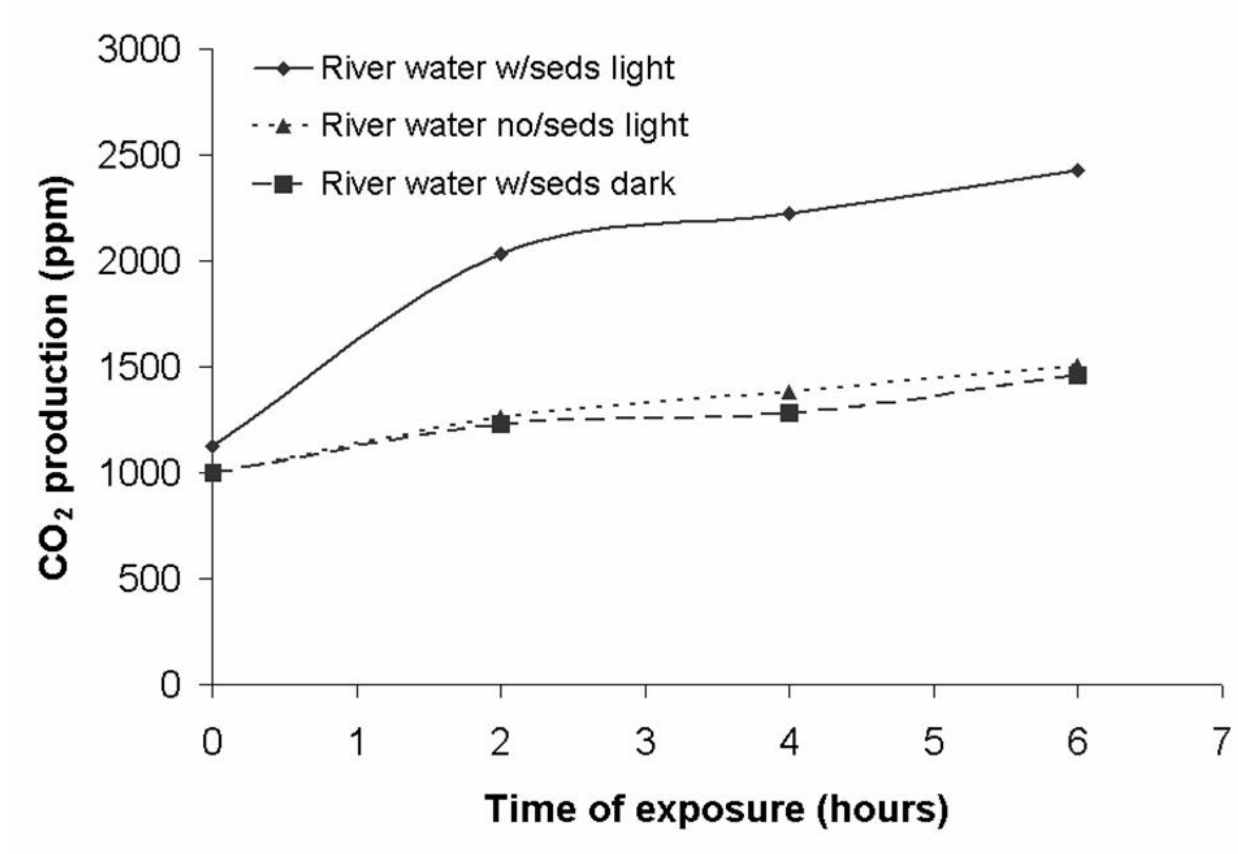


Figure 3.5: CO_2 production rate experiment.

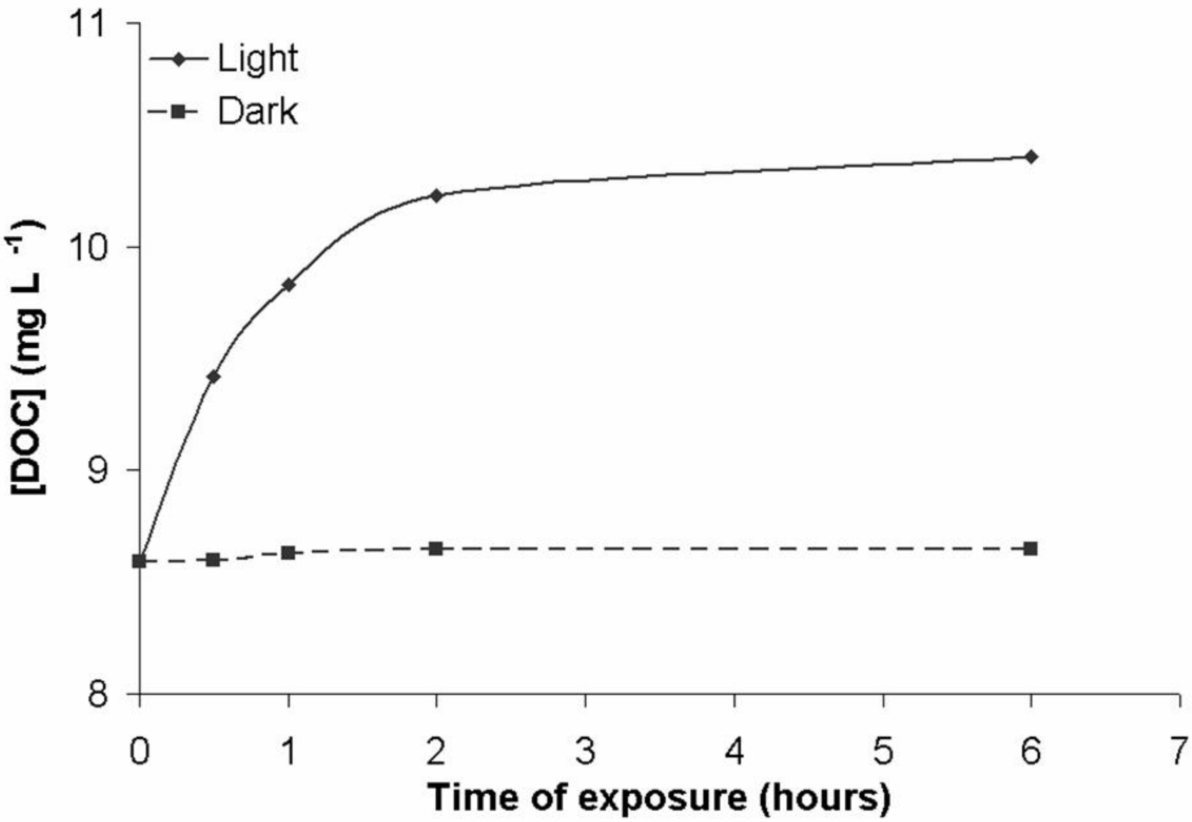


Figure 3.6: DOC desorption rate experiment. Treatments were composed of river water sediment mixtures - light vs. dark. Note that the total desorption of DOC during this experiment (~2 mg L^{-1}) was higher than mean desorption values reported in the text (0.45 mg L^{-1} d^{-1}).

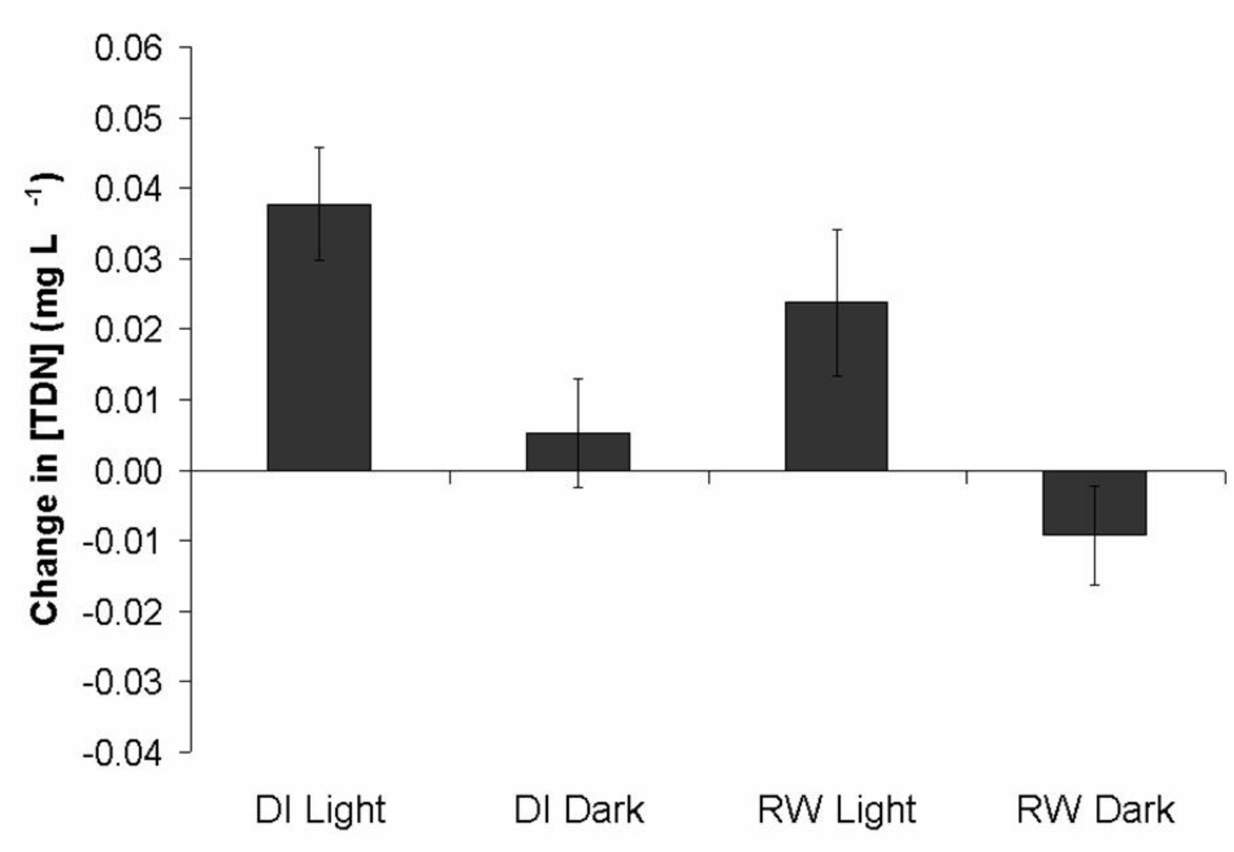


Figure 3.7: TDN desorption from suspended sediments in light and dark treatments. Error bars represent ±1 SE, n = 9 per treatment.

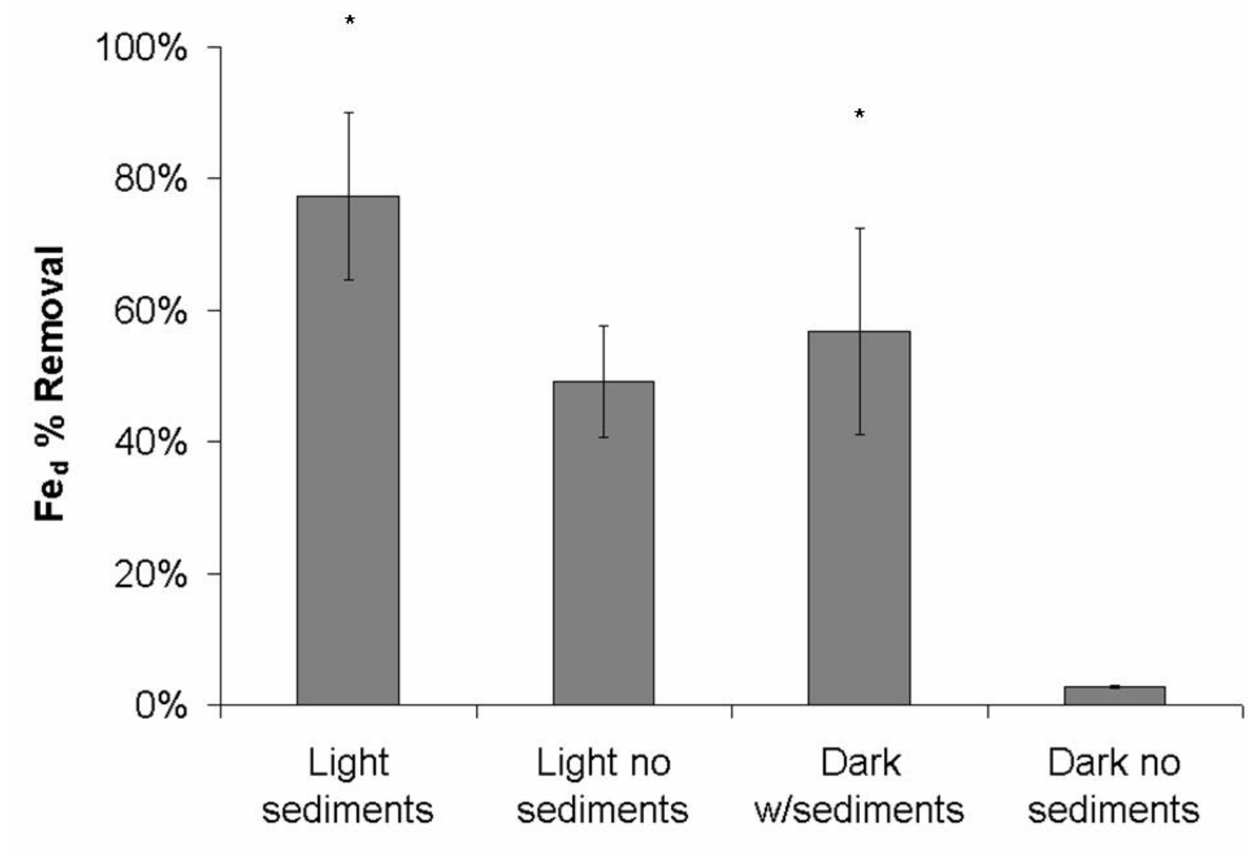


Figure 3.8: Fe_d removal efficiency. '*' represent treatments with significantly different means compared to dark/no sediment controls. Error bars represent $\pm$1 SE, n = 3 per treatment.

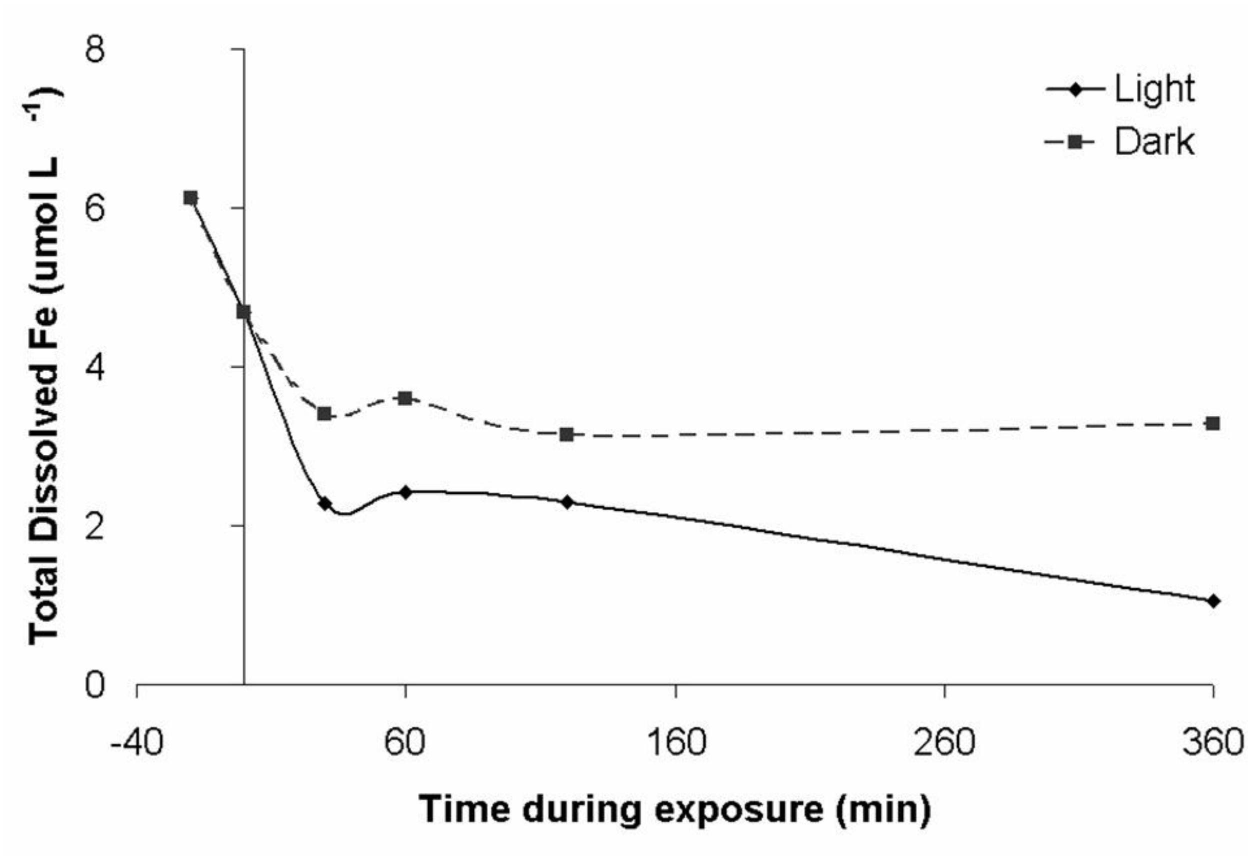


Figure 3.9: Fe removal rates for sediment and river water mixtures in light and dark conditions.

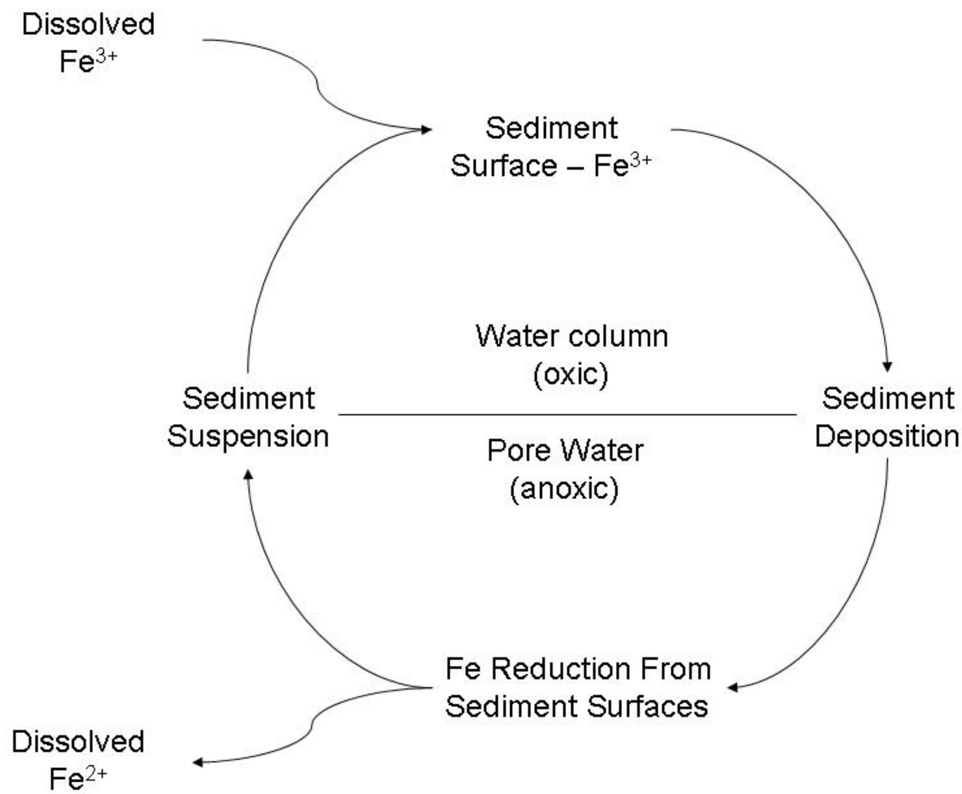


Figure 3.10: Fe and fine sediment interactions in river ecosystems. Dissolved Fe is adsorbed to sediment surfaces as Fe^{3+} during suspension. Following deposition and anoxia, Fe is reduced microbially and resulting in high pore water concentrations of Fe^{2+}. Mineral surfaces are again available for Fe^{3+} adsorption following resuspension.

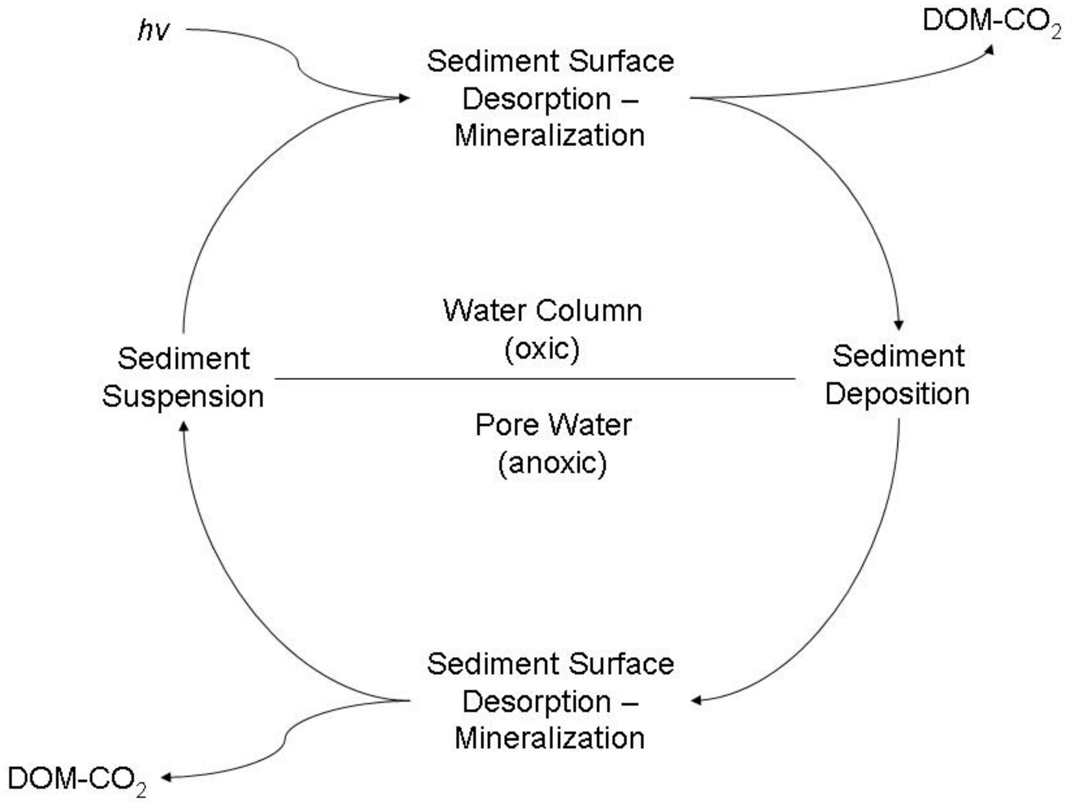


Figure 3.11: Photoassisted DOM desorption from suspended sediments in river ecosystems. Fine sediments -originating from watershed hill slopes- with OM adsorbed to mineral surfaces are suspended during flood events. In the presence of light (*hv*), DOM is desorbed from mineral surfaces. Sediments are redeposited on channel beds where microbial OM decomposition further reduces OM content of sediments. Supplies of OM-rich sediments are continuously supplied along river channel length by hill slope erosion.

CHAPTER IV: PHYSICAL AND PLANT COMMUNITY CONTROLS ON NITROGEN AND PHOSPHORUS LEACHING FROM IMPOUNDED RIVERINE WETLANDS FOLLOWING DAM REMOVAL

ABSTRACT

This study investigated an impounded riverine wetland complex on the Little River, North Carolina before and after the removal of a low-head dam. We quantified the leaching of interstitial N and P to the adjacent river channel during wetland dewatering, and clarified differences between physical (soil) and biological (plant) controls on N and P leaching from dewatering impoundment sediments. We found that the rate and quantity of N and P leaching from impounded dewatering sediment is predominately controlled by sediment porosity and specific yield. Plant controls on N and P leaching were significant but minimal during the first growing season following dam removal.

INTRODUCTION

Former reservoirs: variable sources of downstream disturbance

The removal of low head, run-of-river dams may result in downstream ecological disturbances of varying magnitude caused by increased downstream fluxes of sediment, nutrients and organic matter (for examples see: Bushaw-Newton et al., 2002; Doyle et al. 2003; Sethi et al., 2004; Ahearn and Dahlgren, 2005; Riggsbee, 2006). Further, it is becoming apparent that such downstream disturbances are dependent on the frequency and magnitude of material export from former impoundments (Stanley and Doyle, 2003;

Ahearn and Dahlgren, 2005; Riggsbee, 2006). Such materials consist mainly of sediment, nutrients and organic matter, and all can have profound consequences on downstream ecosystems. In particular, the export of N and P to sensitive receiving waters such as coastal ecosystems and reservoirs may contribute to ongoing eutrophication problems.

Materials are stored differently within impoundments based on regional and structural attributes of reservoirs (Riggsbee, 2006). Many reservoirs produce wide impoundments relative to natural channel widths, creating conditions conducive for sediment storage across the width of the impoundment. In these systems, materials are exported following dam removal as a new channel forms and equilibrates within the stored reservoir sediment, often leaving significant accumulations of nutrient-rich, fine sediments lateral to the active channel un-eroded (Doyle et al., 2003a). These channel adjustments following dam removal can lead to the excavation of substantial quantities of materials to downstream environments, but a considerable portion can remain within the impoundment. In Wisconsin, such channel adjustments resulted in the evacuation of ~ 14% (40,000 m^3) of the stored sediment following the removal of Rockdale Dam on the Koshkonong River (Doyle et al., 2003a), with the other portion of sediment remaining in place lateral to the newly formed channel.

In other systems where the impoundment width is similar to the natural channel width, impoundments may not be effective at sediment storage, or may store appreciable quantities of sediment and water outside of the active channel in low-lying floodplains such as riverine wetlands (Riggsbee, 2006). Such accumulations are less likely to be exported from the impoundment because of their position relative to the active river channel (Riggsbee, 2006). In both impoundment types, it seems that appreciable quantities of stored sediment will remain within the former reservoirs after dam removal.

Fine sediments in river ecosystems are sorptive sinks for organic matter (OM), soluble reactive phosphorus (SRP) and NH_4 (Meyer, 1979; McDowell and Wood, 1984; Klotz, 1988; Nelson et al., 1993; Triska et al., 1994; Mulholland, 1992). Thus, sediment accumulations exposed by dam removal represent appreciable potential sources of N and P to downstream environments. Fluxes of N and P from fine sediment will be exported from reservoirs after dam removal via erosion or interstitial dewatering. While the mobilization of formerly impounded sediment represents a potentially substantial source of nutrients to downstream, interstitial dewatering may also be a significant portion of the total N and P fluxes from impoundments following dam removal. To date, most dam removal research has been coarsely focused on the whole impoundment as a source of downstream loading (Stanley and Doyle, 2002; Doyle et al., 2003a; Ahearn and Dahlgren, 2005; Riggsbee, 2006). However, little attention has been focused on the specific roles of immobilized impoundment sediments as prolonged sources of downstream nutrient enrichment and how nutrients are mobilized as part of these stored sediments.

Previous research has revealed that exposed sediment accumulations within former impoundments are rapidly colonized by opportunistic plant communities, which are supported by nutrient-rich pore water and mineral surfaces (Shafroth, 2002; Orr and

Stanley, 2006). Thus, plants may represent an appreciable demand for interstitial N and P. However, the ability of these communities to control N and P leaching may be limited. For example, NO_3 leaching was observed from sediment accumulations with and without plant communities following dam removal on Murphy Creek, CA (Ahearn and Dahlgren, 2005). Thus, questions have begun to emerge regarding the role of fine sediment accumulations in nutrient export. In particular, little is known about the mechanisms that control N and P fluxes from dewatering sediments or about the quantity of N and P derived from such sediments following dam removal.

Physical and biological controls on N and P leaching following dam removal

Following the exposure of impounded fine sediments, physical properties may exert some control over the rates and quantities of N and P leaching to adjacent channels during interstitial dewatering. Among these physical properties are mineral surface sorption, porosity and specific yield, all of which are directly related to sediment grain size. Fine sediments (i.e., clay and silt) are expected to have greater surface area for surface adsorption of NH_4 and SRP, high porosity and low specific yields (Todd and Mays, 2005). Such properties associated with fine sediment will result in high interstitial water volume with appreciable N and P concentrations, which will be slowly released down gradient following dam removal.

In addition to physical properties influencing water release from sediments, plants which rapidly colonize exposed sediments may also exert considerable controls on the leaching of N and P. Floodplain plant colonization rates are likely to be high initially following dam removal (Shafroth 2002; Orr and Stanley, 2006), generating a demand for interstitial N and P pools. This should result in the assimilation of inorganic N and P into plant tissues, which will reduce the overall rate of export from dewatering sediments. Additionally, in saturated riverine wetland sediments, plants can contribute O_2 to the rhizosphere which may produce steep redox gradients (Wetzel, 2001). Biogeochemical transformations resulting from altered redox gradients can lead to the nitrification of NH_4 to NO_3, the most mobile form of inorganic N. Thus, increased plant biomass may have divergent effects on the leaching of interstitial nutrients: sequestration into plant tissue leading to longer-term retention or transformation via nitrification leading to rapid mobilization.

Purpose and structure of paper

We investigated an impounded riverine wetland complex on the Little River, North Carolina before and after dam removal. We collected data describing riverine wetland plant community biomass, interstitial and surface water biogeochemistry, groundwater table elevation and various physical measures of wetland sediments/soils. Collectively, these data were used to quantify the leaching of interstitial N and P to the adjacent river channel during wetland dewatering. These data were also used to clarify differences between physical (soil) and biological (plant) controls on N and P leaching from ‘new floodplain’ sediments/soils following dam removal. Further, because the dam removal literature is dominated by examples of lateral, non-wetland sediment exposure, we present a comparative analysis of N and P leaching from wetland and non-wetland sediment accumulations. Our results are useful for assessing the potential use of vegetation as a method of reducing the impact of dam removal on downstream water quality impacts.

STUDY SITE

Little River and Lowell Mill Dam

The Little River watershed drains approximately 600 km^2 within the lower piedmont and upper coastal plain physiographic regions of the Middle Neuse River basin in North Carolina. Land use within this portion of the Neuse Basin is comprised of 44% forest, 39% agriculture, 12% wetland and 5% developed. The Lowell Mill Dam impounds nearly 8 km of this 4th order stream located on the border of the piedmont and coastal plain regions (Figure 4.1). This particular dam was a low-head, run-of-river structure constructed ca. 1902 of brick and concrete, which provided ~ 3 m of head storage within the channel (dimensions are 76 m x 3.5 m) for grist mill operations. Impoundment channel bed sediments were a matrix-supported sand and gravel mix with a thin veneer of fine sediments (< 1%).

Impounded riverine wetlands: pre and post-removal

Approximately 200,000 m^2 of floodplain wetland habitat were impounded by Lowell Mill Dam near the confluence of the Little River and Little Buffalo Creek (Figure 1), creating a mean depth of inundation of ~1 m. Prior to the removal, ~44% of impounded water was stored within the riverine wetland complex, while the remaining ~56% of impounded surface water was stored within the channel (Riggsbee, 2006). During more frequent flood events (Q_{1-2}) floodplain and riverine wetland areas act as depositional features, retaining organic matter and alluvial sediments. The geomorphology of the wetland network impounded by Lowell Mill Dam is characteristic of groundwater slope swamps (as described by Mitsch and Gosselink, 2000). Several groundwater inputs emerge from seeps and springs across the wetland, and surface waters inundate the wetland complex during floods. Prior to dam removal, this wetland system was permanently inundated with minimal exposure of wetland surfaces during extreme low flow conditions. Additionally, a small, poorly-defined thalweg with a mean depth of 35 cm drained the impounded wetland into the adjacent river channel.

The impoundment was dewatered by modifying and opening a series of flood gates leading to the dam's wheel housing cells. Gates were removed on 21 July 2004, 07 August 2004 and 28 April 2005. The most dramatic alteration of wetland hydrology was seen following the 28 April 2005 gate removal (Riggsbee, 2006). Following impoundment dewatering, the dam was breached to grade by completely removing the wheel housing (15 December 2005). The structure was completely removed using small, controlled blasts which fractured the structure in order for heavy equipment to remove materials directly from the channel (28 December 2005). During the dewatering process (28 April 2005), impounded wetlands drained surface waters in approximately 3 hours following gate removal (Figure 4.2). As wetland hydrology was altered by the dam dewatering, and initiated rapid plant colonization of exposed wetland soils, this date is hereafter referred to as dam removal. Additionally, this is the point at which data are analyzed for statistical differences (e.g. pre vs. post analyses).

Surface water was completely drained from the research site immediately following dam removal (approximately 3 hours). Other impounded areas of the riverine wetland complex were also drained of significant portions of surface water. Stored groundwater drained into the previously described thalweg at low discharge values of ~ 0.02 m^3s^{-1}, and has continued at a similar rate up to the present. Over time, the thalweg became more defined through a series of small head-cuts (5-10 cm in depth). Currently, there are two distinct wetland channel geometries controlled by upstream headcut migration: mean channel dimensions below the head-cut are 15 cm in width and 10 cm in depth, while mean channel dimensions above the head-cut are 35 cm in width and 4 cm depth. Portions (< 25%) of the wetland surface are continuously saturated in regions of groundwater upwelling.

METHODS

To differentiate between physical and biological controls on N and P leaching from exposed sediments, a series of physical, chemical and biological data were collected and analyzed from 9800 m^2 of impounded riverine wetlands before and after dam removal. First, water table elevation measurements were collected to determine the rate of sediment dewatering. Second, vegetation plots (with and without plants) were used to quantify the influence of plant community biomass on interstitial N and P concentrations. Third, wetland surface water draining into the adjacent river channel was monitored for N and P concentrations to determine if wetland contributions to the river main stem changed with time following the dam removal. Finally, physical properties of the wetland sediment/soil such as porosity and specific yield were used to construct N and P fluxes to the adjacent river channel during the interstitial dewatering process.

Vegetation experimental design

Prior to the dewatering of the impoundment, 14 vegetation plots were established in the study wetland (Figure 1 inset). Plots were constructed using a series of curtains made of corrugated metal roofing cut into 1 m x 1 m squares. These curtains were used to prevent encroachment of exterior roots, which could affect in-plot biogeochemistry. Curtains were installed by pushing them approximately 0.5 m into the impounded wetland sediments. Plot dimensions measured 1 m in width and 5 m in length, creating a surface area of 5 m^2. Plots were established on both sides of the wetland thalweg oriented perpendicular to channel flow in order to accommodate groundwater flow paths. Following dam removal, seven randomly selected plots were maintained as barren wetland soils by physically removing all plant materials on a monthly basis. The remaining seven plots were allowed to accrue plant biomass without manipulation.

Groundwater table elevation

A network of 50 shallow piezometers was established in a grid pattern enveloping the 14 vegetation plots described above. Piezometers were organized in rows of 5 and columns of 10 with respect to the adjacent river channel. Piezometers within rows were spaced approximately 6 m apart, while those within columns were spaced approximately 10 m apart. PVC pipe measuring 3.05 m in length and 1.9 cm in diameter was used for piezometer construction. The lower 20 cm of each was perforated and covered using

“Drain-Sleeve” (Cariff, Inc.) to prevent clogging with fine, saturated wetland soils. Water table elevation measurements were taken once a month during the growing season (April to October, 2005) and once again following the complete senescence of the floodplain plant community (February, 2006).

Plant community biomass

Plant biomass was sampled monthly along three transects established perpendicular to the wetland thalweg. Transects were spaced approximately 50 m apart, and located at the upstream, center and downstream portions of the wetland (relative to wetland thalweg). Each month a new transect was established 2-3 m upstream or downstream of the previously sampled transect to limit measurement errors associated with destructive sampling. Five quadrats (0.25 m^2) were sampled per transect during a given sample period; sampling consisted of removing all above and below ground tissues. Quadrat samples were separated into above and below ground living biomass, dried at 100^oC for at least 24 hours, weighed, subsampled, combusted at 550^oC for at least 24 hours and reweighed (Wetzel and Likens, 2000). Biomass is reported as gCm^{-2}. These measurements represent plant biomass within plots which were not manipulated.

Interstitial and surface water biogeochemistry

Samples for interstitial concentrations of dissolved inorganic nitrogen (DIN) and soluble reactive phosphorous (SRP) were collected using interstitial water samplers, hereafter referred to as IWS (Roden and Wetzel, 1996; Winger et al., 1998). Each IWS was constructed of acrylic with ten 40-ml wells spaced at 2 cm intervals. All ten wells were filled with deoxygenated Milli-Q water (> 17 MΏ), covered with a moistened polycarbonate membrane (pore size of 0.22 μm; Whatman/Nucleopore track-etch membrane) which was secured in place. IWS were transported into the field in a solution of deoxygenated water that was continuously stripped using N_2 compressed gas. Once an IWS was deployed, wells were allowed to equilibrate with interstitial water for at least 15 days before sample collection. Equilibration was driven by diffusion of DIN (as NO_3-NO_2 and NH_4) and SRP across the polycarbonate membrane into each well. Samples were recovered using a syringe and transferred immediately into acid washed 60 ml amber HDPE bottles (Nalgene), acidified to pH 2 using 2M ultrapure HCl, and stored on ice during transport to the lab. Upon reaching the lab, samples were frozen at -20^{o}C until analyzed for DIN and SRP by Waters Agricultural Laboratory in Camilla, GA using standard EPA methods (250.1 for NH_4-N, 353.1 for NO_2 & NO_3-N, and 200.7 for PO_4-P). For purposes of interpreting plant community controls on interstitial biogeochemistry, samples were separated into two categories: upper (4-12 cm) and lower (14-22 cm) wetland soils. This separation was based on the maximum observed rooting depth of 10 cm.

IWS have typically been used in completely saturated conditions. Wetland sediments during our study were completely saturated during baseline sampling, and variably saturated following dam removal, with the majority of areas being completely saturated. Each vegetation plot was located directly adjacent to the wetland thalweg, which provided greater saturation than reported mean groundwater table elevations. IWS were never deployed into soils that were not saturated, nor were samples collected from wells that experienced noticeable desiccation (desiccation occurred in only 2 of 120 samples collected after dam removal). Additionally, groundwater drawdown was reset once during the growing season because of local beaver activity (discussed in subsection 4.2 below). Data were not collected in September or October because of noticeable reductions in soil saturation.

Inundated wetland sediments were analyzed for porosity and particle size using methods described in Dane and Topp (2002). Physical measures were used to provide parameters for reach-scale extrapolations for wetland N and P loading to the adjacent river channel.

Wetland surface water samples were collected at a pre-determined sampling station positioned 20 m within the wetland, just upstream of the wetland's confluence with the Little River. Sample collection occurred monthly during baseline and post-removal efforts (April to October, 2005). Samples were collected in acid washed, amber 125 ml HDPE bottles (Nalgene) following filtration in the field (Whatman GF/F). Samples were stored on ice during transport to the lab, where they were frozen and analyzed as detailed above for interstitial samples.

Analyses

Statistical analyses were performed using analysis of variance tests controlling for time and presence or absence of vegetation for interstitial biogeochemistry. Assumptions made include: 1) there are no statistical differences (e.g., all variation is random) between vegetation plots previous to dam removal, 2) there are time effects for post-removal samples only, and 3) each population of samples fits a normal distribution. Analysis for wetland drainage (surface water) biogeochemistry was accomplished using analysis of variance tests controlling for time assuming equal variance. All analyses were performed using the SAS 9.1.3 statistical software package.

RESULTS

Plant biomass

Plant biomass stored in aboveground tissues showed a nearly exponential rate of increase during the first 4 months of wetland recovery followed by the initiation of senescence in October and complete senescence by February 2006 (Figure 4.3a). In contrast, belowground biomass initially increased, but then there was a slight decrease in the month following groundwater table elevation lows (discussed further below) Belowground tissues stayed remarkably consistent with aboveground biomass until August 2005, at which time aboveground biomass outpaced belowground tissue accrual. Maximum aboveground biomass was nearly an order of magnitude greater than that of belowground biomass, 484 and 4130 gCm^{-2}, respectively (Figure 4.3a).

Groundwater table elevation

Monthly mean groundwater table elevations across the 9800 m^2 wetland study area generally decreased following dam removal (Figure 4.3b). During April (pre-removal during baseflow conditions, ~2.5 m^3s^{-1}), wetland hydrology was dictated by river stage. Mean groundwater levels were 76 cm above the wetland soil surface. Following dam removal, the mean groundwater table elevation declined to -22 cm (relative to the soil surface) in June. An increase in the mean groundwater table elevation was seen in mid-July because of the construction of a beaver dam at the outlet of the wetland thalweg. The beaver dam was in place for less than two weeks, and was immediately removed once detected. The beaver dam was not reestablished following its removal. The resulting mean groundwater table elevation during July was 2 cm above the wetland soil surface. The lowest groundwater table elevations were seen in August 2005 at a level of -38 cm below the wetland soil surface. Water table minima coincided with high plant biomass (biomass trends discussed above), and is explained by high evapotranspiration rates in the absence of measurable precipitation. There was a slight rise in groundwater elevation from August to October, which coincided with the onset of plant community senescence.

Interstitial and surface water biogeochemistry

Statistical analysis was first performed to compare across wetland interstitial concentrations before and after removal, thus ignoring plant effects. For the baseline data set, there were no significant differences between mean upper (4-12 cm) and lower (14-22 cm) portions of the wetland sediments for either DIN or SRP concentrations (Figure 4.4). Following dam removal, mean concentrations of NO_3-NO_2-N increased significantly ($n = 60$, $p < 0.001$) within the upper and lower wetland soils nearly 2.5-fold compared to pre-removal concentrations. Conversely, mean concentrations for NH_4-N in upper and lower soils across the wetland decreased significantly ($n = 60$, $p < 0.001$), by almost a 3-fold decrease. Likewise, PO_4-P within upper and lower soils of the wetland significantly decreased 3-fold ($n = 60$, $p < 0.001$).

Additional statistical analyses were used to determine whether plants exerted significant controls on interstitial biogeochemistry. The floodplain plant community exerted significant, yet minimal controls on interstitial biogeochemistry within the upper portion of the recovering wetland soils following the dewatering of the impoundment. There were no significant effects of plant presence on the biogeochemistry of the lower wetland soils. However, the last data points collected in August 2005 are suggestive of an emerging trend that plant presence was driving transformations in N pools and lowering P concentrations within lower wetland soils. Within the upper soils, floodplain plant community presence exerted some control on NH_4-N and PO_4-P concentrations (Figure 4), but not on NO_3-NO_2 –N concentrations: unvegetated plots showed mean NH_4-N concentrations 3.5 times greater than vegetated plots ($n = 40$, $p < 0.001$), and PO_4-P concentrations were 2.5 times higher in unvegetated plots than vegetated plots ($n = 40$, $p < 0.001$).

Wetland surface water draining into the adjacent river channel delivered appreciable amounts of N in the form of NH_4 to the adjacent river channel (Figure 4.5). Post-removal wetland drainage concentrations of NH_4-N were an order of magnitude greater than pre-removal concentrations ($n = 40$, $p < 0.001$). Conversely, there were no temporal effects seen for NO_2-NO_3 -N or PO_4-P concentrations within wetland surface water entering the Little River main stem. It is apparent that wetland surface water

(Figure 4.5) and interstitial water affected by the burgeoning plant community (Figures 4.4) were not chemically similar.

DISCUSSION

Physical controls on wetland nitrogen and phosphorous leaching

Based on our data, the rate at which the top 25 cm of wetland soils dewatered was faster than the rate of plant colonization (Figure 4.3). Therefore, a developing plant community could not limit the quantity of N or P that entered the Little River main stem during the initial phases of sediment dewatering. Based on this assessment, plant controls on the immobilization of N and P from formerly impounded sediments are limited by the rate of colonization relative to the rate of soil dewatering. Instead, it was the physical properties such as specific yield and porosity of wetland soils that controlled the quantities of N and P delivered to the channel during the initial phases of sediment dewatering.

Since it is apparent that some interstitial water within sediment accumulations will inevitably be released to adjacent river channels, it is important to understand how the magnitudes of potential interstitial N and P fluxes compare to riverine fluxes of N and P. Theoretical maximum porewater N and P loads in the top 25 cm of wetland sediments can be estimated using mean baseline concentrations of interstitial N and P and the total porosity of the wetland sediments prior to dewatering (neglecting evaporation for ease of analysis). This approach provides a high-end calculation for channel loading from exposed, fine sediments immediately following dam removal. Prior to dewatering, wetland sediments were composed of mostly silt (68%) and clay (28%), and the measured total porosity was high at 96%. The porosity of the impounded wetland sediments was extremely high because with the dam in place, the sediment surface was a saturated, recently suspended and deposited flocculent layer; unsaturated sediments of this type normally have porosities of 42-46% (Todd and Mays, 2005). Mean NH_4-N and NO_3-NO_2-N concentrations were 15.3 and 0.82 mg/L respectively, or 16.12 mg/L for DIN. The wetland surface area studied during this investigation was 9800 m^2, and the depth of sediments which dewatered prior to significant development of plant biomass was 25 cm, or a sediment pore water control volume of ~2350 m^3. Thus, the study wetland contained 38 kg of DIN and 1.6 kg of SRP before the impoundment dewatered. This gives an average of 3.8 g of N m^{-2} and 0.16 g of P m^{-2} throughout the wetland complex. If it is assumed that this study wetland is comparable to the total 200,000 m^2 of riverine wetlands impounded by Lowell Mill Dam (Figure 4.1), then the total DIN and SRP content in rapidly dewatered wetland soils equaled 774 kg and 33 kg, respectively. Based on the high silt and clay content of the wetland sediments, the maximum specific yield of interstitial water from these saturated wetland sediments would be 8% (from Todd and Mays, 2005). This then suggests that an approximate load derived from the interstitial waters of the impounded riverine wetlands would be 62 kg of DIN (8% of 774 kg) and 2.6 kg of SRP (8% of 33kg), respectively. Smaller values of sediment porosity for more consolidated wetland sediments (i.e., < 96%) would further reduce the potential loads from these areas. It is likely, based on silt and clay content, that wetland sediments were slow to drain because of low hydraulic conductivity. These sediments likely released their available interstitial waters (specific yield) over the course of several days to a few weeks.

It is also important to consider the size of the N and P loads from interstitial waters relative to the loads in the main channel. Based on continuous discharge and water quality monitoring upstream of the impoundment, we found that the river load of DIN entering the impoundment was 90 kg over the 30 hour period of 28 April 2005 (Riggsbee, 2006). Thus, the DIN load from the interstitial waters was over half the incoming riverine load. However, because of the low hydraulic conductivity of these sediments, the rate of water release from the wetland sediments would be extremely low. As the N load associated with sediment leaching was likely delivered over the course of several days, the daily flux would enrich the river channel, but only slightly. Slight enrichment of the N fluxes exiting the impoundment, relative to input loads, was indeed observed on many occasions (Riggsbee, 2006). Essentially, wetland sediment specific yield controlled the initial loss and retention of interstitial N from the rooting zone before plants were able to colonize the site.

Riverine P input load data are not available for comparison with theoretical interstitial P loads. Based on surface water samples presented here (Figure 4.5b and 4.5d), it is likely that interstitial P exports from the dewatering sediments were controlled by mineral surface adsorption as increased concentrations were not detectable in receiving channel waters. Additionally, iron oxidation was clearly visible on the surface of wetland sediments during the dewatering (28 April 2005), and iron oxides were consistently visible on wetland channel sediments following dam removal. Iron oxides are known to bind PO_4 in freshwater sediments, making P biologically unavailable (Wetzel, 2001).

Plant controls on channel nitrogen and phosphorus leaching

Plant communities rapidly colonize nutrient-rich exposed sediments, becoming the new floodplain and riparian communities (Orr and Stanley, 2006). Our data demonstrate this point as the mean plant colonization rate was 34 $gCm^{-2}d^{-1}$ (including both above and belowground biomass; Figures 4.2 and 4.3). In the initial days to weeks following dam removal, these floodplain communities exerted little control over sediment biogeochemistry. In this study, the successional plant community did not affect N loads entering the channel from the wetland during sediment or groundwater dewatering. Further, the plant community only affected N and P concentrations within the upper 14 cm of the dewatering wetland, which is insignificant compared to the depth of most lateral or floodplain wetland sediments exposed following dam removal. Even in impoundments created by small dams, over 1-2 m of sediment can accumulate along channel margins such as the Deep River, NC (personal observation) and the Koshkonong River, WI (Doyle et al., 2003a). It is likely that there are multiple flow paths among such sediment accumulations, which are deeper than the typical rooting zone. Thus, the role of the new floodplain plant community on influencing nutrient concentrations within sediments exposed by dam removal is limited, at least during the first growing season. This then suggests that vegetation will have a limited role on controlling the initial and short-term (i.e., within months of dam removal) flux of nutrients from interstitial waters following dam removal.

However, vegetation controls on N and P fluxes are likely to become more important over longer timescales. While the plant community in our study was not effective at reducing interstitial N and P leaching to local channels during the first growing season following dam removal, plants in general are extremely effective at sediment stabilization. Plant roots provide bank strength by reducing porewater pressures, altering bank hydrology and flow hydraulics as well as providing supplemental strength to bank materials (Abernethy and Rutherfurd, 2000, 2001). This property of floodplain plant communities is important in the case of dam removal (Doyle et al., 2003b), as mass wasting of formerly impounded fine sediment accumulations can release appreciable quantities of particulate N and P as well as TSS from formerly impounded reaches. Plants may also limit loading of N and P from the new flood plains to adjacent channels during periodic re-wetting events by generating a demand for N and P from soil pools for tissue development, which will reduce the amount of DIN and SRP mobilized during these events. Additionally, plant communities reduce the quantity of N and P available for transport by supporting microbial communities, which immobilize both N and P (Baldwin and Mitchell, 2000). Thus, biological controls are more likely to regulate the magnitude of nutrient pulses from former impoundments over longer timescales (years to decades). Such biological controls are a direct result of ecosystem succession on the new floodplain, which promotes both sediment stability and community sequestration of accumulated N and P (Vitousek and Reiners, 1975).

Wetland and non-wetland variable source areas within recovering impoundments

While this investigation focused on physical and biological controls on interstitial biogeochemistry in formerly impounded riverine wetland soils, the results are applicable to non-wetland, lateral sediment accumulations within formerly impounded channels. Both environments are typically characterized by fine, saturated, nutrient-rich sediments with groundwater flow paths capable of contributing interstitial N (and perhaps P) to adjacent channels. Differences between these two source area types are not numerous, but are significant. In the case of non-wetland, lateral sediments, the pioneer plant community may "crash" following the complete desiccation of the new floodplain soils (Orr and Stanley, 2006). While wetlands, on the other hand, will be able to provide hydrological stability to their plant and soil microbial communities as variations in soil saturation are limited. Complete desiccation of sediments can kill soil microbes and decrease mineral surface affinity for P. These two consequences can result in considerable flushing of N and P following precipitation events which rewet desiccated soils (Baldwin and Mitchell, 2000). Thus, the complete desiccation of lateral sediment accumulations within formerly impounded channels leads to contributions of N and P to downstream environments that would not be contributed by riverine wetland sediments, which are less variably saturated.

While wetland source areas are less likely to contribute N and P because of reduced saturation variability, they may represent a significant source of N and P during dewatering following dam removal. Much like surface water, groundwater flow paths may also be affected by the presence of downstream dams. For example, networks of small dams are used to recharge aquifers in arid Middle Eastern countries (Abdelrahman and Abdelmagid, 1993; Alhassoun and Alturbak, 1995; Abu-Taleb, 2003). Dewatering groundwater supplies within the impounded riverine wetlands on the Little River likely contributed more N-rich water to the channel than lateral channel sediments isolated from groundwater flow paths. It is apparent that the NH_4-N and NO_3-NO_2-N concentrations within wetland drainage waters (Figure 4.5) are not similar to interstitial concentrations (Figure 4.4). Groundwater released into the channel was possibly 'old' impounded groundwater delivered via deep flow paths not associated with the wetland soils supporting plant colonization. Old groundwater is typically characterized by higher concentrations of solutes compared to new groundwater, which is solute poor (Burt and Pinay, 2005). In the case of Lowell Mill, the dam impounded the riverine wetland complex for more than 100 years. Over this time, considerable groundwater supplies were accumulated, and these supplies were slowly released into the Little River main stem following dam removal. Thus, wetland groundwater dewatering on the Little River represented a slow, constant source of N which could persist for years to decades.

It is not apparent which variable source area within impoundments can be expected to contribute more N and P to adjacent channels. It is clear, however, that the respective frequencies of such contributions are different. On one hand, lateral channel sediment desiccation can lead to plant community crashes, which reduces sediment stability while increasing soil N and P mobility during episodic rewetting events. On the other hand, wetland sediments can slowly and continuously release appreciable loads of N, over long time periods, from dewatering groundwater following dam removal.

Regardless of their relative magnitude of N and P loading, both source areas are likely to contribute significantly to downstream nutrient enrichment.

CONCLUSIONS

Lateral channel and riverine wetland sediment grain size within impoundments directly affects nutrient affinity, porosity and specific yield of sediment pore water following dam removal. Thus, impounded sediment grain size can provide some insight into downstream disturbances following dam removal. Coarse sediments are likely to be nutrient-poor because of low surface area availability for ion sorption, and low porosity results in low interstitial water volume and rapid dewatering rates. Conversely, fine sediments exhibit strong mineral surface affinity for N and P, and high porosity provides high interstitial water volume and slow dewatering rates. These physical properties of impounded sediments control the magnitude and timing of loads of N and P delivered to adjacent river channels.

Plant community succession can sequester N and P, but this effect is temporally delayed and during the first growing season is limited to the rooting zone. Additionally, plant communities provide sediment stability that reduces the overall erosion of sediments, which leads to mass wasting of N and P-rich sediments. Thus, plant controls on N and P loading to adjacent channels is more important during episodic re-wetting and flood events because of nutrient immobilization and sediment stabilization. Impounded wetlands may also represent an additional source of N as formerly impounded aquifers dewater. These aspects of fine sediment accumulations behind reservoirs drive downstream disturbances following dam removal, which are directly related to excessive material export from former impoundments.

REFERENCES

Abernethy B and Rutherfurd ID. 2000. The effect of riparian tree roots on the mass-stability of riverbanks. *Earth Surface Processes and Landforms* 25: 921-937.

Abernethy B and Rutherfurd ID. 2001. The distribution and strength of riparian tree roots in relation to riverbank reinforcement. *Hydrological Processes* 15: 63-79.

Abdelrahman HA and Abelmagid IM. 1993. Water conservation in Oman. *Water International* 18: 95-102.

Abu-Taleb MF. 2003. Recharge of groundwater through multi-stage reservoirs in a desert basin. *Environmental Geology* 44: 379-390.

Alhassoun SA and Alturbak AS. 1995. Recharge dam efficiency based on subsurface flow-analysis. *Water International* 20: 40-45.

Ahearn DS and Dahlgren RA. 2005. Sediment and nutrient dynamics following a low-head dam removal at Murphy Creek, California. *Limnology and Oceanography* 50: 1752-1762.

Baldwin DS and Mitchell AM. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: A synthesis. *Regulated Rivers-Research and Management* 16: 457-467.

Bushaw-Newton KL, Hart DD, Pizzuto JE, et al. 2002. An integrative approach towards understanding ecological responses to dam removal: The Manatawny Creek study. *Journal of the American Water Resources Association* 38: 1581-1599.

Burt TP and Pinay G. 2005. Linking hydrology and biogeochemistry in complex landscapes. *Progress in Physical Geography* 29: 297-316.

Dane, J.H. and Topp, G.C., 2002. *Methods of Soil Analysis: Part 4 - Physical Methods*. Soil Science Society of America, Madison, 1692 pp.

Doyle MW, Stanley EH, and Harbor JM. 2003a. Channel adjustments following two dam removals in Wisconsin. *Water Resources Research* 36: 1147, doi: 10.1029/2003WR002038.

Doyle, M.W., Stanley, E.H., Selle, A.R., Stofleth, J.M., and Harbor, J.M., 2003b. Predicting the depth of erosion in reservoirs following dam removal using bank stability analysis. *International Journal of Sediment Research* 18: 115-121.

Klotz RL. 1988. Sediment control on soluble reactive phosphorus in Hoxie Gorge Creek, New York. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 2026-2034.

McDowell WH and Wood T. 1984. Podzolization: Soil processes control dissolved organic carbon concentrations in stream water. *Soil Science* 137: 23-32.

Meyer J. 1979. The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. *Limnology and Oceanography* 24: 365-375.

Mitsch WJ, Gosselink JG. 2000. *Wetlands*. 3rd Edition. John Wiley and Sons, Inc., New York. 920 pp.

Mulholland PJ. 1992. Regulation of nutrient concentrations in a temperate forest stream: Roles of upland, riparian and instream processes. *Limnology and Oceanography* 37: 1512-1526.

Nelson PN, Baldock JA, Oades JM. 1993. Concentration and composition of dissolved organic carbon in streams in relation to catchment soil properties. *Biogeochemistry* 19: 27-50.

Orr CH, and Stanley EH. 2006. Vegetation development and restoration potential of drained reservoirs following dam removal in Wisconsin. *Rivers Research and Applications* 22: 281-295.

Riggsbee JA. 2006. Spatial and temporal heterogeneity of impoundment nutrient and sediment fluxes following dam removal. PhD. dissertation. University of North Carolina at Chapel Hill.

Roden EE and Wetzel RG. 1996. Organic carbon oxidation and suppression of methane production by microbial Fe(III) oxide reduction in vegetated and unvegetated freshwater wetland sediments. *Limnology and Oceanography* 41: 1733-1748.

Stanley EH, and Doyle MW. 2002. A geomorphic perspective on nutrient retention following dam removal. *BioScience* 52: 693-701.

Stanley EH and Doyle MW. 2003. Trading off: the ecological effects of dam removal. *Frontiers in Ecology and the Environment* 1: 15-22.

Sethi SA, Selle AR, Doyle MW, Stanley EH, and Kitchel HE. 2004. Response of unionid mussels to dam removal in Koshkonong Creek, Wisconsin (USA). *Hydrobiologia* 525:157-165.

Todd DK and Mays LW. 2005. *Groundwater Hydrology*. 3rd Edition. John Wiley and Sons, Hoboken, NJ. 636 pp.

Triska FJ, Jackman AJ, Duff JH and Avanzino RJ. 1994. Ammonium sorption to channel and riparian sediments: A transient storage pool for dissolved inorganic nitrogen. *Biogeochemistry* 26: 67-83.

Vitousek PM and Reiners WA. 1975. Ecosystem succession and nutrient retention: A hypothesis. *Bioscience* 25: 376-381.

Wetzel RG. 2001. *Limnology: Lake and River Ecosystems*. 3rd Edition. Academic Press, San Diego. 1066 pp.

Wetzel RG and Likens GE 2000. *Limnological Analyses*. 3rd Edition. Springer-Verlag, New York. 429 pp.

Winger PV, Lasier PJ and Jackson BP. 1998. The influence of extraction procedure on ion concentrations in sediment pore water. *Archives of Environmental Contamination and Toxicology* 35: 8-13.

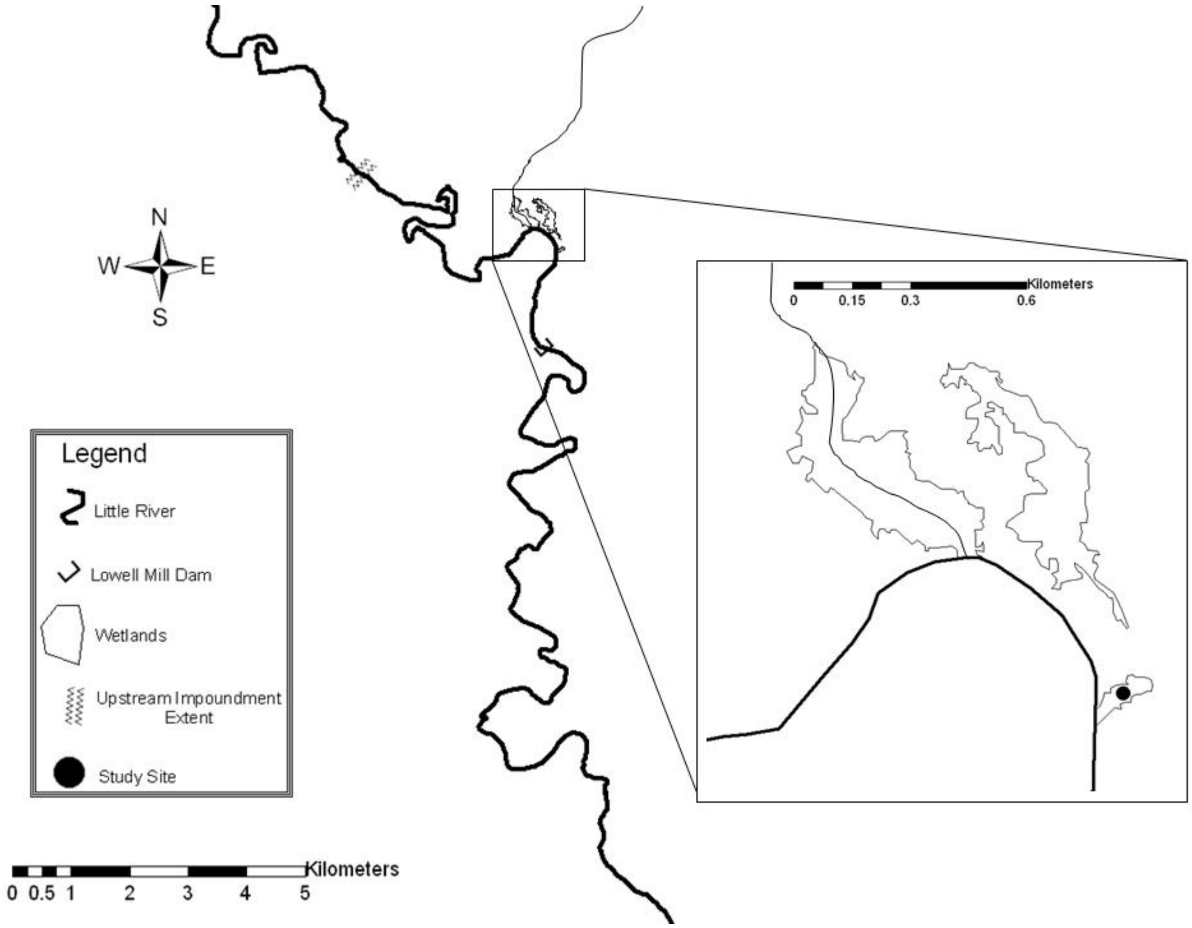


Figure 4.1: Site map of Lowell Mill Impoundment. Map inset shows the confluence with Little Buffalo Creek and the Little River. Study wetland is denoted within the inset with a small black dot.

Figure 4.2: Riverine wetland recovery following dam removal. A) Wetland previous to dam removal (25 April 2005), B) immediately following dam removal (28 April 2005), C) six weeks after removal (16 June 2005), and D) four months after removal (15 September 2005).

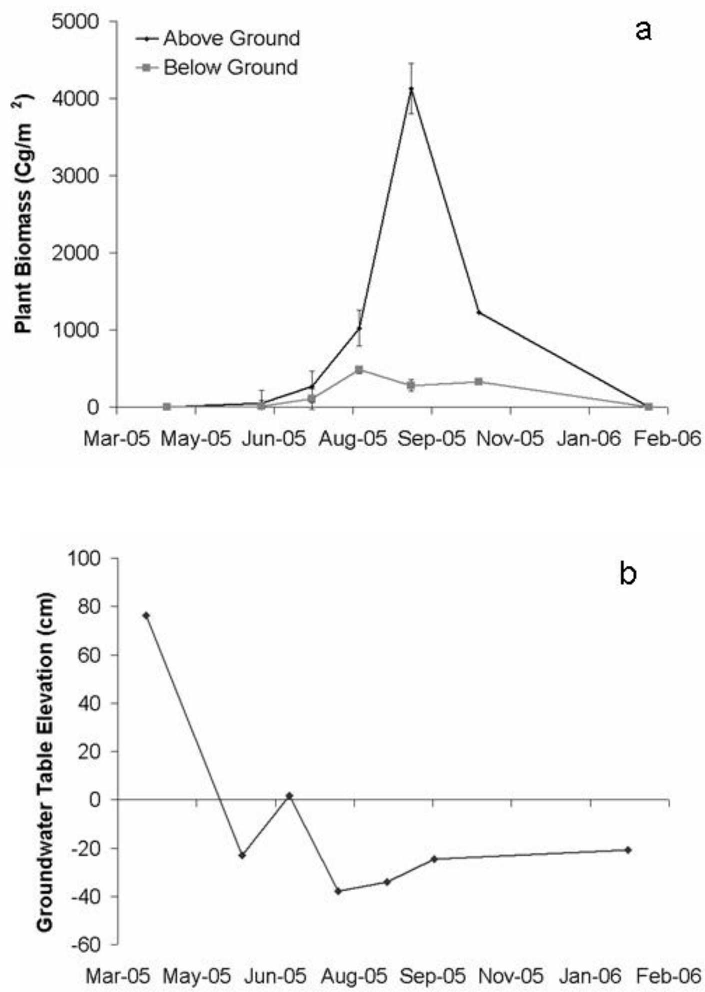


Figure 4.3: Riverine wetland plant biomass and groundwater table elevations following dam removal. A) Floodplain wetland plant biomass following dam removal. B) Mean groundwater table elevations. July mean elevations were influenced by beaver activity, while August and September levels were controlled by high evapotranspiration rates associated with high plant biomass. Error bars represent ± 1 SE.

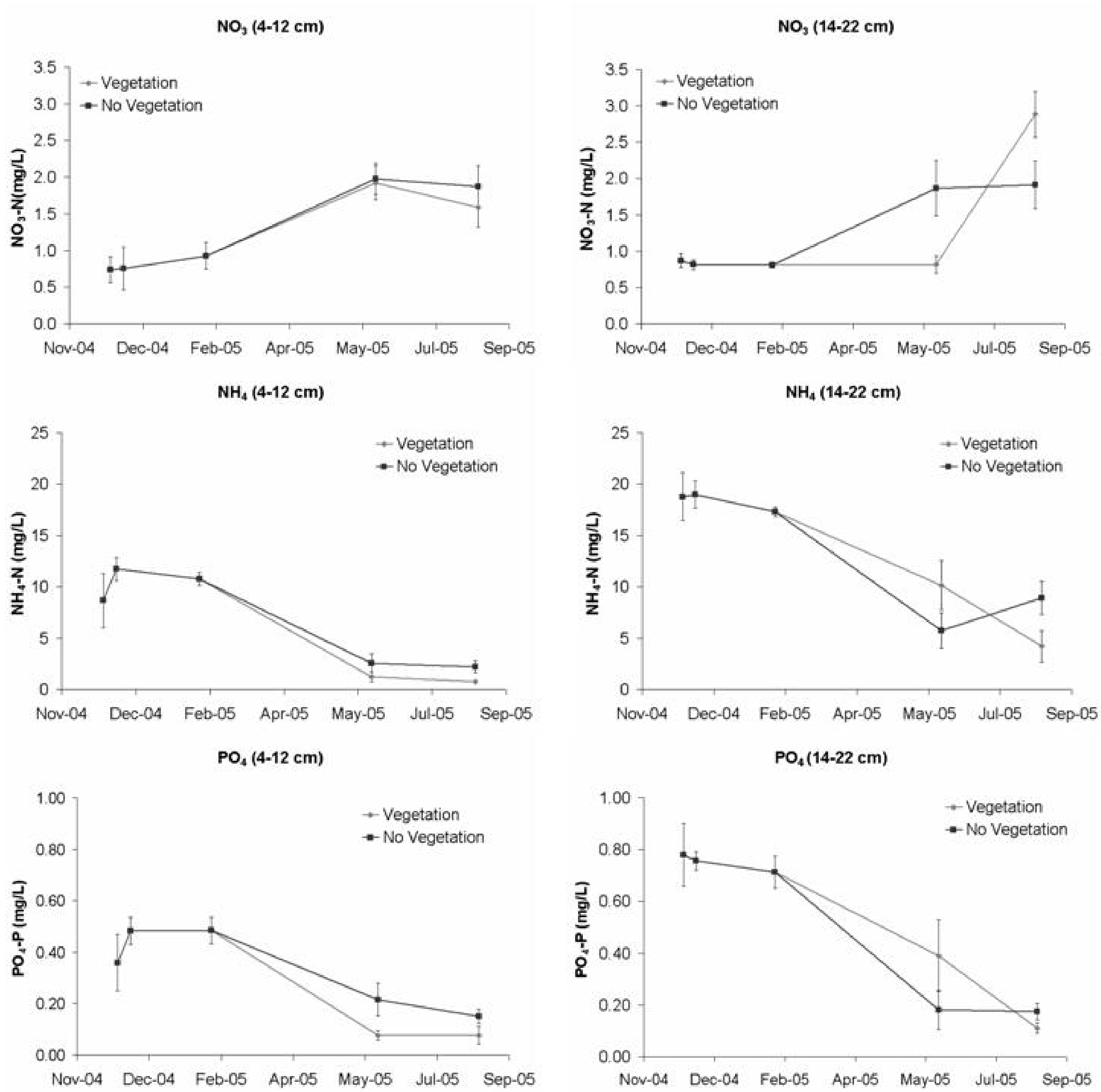


Figure 4.4: Plant community controls on riverine wetland interstitial biogeochemistry following dam removal. Error bars represent ± 1 SE

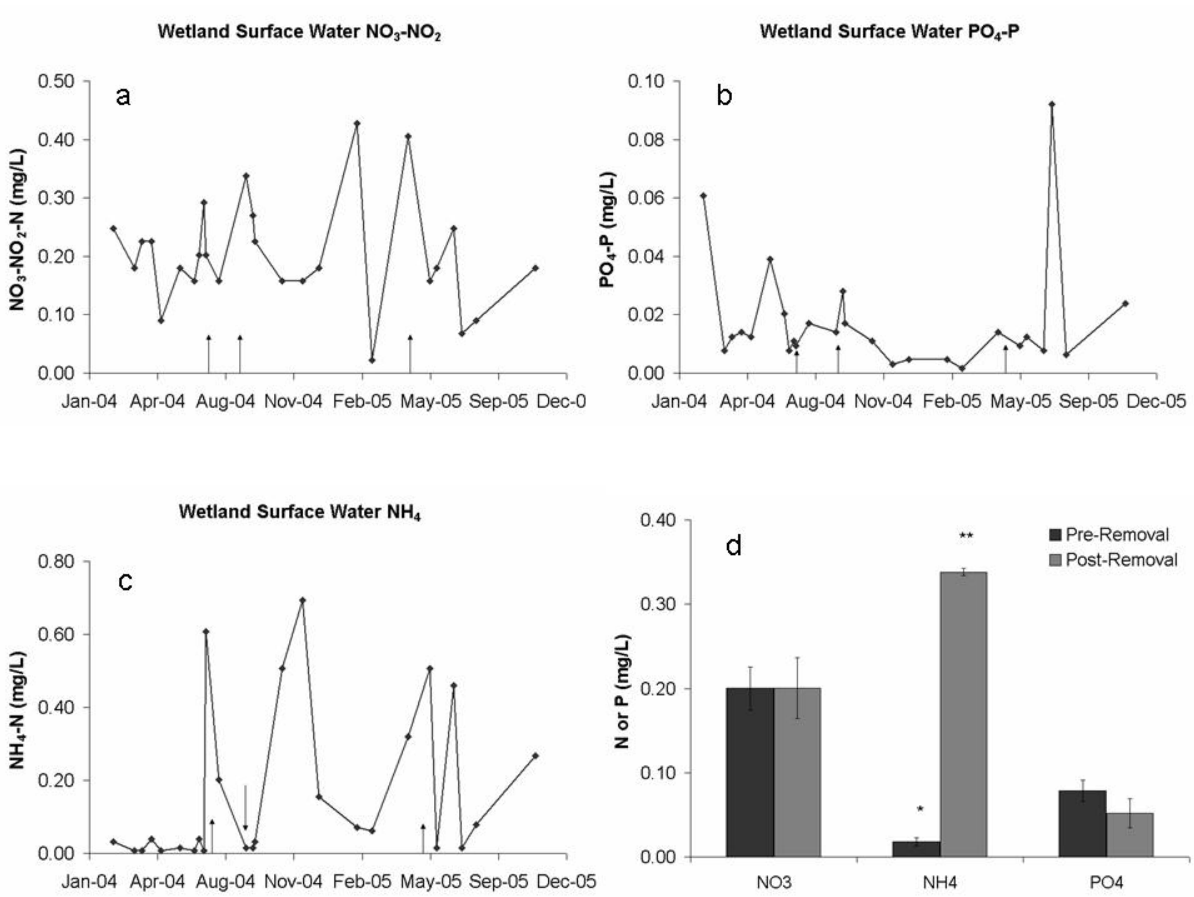


Figure 4.5: Wetland surface water drainage biogeochemistry dynamics. Arrows indicate gate removals. A) NO_3-NO_2-N, B) PO_4-P, C) NH_4-N, D) Summary of all surface water concentrations before and after dam removal. Error bars represent $\pm$1 SE.

CHAPTER V: DISTURBANCE DIVERGENCE: SPATIAL AND TEMPORAL HETEROGENEITY AMONG UPSTREAM AND DOWNSTREAM DISTURBANCES CAUSED BY DAM REMOVAL

ABSTRACT

Dams and dam removal can be viewed as ecological disturbances, which alter hydrogeomorphic, ecological and biogeochemical processes in rivers. Damming produces distinctly different physical and ecological effects on upstream and downstream reaches (Ward and Stanford, 1983; Petts, 1984). We assert that dam removal produces an equally divergent, bidirectional pattern of disturbance in river ecosystems, eventually reconnecting historically fragmented reaches. In an attempt to better understand and anticipate the geomorphic, ecological and biogeochemical implications of dam removal, we have provided conceptual models describing ecosystem disturbance and recovery following dam removal, incorporating both upstream and downstream responses. Additionally, we explore the spatial and temporal heterogeneity among the bidirectional disturbances and recovery associated with dam removal, and how these processes may affect ecosystem nutrient retention, which we conclude is controlled by physical processes, not biological succession.

DAMS AND DAM REMOVAL: THE STATE OF THE SCIENCE

The introduction and removal of dams has offered an opportunity for scientists to investigate river ecosystems from an interdisciplinary perspective. Seminal studies focused on the damming of rivers highlight the interconnectivity among the diverse fields of hydrology, geomorphology, ecology and biogeochemistry. Early damming studies provided some of the best examples of how hydrologic alterations affect channel form (Leopold et al., 1964). In general, dams are effective at retaining sediments, which release sediment-starved flows to downstream environments. This can cause channel degradation, bed armoring, bank failure, and in some cases, bank accretion (Leopold et al., 1964; Petts, 1984). The understanding of hydrogeomorphic consequences from river regulation supported the exploration of ecological impacts (Petts, 1984; Wootton et al., 1996; Power et al., 1996). Likewise, hydrogeomorphic character in regulated rivers exerts considerable influence over river, coastal and global biogeochemical cycles, largely because of the retentive nature of reservoirs (Humborg et al., 1997; Conley et al., 2000; Stanley and Doyle, 2002; Gergel et al., 2005; Teodoru and Wehrli, 2005). Thus, dams are responsible for hydrogeomorphic, ecological and biogeochemical fragmentation within the world's rivers (Ward and Stanford, 1983).

Similar to damming, dam removal offers another opportunity to further the interdisciplinary exploration of river ecosystems. Because much is unknown about the effects of dam removal, anticipating river responses to such activities is challenging. Unfortunately, using generalizations that emerged from the decades of damming literature is somewhat limited when anticipating ecosystem responses to dam removal. The damming literature is dominated by investigations of large storage reservoirs, while the vast majority of dams removed within the United States have been small, run-of-river structures (Hart et al., 2002; Doyle et al. 2003b). This rift poses a problem because the degree of hydrological alteration caused by dams is associated with structural size and design. Regardless of this disconnect, dam removal is an increasingly popular river management strategy. Dam removal research serves a dual purpose; it improves our ability to manage the world's aquatic resources, while providing the research community with a large scale experimental mechanism. Such ecosystem experiments provide researchers with a chance to investigate the connectivity among hydrological, geomorphic, ecological and biogeochemical mechanisms of ecosystem recovery (for a recent review of dam removal research, see Doyle et al., 2005).

The examination of hydrogeomorphic responses to dam removal was accomplished early among dam removal studies. Upstream channel evolution, via the erosion of former lake bed sediments was shown to follow a predictable pattern in mid-western impoundments exhibiting substantial fine sediment accumulations (Doyle et al., 2003a). Additionally, early dam removal studies documented enhanced sediment export from impoundments responding to hydraulic alteration. In general, a trend has emerged from such studies, dam removal leads to the degradation of formerly impounded channels, and the aggradation of downstream reaches (Doyle et al., 2003a; Stanley et al., 2002; Wohl and Cenderelli, 2000; Lisle et al., 2001; Riggsbee, 2006).

Dam removal studies involving hydrogeomorphic effects provide clear examples of channel and hydraulic responses to dam removal, and such behavior has direct implications for river biogeochemistry (Riggsbee, 2006). Some studies have emerged exploring these connections, and a wide range of responses have been reported. Bushaw-

Netwon et al. (2002) concluded that dam removal did not result in altered downstream water chemistry or increased transport of total suspended solids (TSS). However, other studies have found that dam removal resulted in significant increases in downstream transport of TSS, total phosphorus (TP), total nitrogen (TN), total dissolved nitrogen (TDN) and dissolved organic carbon (DOC) (Doyle et al., 2003a; Ahearn and Dahlgren, 2005; Riggsbee, 2006). Hydrogeomorphic and biogeochemical responses to dam removal seem to reunite historically fragmented reaches along the river continuum by reversing the depositional nature of impoundments. However, a new equilibrium condition among formerly fragmented reaches does not necessarily come free of negative consequences to stream biology.

Based on the brief synthesis of hydrogeomorphic, ecological and biogeochemical effects of damming and dam removal on river ecosystems, both can be considered disturbances to river ecosystems that is, "a relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resource, substrate availability, or the physical environment (White & Pickett, 1985). Both disturbances are caused by distinct changes in system hydraulics that can result in the alteration of various geomorphic, ecological and biogeochemical properties within affected reaches, which can be, and has been, used to justify dam removal for the purposes of river restoration.

DAM REMOVAL AS RIVER RESTORATION

Increasingly, dam removal is used as means of river restoration. Much of the justification for this particular restoration technique is valid; return impoundment hydraulics to a natural, free-flowing character, reestablishing habitat and passage for riverine organisms. Removals have successfully removed migration barriers on the Neuse River, which have historically limited anadromous fish movements (Beasley and Hightower, 2000; Bowman and Hightower, 2001). Following the removal of Woolen Mills Dam, lotic fish assemblages replaced lentic assemblages within formerly impounded habitat (Kanehl et al., 1997). Additionally, macroinvertebrate communities in the Baraboo River, Wisconsin were found to be indistinguishable from free-flowing references one year following dam removal (Stanley et al., 2002). As positive as these results are, it is important to be mindful that the restoration of impoundments back to their historic free-flowing conditions may necessitate the transport of appreciable quantities of nutrient-rich sediments to downstream environments.

As stated earlier, dams are generally effective at retaining fine sediments, which often characterized by nutrient-rich pore water and mineral surfaces (Stanley and Doyle, 2002; Shafroth et al., 2002; Riggsbee, 2006). Specifically, fine sediments, such as those that commonly accumulate within impounded channels, can be significant sources of DOM (McDowell and Wood, 1984; Nelson et al., 1993; Aufdenkampe et al., 2001; Riggsbee, 2006), NH_4 (Triska et al., 1994) and P (Meyer, 1979; Klotz, 1988; Mulholland, 1992). Thus, dam removals resulting in the transport of impoundment materials present a double threat to downstream ecosystems: sedimentation and eutrophication. This view of dam removal as a source of downstream ecosystem disturbance poses important implications for dam removal as a means of river restoration because the unintended downstream consequences of removal activities may nullify restoration benefits.

TEMPORAL AND SPATIAL DIVERGENCE

Because of the longitudinal character of rivers, the disturbance elicited by dam removal, similar to that of damming, must be considered bidirectional. Even though one event, dam removal, acts as the major force of perturbation for both upstream and downstream environments, there are two fundamentally different disturbances initiated by this action. As defined above, disturbances have profound effects on ecosystem structure and resource availability (i.e., habitat, nutrients, light, etc.). Therefore, by altering the physical environment, disturbances initiate secondary succession, which can exert appreciable influence over ecosystem biogeochemical cycling (Odum, 1969; Vitousek and Reiners, 1975). Thus, biogeochemical fluxes (i.e., N) through reaches upstream and downstream of removed dams can be used characterize upstream and downstream ecosystem recoveries from the disturbance of dam removal. For clarification, we define steady state as Vitousek and Reiners (1975) did; ecosystem limiting nutrient inputs equal outputs. Also, we define the upstream ecosystem as the former impoundment, and the downstream ecosystem as the channel, equal in length to that of the former impoundment, located immediately downstream of the former dam site.

Upstream perturbations are constrained in space and time, as controlled by hydraulic alterations that affect stream velocity and channel geometry (Doyle et al. 2003a; Ahearn and Dahlgren, 2005). The duration of the hydraulic alteration is limited to the time required to perform the actual removal of the dam. The temporal extent of upstream ecosystem response to dam removal is variable depending on the ecosystem component of interest (Figure 5.1). Upstream responses to dam removal are the fundamental basis of river restoration strategies, and are initiated immediately following dam removal (hours – days; Riggsbee, 2006), which, over longer timescales (10^0 – 10^1 years; Doyle et al., 2005), will drive geomorphic change, thus producing lotic habitat appropriate for channel and riparian communities. These communities are established over even broader timescales (10^0 – 10^2 years; Stanley et al., 2002; Kanehl et al., 1997; Orr and Stanley, 2006; Figure 5.1). Upstream dam removal disturbance is spatially limited to the original extent of the impoundment, as dictated by the dimensions of the dam. Flows upstream of the former impoundment are assumedly not affected by the removal of the dam.

The alteration of upstream hydraulics has profound effects on reach-scale sediment and biogeochemical fluxes as ecosystems abruptly shift from lentic to lotic conditions. As detailed in the preceding paragraph, this nearly instantaneous disturbance associated with dam removal results in the recovery of various components within former impoundments with considerable temporal heterogeneity (Figure 5.1). As all ecosystem components are directly connected to sediment and biogeochemical exports, it may be cursorily assumed that steady state is only possible once all components have fully recovered from dam removal. However, since dams produce highly retentive impoundments, the physical storage of sediment and associated N, P and DOC pools represent finite sources for downstream transport. Therefore, the sediment and biogeochemical steady states of recovering impoundments should be reached simultaneously once geomorphic reworking is complete (Figure 5.1). Geomorphic reworking is controlled by a system's flow regime, and the quantity of transportable materials stored over the life of the impoundment.

Downstream disturbance patterns are much more complex spatially and temporally than those of reaches located upstream of removed dams. Contrary to upstream disturbance, which is caused by the one time alteration of water residence time within the former impoundment, downstream disturbance is caused by the transport of impounded materials, which occurs repeatedly during impoundment geomorphic reworking. Imported fluxes of sediments and nutrients from recovering impoundments may directly affect stream biology by way of sedimentation and/or eutrophication. Perhaps of more importance to downstream biology are the secondary effects of increased sediment inputs, altered channel morphology and bed texture (Figure 5.2). These changes then influence in-channel nutrient retention by restructuring the substrate supporting indigenous biological communities.

Since the transport of impounded materials is related to system transport capacity, the spatiotemporal variability of downstream perturbations is dependent on the post-removal flow conditions, which determines the frequency and magnitude of impounded material transport. In contrast to environments upstream of removed dams, the spatial extent of such disturbances exhibits considerable variation from spate to spate; expanding and contracting through time, depending on the transport capacity of each flood. Further, the cessation of downstream eutrophication and sedimentation perturbations related to dam removal is dependent on the completion of impoundment geomorphic reworking (Figures 5.1 and 5.2).

DISTURBANCE, BIOLOGICAL SUCCESSION AND GEOMORPHIC REWORKING

Significant portions of the ecological disturbance literature are focused on the biological controls of ecosystem function following physical disturbances. In particular, the ecosystem succession and nutrient retention hypothesis (ESNR) (Odum, 1969; Vitousek and Reiners, 1975) states that the ability of an ecosystem to retain limiting elements (i.e., N and P) increases as the community "matures" (progresses through its successional sere). The seminal work of Vitousek and Reiners (1975) demonstrated that nutrient retention is related to forested watershed succession, following a predictable trajectory toward steady state. They present a figure of this relationship, which shows disturbance (causing secondary succession) results in the immediate and dramatic loss of N from the system. This was followed by an accumulation (high retention) of N because of biomass accrual within the recovering watershed, and a gradual decrease in retention as the ecosystem approaches steady state, the point at which elemental inputs equal outputs (Figure 5.3).

The ESNR hypothesis was originally designed for and tested in terrestrial ecosystems, but it has also been tested and supported within the field of stream ecology (Grimm and Fisher, 1986; Grimm, 1987; Marti et al., 1997). The temporal resolution of the cited studies was short (days to months), and the ESNR hypothesis explained in-channel nutrient retention following the scouring disturbance of flash floods. As previously described, the spatially and temporally divergent disturbances associated with dam removal are more complex than scour associated with flash floods. It is our position, therefore, that the ESNR hypothesis does not adequately explain either upstream or downstream biogeochemical dynamics following dam removal. Moreover, because the natures of upstream and downstream disturbances are different (Figures 5.1 and 5.2), we assert that there are distinctly different patterns of nutrient retention (i.e., trajectories toward steady state) following the disturbance of dam removal for upstream and downstream environments (Figure 5.3).

The ESNR hypothesis is predicated on the assumption that ecosystem biological communities control nutrient retention. However, the physical alteration of river flows control material storage within reservoirs (Petts, 1984), and likewise, the export of materials from impoundments following dam removal is also a physically mediated process (Doyle et al., 2003; Riggsbee, 2006). Therefore, the assumption that ecosystem succession can control the export of materials from recovering impoundments is not appropriate. Impoundments are typically effective at retaining sediment, organic matter and nutrients over timescales of decades to centuries (typical range of impoundment lifespan), and the removal of their dams reverses the relationship impoundments have with their rivers. Examples within the literature demonstrate that sediment accumulations within reservoirs can reach depths of 1-2 meters (Doyle et al., 2003; Riggsbee, 2006), well beyond the maximum rooting depth of early successional plant communities colonizing these sites (10 cm; Riggsbee, 2006). Therefore, the role of biological succession is likely limited on nutrient retention within former impoundments (Riggsbee, 2006). Further, previous research has suggested that the export of impounded materials (i.e., sediments and associated nutrients) is driven by local flow regimes (Ahearn and Dahlgren, 2005; Riggsbee, 2006). Thus, the trajectory toward biogeochemical steady state within former impoundments, and downstream reaches is controlled by physical, not biological, processes associated with geomorphic reworking, resulting in distinctly different trajectories than that proposed by Vistousek and Reiners (1975) (Figure 5.3).

Recall from the earlier explanation of upstream and downstream disturbances following dam removal that upstream disturbances are high-magnitude, one time events, while downstream disturbances are repetitive and of varying magnitudes. This key difference produces different trajectories toward biogeochemical steady state conditions for upstream and downstream reaches (Figure 5.3). Upstream equilibria depend on the exhaustion of supplies from the former impoundment before reach biogeochemical inputs equal outputs. On the other hand, downstream biogeochemical equilibria are dependent on the magnitude of inputs. Since biogeochemical inputs are accompanied by considerable sediment loads that dictate downstream geomorphic character (i.e., channel geometry, bed grain texture; Figure 5.2) downstream ecosystems experience multiple cycles of disturbance and recovery, thus forcing downstream ecosystems to oscillate around biogeochemical equilibria (Figure 5.3).

Different from biogeochemical equilibria, upstream and downstream geomorphic equilibria exhibit a mirror image of one another (Figure 5.4). Simply put, upstream exports exceed inputs as former impoundments degrade during geomorphic reworking.

Downstream environments aggrade in response to the increased sediment inputs, until all impoundment sources associated with the active channel have been excavated. Physical processes control material transport from recovering impoundments, and because impounded sediments are often nutrient-rich. Therefore, geomorphic and biogeochemical equilibria should coincide temporally (Figure 5.5).

CONCLUSIONS

Dam removal, similar to damming, induces bidirectional disturbances, which vary spatially and temporally. Upstream disturbances initiated by dam removal result from the alteration of water residence time, which represent one-time, high-magnitude events. This hydraulic alteration causes the upstream environment to shift from lentic to lotic conditions, which is spatially limited to the upstream extent of the impoundment. Downstream disturbance dynamics are more complex, repetitive and of varying magnitude. This is because disturbances of ecosystems downstream of removed dams are dependent on the export of excess sediments and nutrients from recovering impoundments. The recovery of both upstream and downstream ecosystems is controlled by two factors: local flow regime and the quantity of impounded sediments and nutrients available for transport. Additionally, the trajectories of recovery for both upstream and downstream ecosystems are distinctly different (Figure 5.3), but are intimately linked to channel adjustment processes within the impoundment (Figure 5.5). Such considerations should be kept in mind when utilizing dam removal for the purposes of river restoration.

REFERENCES

Ahearn DS and Dahlgren RA. 2005. Sediment and nutrient dynamics following a low-head dam removal at Murphy Creek, California. *Limnology and Oceanography* 50: 1752-1762.

Aufdenkampe AK, Heges JI, Richey JE, Krusche AV and Llerena CA. 2001. Sorptive fractionation of dissolved organic nitrogen and amino acids onto fine sediments with the Amazon Basin. *Limnology and Oceanography* 46: 1921-1935.

Bowman S and Hightower JE. 2001. American shad and striped bass spawning migration and habitat selection in the Neuse River, North Carolina. Final report to the North Carolina Marine Fisheries Commission, NC.

Bushaw-Newton KL, Hart DD, Pizzuto JE, et al. 2002. An integrative approach towards understanding ecological responses to dam removal: The Manatawny Creek study. *Journal of the American Water Resources Association* 38: 1581-1599.

Conley DJ, Stalnacke P, Pitkanen H and Wilander A. 2000. Transport and retention of dissolved silicate by rivers in Finland and Sweden. *Limnology and Oceanography* 45: 1850-1853.

Doyle MW, Stanley EH, and Harbor JM. 2003a. Channel adjustments following two dam removals in Wisconsin. *Water Resources Research* 36: 1147, doi: 10.1029/2003WR002038.

Doyle MW, Stanley EH, Harbor JM and Grant GE. 2003b. Dam removal in the United States: Emerging Needs for Science and Policy, EOS 84(4): 29-33.

Doyle M.W Stanley EH, Orr CH, Selle AR, and Harbor JM (2005). Response of stream ecosystems to dam removal: Lessons from the heartland. *Geomorphology* 71: 227-244.

Gergel SE, Carpenter SR and Stanley EH. 2005. Do dams and levees impact nitrogen cycling? Simulating the effects of alterations on floodplain denitrification. *Global Change Biology* 11: 1352-1367.

Graf WL, Stromberg J and Valentine B. 2002. Rivers, dams and willow flycatchers: a summary of their science and policy connections. *Geomorphology* 47: 169-188.

Grimm NB and Fisher SG. 1986. Nitrogen limitation in a Sonoran Desert stream. *Journal of North American Benthological Society* 5: 2-15.

Grimm NB. 1987. Nitrogen dynamics during succession in a desert stream. *Ecology* 68: 1157-1170.

Hart DD, Johnson TE, Bushaw-Newton KL, Horwitz RJ, Bednarek AT, Charles DF, Kreeger DA and Velinsky DJ. 2002. Dam removal: challenges and opportunities for ecological research and river restoration. *Bioscience* 52: 669-681.

Humborg C, Ittekkot V, Cociasu A and VonBodungen B. 1997. Effect of Danube River Dam on Black Sea biogeochemistry and ecosystem structure. *Nature* 386: 385-388.

Kanehl PK, Lyons J and Nelson JE. 1997. Changes in the habitat and fish community of the Milwaukee River, Wisconsin, following removal of the Woolen Mills Dam. *North American Journal of Fisheries Management* 17: 387-400.

Klotz RL. 1988. Sediment control on soluble reactive phosphorus in Hoxie Gorge Creek, New York. *Canadian Journal of Fisheries and Aquatic Sciences* 45: 2026-2034.

Leopold LB, Wolman MG and Miller JP. 1964. *Fluvial Processes in Geomorphology*. Dover Publications. New York. pp 522.

Lisle, TE, Cui Y, Parker G, Pizzuto JE, and Dodd AM. 2001. The dominance of dispersion in the evolution of bed material waves in gravel-bed rivers. Earth Surface Processes and Landforms 26: 1409-1422.

Marti E, Grimm NB and Fisher SG. 1997. Pre- and post-flood retention efficiency of nitrogen in a Sonoran Desert stream. *Journal of North American Benthological Society* 16: 805-819.

Orr CH, and Stanley EH. 2006. Vegetation development and restoration potential of drained reservoirs following dam removal in Wisconsin. *Rivers Research and Applications* 22: 281-295.

McDowell WH and Wood T. 1984. Podzolization: Soil processes control dissolved organic carbon concentrations in stream water. *Soil Science* 137: 23-32.

Meyer J. 1979. The role of sediments and bryophytes in phosphorus dynamics in a headwater stream ecosystem. *Limnology and Oceanography* 24: 365-375.

Mulholland PJ. 1992. Regulation of nutrient concentrations in a temperate forest stream: Roles of upland, riparian and instream processes. *Limnology and Oceanography* 37: 1512-1526.

Odum EP. 1969. The strategy of ecosystem development. *Science* 164: 262-270.

Petts GE. 1984. *Impounded Rivers: Perspectives for Ecological Management*. John Wiley and Sons, Chichester, UK. pp. 326.

Power ME, Dietrich WE and Finaly JC. 1996. Dams and downstream aquatic biodiversity: potential food web consequences of hydrologic and geomorphic change. *Environmental Management* 20: 887-895.

Shafroth PB, Friedman JM, Auble GT, Scott ML and Braatne JH. 2002. Potential response of riparian vegetation to dam removal. *Bioscience* 52: 703-712.

Stanley EH, Luebke MA, Doyle MW and Marshall DW. 2002. Short-term changes in channel form and macroinvertebrate communities following low-head dam removal. *Journal of the North American Benthological Society* 21: 172-187.

Stanley EH and Doyle MW. 2002. A geomorphic perspective on nutrient retention following dam removal. *Bioscience* 52: 693-701.

Riggsbee JA. 2006. Spatial and temporal heterogeneity of impoundment nutrient and sediment fluxes following dam removal. PhD. dissertation. University of North Carolina at Chapel Hill.

Teodoru C and Wehrli B. 2005. Retention of sediments and nutrients in the Iron Gate I Reservoir on the Danube River. *Biogeochemistry* 76: 539-565.

Triska FJ, Jackman AJ, Duff JH and Avanzino RJ. 1994. Ammonium sorption to channel and riparian sediments: A transient storage pool for dissolved inorganic nitrogen. *Biogeochemistry* 26: 67-83.

Vitousek PM and Reiners WA. 1975. Ecosystem succession and nutrient retention: a hypothesis. *Bioscience* 25: 376-381.

Ward JV and Stanford JA. The serial discontinuity concept of lotic ecosystems. pp. 29-42 in TD Fontaine and SM Bartell (eds) *Dynamics of Lotic Ecosystems*. Ann Arbor Science. Ann Arbor. pp. 494.

White PS and Pickett STA. 1985. Natural disturbance and patch dynamics: an introduction. Pp 3-13 in PS White and STA Pickett (Eds), *The Ecology of Natural Disturbance and Patch Dynamics*. Academic Press, New York.

Wohl EE, and Cenderelli DA. 2000. Sediment deposition and transport patterns following a reservoir sediment release. *Water Resources Research* 36: 319-333.

Wootton JT, Parker MS and Power ME. 1996. Effects of disturbance on river food webs. *Science* 273: 1558-1561.

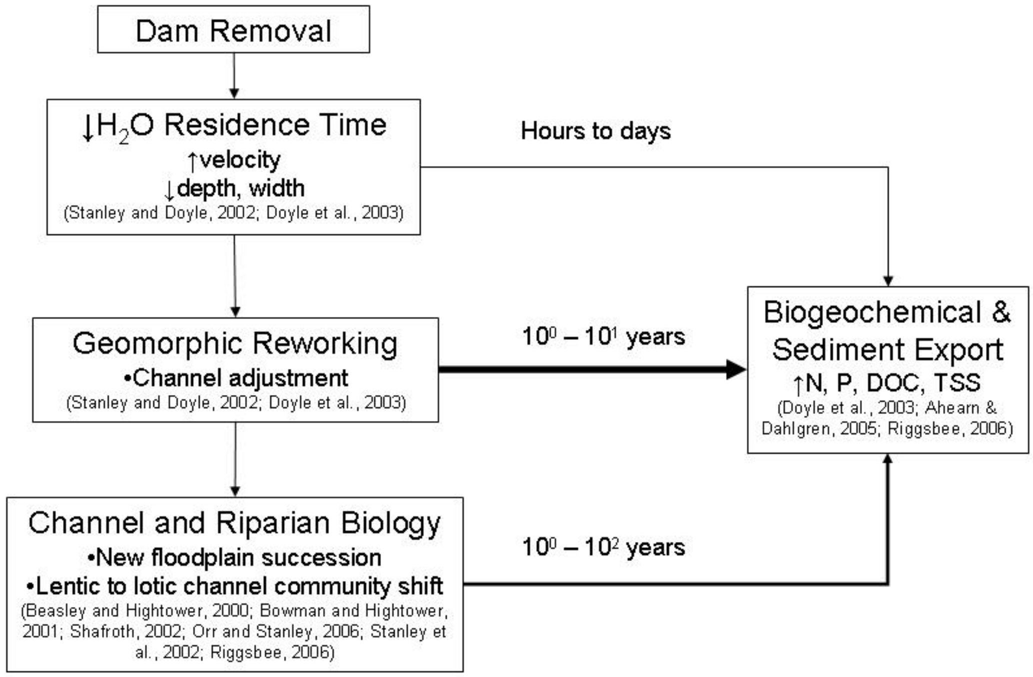


Figure 5.1: Upstream disturbances following dam removal. The gage of arrows denotes the importance of the pathway to downstream biogeochemical and sediment fluxes.

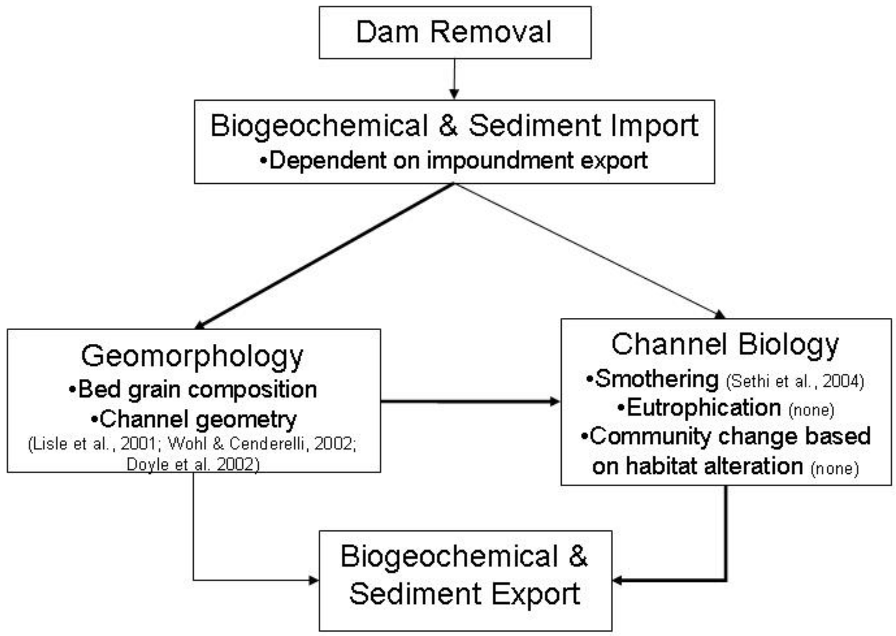


Figure 5.2: Downstream disturbances following dam removal. The gage of arrows denotes the importance to altering downstream biogeochemical and sediment fluxes.

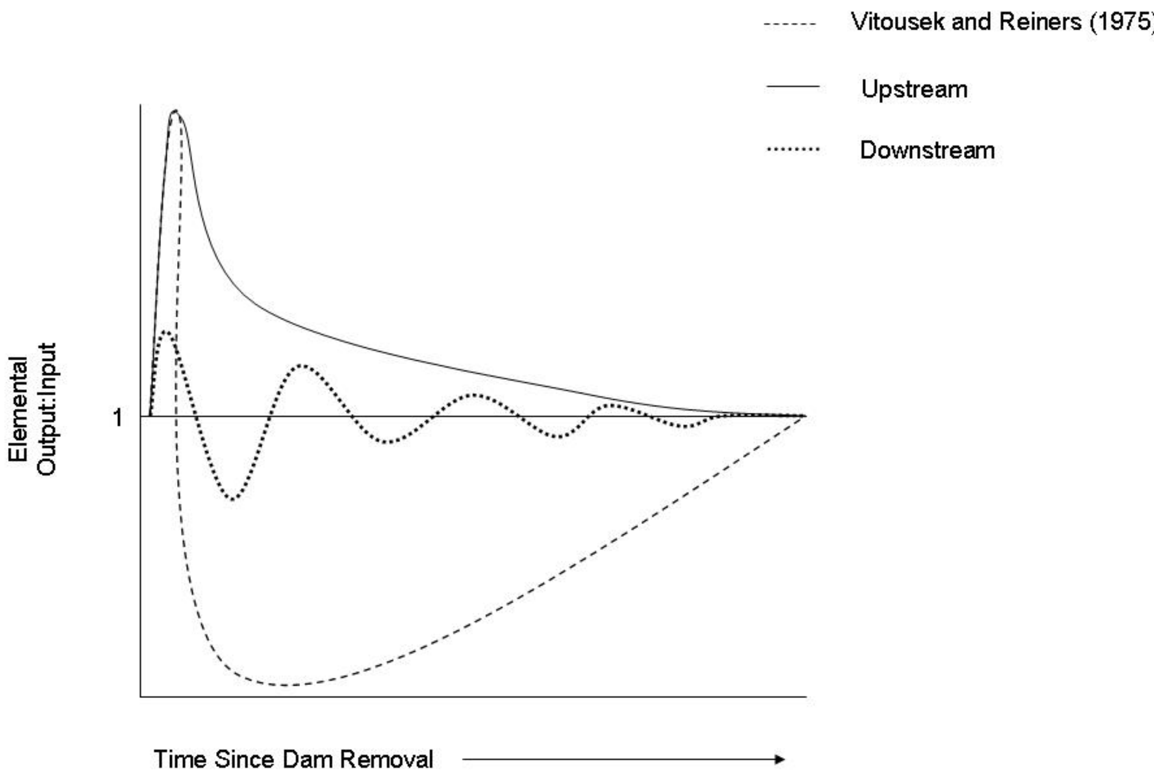


Figure 5.3: Trajectories toward biogeochemical steady states for upstream and downstream reaches following dam removal. Modified from Vitousek and Reiners (1975) to represent ecosystem recovery of upstream and downstream ecosystems following dam removal.

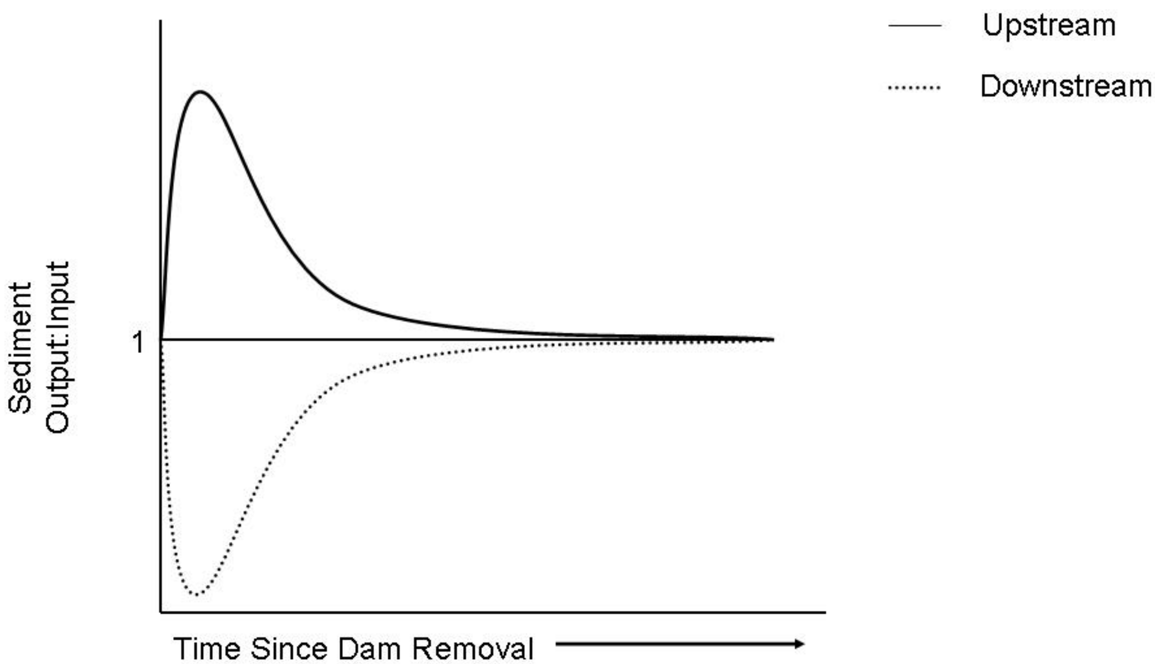


Figure 5.4: Trajectories towards geomorphic steady state for upstream and downstream reaches following dam removal. Upstream sediment budgets show a degrading system, producing aggrading conditions downstream of removed dams.

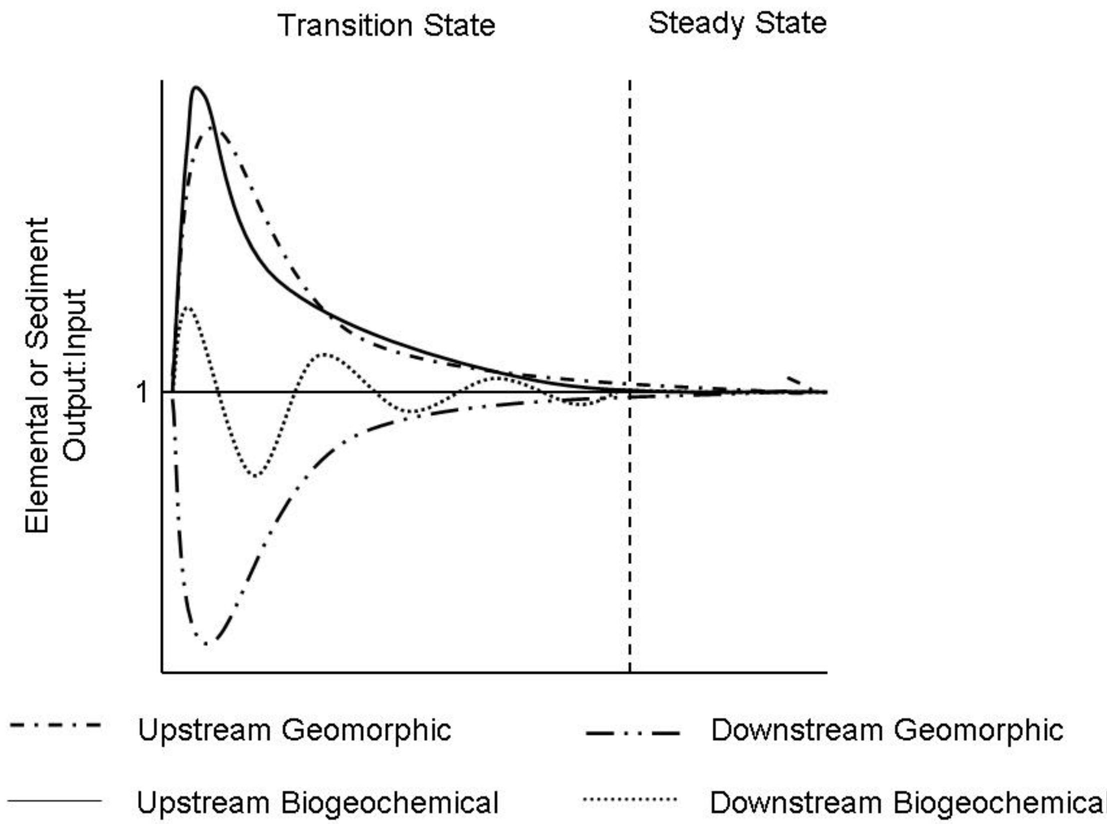


Figure 5.5: Hypothetical trajectories toward biogeochemical and geomorphic steady states in upstream and downstream reaches following dam removal. Note that sediment and elemental losses are in step. This is because geomorphic reworking of the former impoundment controls sediment and elemental fluxes.