Patapsco River Dam Removal Study: Assessing Changes in American Eel Distribution and Aquatic Communities

Final Report

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Foreword

This report describes the results of efforts by the Maryland Department of Natural Resources Monitoring and Non-Tidal Assessment Division to monitor and assess ecological changes in the Patapsco River associated with the removal of Simkins and Union Dams. There are six aspects of the Patapsco River's ecosystem that were examined: anadromous fish, American eels, resident fish, benthic macroinvertebrates, freshwater mussels, and water quality. Each of these aspects is described in a separate chapter that includes a brief review of literature followed by a discussion of results from Patapsco monitoring. Prior to these chapters is an Introduction that describes the river and its dams. There is an Ecological Monitoring Methods chapter that includes sampling locations and general sampling methods. The last chapter prior to the six ecological condition chapters is a Physical Changes chapter which briefly outlines physical changes in the river resulting from dam removals. The final chapter of the report provides Conclusions and Recommendations based on the results of monitoring and assessments. The conclusions and recommendations from monitoring associated with the removal of Union and Simkins Dams are intended to guide continued Patapsco River restoration efforts and other attempts to restore river connectivity in Maryland and elsewhere.

Executive Summary

The Maryland Biological Stream Survey (MBSS) within the Department of Natural Resources (DNR), in collaboration with American Rivers, NOAA, and the DNR Fisheries Service, performed biological monitoring in the Lower North Branch Patapsco River as part of the removal of Simkins and Union dams. The goals of this project were to determine the impacts of the removal of Simkins and Union dams on American eel (*Anguilla rostrata*) and anadromous fish distributions as well as on water chemistry, resident fish, benthic macroinvertebrate, and freshwater mussel communities. The main objectives of this project were to:

1) Determine whether American eels will utilize the river and tributaries to the river upstream of Simkins and Union dams after removal.
2) Quantify changes in water chemistry, resident fish, benthic macroinvertebrate and freshwater mussel communities of the river both upstream and downstream of the dams following removal.
3) Determine the presence and extent of migrating anadromous fishes in the vicinity of Bloede Dam.

To meet these objectives, 26 sites were sampled for anadromous fish, American eels, resident fish, benthic macroinvertebrates, and freshwater mussels for two years before (2009-2010) and two years after (2011-2012) the removal of Simkins Dam in December 2010 (Union Dam was removed in February 2010). Additionally, DNR's Core/Trend long-term water quality data, as well as detailed observations of the physical habitat, were examined to look for changes in water chemistry and physical habitat/bottom condition following the dam removals. Results were analyzed by comparing the pre- and post- dam removal data to find changes that could be attributed to the dam removals, in some cases using data from unaffected control sites (in the Patapsco River or elsewhere) for comparison.

Although we examine ecological changes associated with the removal of both dams, the main focus of this report is on the removal of Simkins Dam – found to have substantially more influence on the ecology of the Patapsco River in comparison to Union Dam. Union Dam, breached in 1972 by Hurricane Agnes, was not a complete blockage, and, as such, trapped less sediment than Simkins Dam. As a result, ecological changes associated with its removal were less pronounced.

The report focuses on eight key topics: physical changes, water quality, anadromous fishes, American eels, resident fish, benthic macroinvertebrates, freshwater mussels, and conclusions and recommendations.

Physical Changes

The dominant bottom substrate, as well as habitat for fish and benthic macroinvertebrates, shifted following the removal of Simkins Dam. Before the dam removal, sand was the dominant substrate at monitoring sites directly upstream of the dam, while cobble and gravel dominated below. After Simkins Dam was removed, the substrate composition changed; cobble and gravel became prevalent upstream of the dam while sand dominated downstream. Fish and benthic macroinvertebrate habitat quality exhibited a similar temporal pattern, decreasing downstream of Simkins Dam and increasing upstream following dam removal. These changes might stem from the movement of sand from the former impoundment behind Simkins Dam, exposing and then subsequently burying more desirable (cobble and gravel) habitat as the sand moved downstream. We expect habitat quality throughout the river to improve over time as the finer-grained sediments continue to move out of the non-tidal Patapsco River.

Water Quality

The impoundments created by Union and Simkins dams were likely too small to have much effect on water quality and nutrient processing. Despite this, differences in several water quality parameters between downstream and upstream monitoring sites on the river increased measurably during the post dam removal period, including phaeophytin a, total suspended solids, particulate nitrogen, particulate carbon, particulate phosphorus, and turbidity. The majority of these differences were seen during sampling that coincided with large storms in the fall of 2011. These storms may have mobilized trapped sediment from behind the dams, presumably releasing nutrients, carbon, and algae. This effect on water quality will likely diminish over time as the sediment and related material eventually leave the river. Monthly water quality sampling will continue to document the long-term water quality influence of dam removal in the Patapsco River.

Anadromous Fishes

Bloede Dam's function as a migration barrier was of particular interest in this study. Bloede Dam is the first major obstacle encountered by fish moving upstream from the Chesapeake Bay and appears to block access to the entire river upstream for the majority of anadromous species. While the dam has a fish ladder, its effectiveness was largely unknown. Anadromous fish were sampled using electrofishing equipment and a fyke net at four sites downstream from Bloede Dam and one site upstream during spring 2011-2012. Five species of anadromous fish were collected at the four sites below the dam, but only one species was collected above the dam. Both abundance and diversity of anadromous species were reduced in 2012 compared to 2011. No anadromous species were observed using the fish ladder during either year, but several resident fish species used the fish ladder in 2012. We could not determine whether the apparent absence of anadromous fishes above Bloede Dam in 2012 resulted from an inability for these species to use the fish ladder or from low densities of pre-spawning adults that were observed throughout the region in that year.

American Eels

Sampling for American eels (*Anguilla rostrata*) was conducted at 21 sites during summer 2009-2012 in an attempt to fully assess the changes in eel size, distribution, and abundance after the two dams were removed. American eels were present at all sites. Overall, eel abundance decreased with increasing distance upstream while average eel size increased. In the two years following the removal of Union and Simkins dams, eel abundance decreased directly below Simkins Dam, and the average size of eels decreased at sites upstream of the dam. It is not known at this time whether these changes are due to habitat alteration related to disturbance from the dam removal project or to changes in eel distribution following the removal of a migration barrier.

Resident Fish

The removal of Simkins Dam restored connectivity within a larger portion of the Patapsco River, allowing for fish dispersal between upstream and downstream reaches. Consequentially, fish assemblages at sites adjacent to Simkins Dam became more similar in species composition, with the greatest increase in similarity at sites previously separated by the dam. Although the long-term effects of dam removal are generally viewed as positive, dam removal is not without short-term, less positive consequences. Downstream sedimentation affected benthic riffle fishes that utilize clean, coarse substrate as refuge and for feeding. Other species sensitive to disturbance also declined following removal of Simkins Dam as released sand filled in their preferred habitats. Because of this, downstream sites had a higher proportion of species loss - specifically benthic species - than other sites sampled in the river. Additionally, we documented a decline in the young-of-year size class of smallmouth bass at downstream sites following dam removal. In general, Patapsco River fish assemblages responded to dam removals similarly to what has been documented in previous dam removal studies in other riverine systems. We expect the fish assemblages will recover in time as the remaining sediment behind the former Simkins Dam is transported out of the study area, geomorphic conditions stabilize, and habitat quality improves.

Benthic Macroinvertebrates

Changes in the macroinvertebrate community in the Patapsco River appear to be associated with shifts in dominant habitat. The most notable changes occurred at the four sites upstream and downstream of Simkins Dam where there were dramatic shifts in dominant substrate type. The percent of EPT (Ephemeroptera, Plecoptera, and Trichopera) taxa generally increased while the percent of burrowing taxa decreased at sites where sand was replaced by cobble and gravel substrates. Overall, lotic-erosional taxa became more abundant than lentic taxa at the formerly impounded sites after Simkins Dam was removed. The highest increases in benthic macroinvertebrate diversity following the dam removal were recorded primarily at sites upstream of Simkins Dam. The habitat at these sites either shifted from sand to cobble/gravel after the removal of dams or was dominated by cobble/gravel throughout the study. Few studies have examined the long-term response of macroinvertebrate communities to dam removal. Continual monitoring of the macroinvertebrate communities at these Patapsco River sites is needed to determine if the communities throughout the river will eventually be comprised primarily of riverine taxa.

Freshwater Mussels

The eastern elliptio, *Elliptio complanata*, tolerates a range of habitats in Piedmont streams of Maryland. This mussel species appears to be sensitive to degraded water quality parameters typically associated with urbanization, and has been extirpated from the Patapsco River for at least 50 years, probably longer. Eastern elliptios were undoubtedly present at one time in the Patpasco River, given the historical and archaeological shell records from the region, and other freshwater mussels (i.e., *Alasmidonta undulata*) currently found in the Patapsco River. In parts of the river, current densities of American eels, the primary host-fish for *E. complanata*, are comparable to eel densities in other streams in Maryland where *E. complanata* is present. Even so, American eel densities in the river decrease upstream, to a point where they are apparently below the level needed to support *E. complanata* recruitment. Even if dam removal efforts increase American eel host densities, *E. complanata* are unlikely to recolonize the Patapsco River from nearby populations. Hence, active reintroduction may be the only viable option for the restoration of this freshwater mussel species.

Conclusions

The conclusions based on our work in the Patapsco River stem from only two years of pre- and two years of post-dam removal data and observations, and should be considered preliminary. Although additional post-dam removal monitoring is needed, we have already learned a great deal about the short-term ecological response of the Patapsco River to dam removal. Based on this knowledge, we offer the following:

1. Ecological monitoring should continue for at least four more years to document the long-term ecological response of Simkins and Union Dam removals. Major ecological changes in the Patapsco River are still in progress and will likely take several years to reach a new dynamic equilibrium. Documenting changes as they occur is the best way to demonstrate the benefits of dam removal. Lessons learned from monitoring in the future will inform decisions pertaining to future fish passage and prospective dam removal projects. All the data collected so far serve as useful indicators of stream condition. These indicators should continue to be used in future years.

2. Bloede Dam is the downstream-most blockage on the Patapsco River and the fish ladder there appears to be largely ineffective at passing anadromous fish. Removing Bloede Dam would provide unimpeded passage for anadromous fish, improve habitat for resident fish and other riverine species, and allow sediment trapped behind it to move downstream and out of the non-tidal Patapsco River. The data described in this report will provide four years of baseline data for examining the ecological benefits of Bloede Dam's eventual removal.

3. If Bloede Dam is removed, Daniels Dam will be the last remaining barrier to fish movement in the mainstem Patapsco River. In lieu of removing this dam, the efficacy of the fish ladder for passing migratory fishes could be examined.

4. Surveys of current freshwater mussel distribution, identification of stream reaches with suitable American eel (freshwater mussel host) habitat, and freshwater mussel habitat suitability and survival studies are needed if freshwater mussel re-introduction is a desired goal for the Patapsco River.

5. The sand and gravel released from upstream of Union and Simkins dams (and other sources) have continually moved downstream, making their way into the tidal portion of the river, potentially degrading habitat for resident and migratory species. The rate and pattern of movement of this material over time could be an important controlling factor for restoring abundant anadromous fish runs up the Patapsco River.

Chapter 1: Introduction

Until recently, the Patapsco River, as it flows almost 35 miles through Patapsco Valley State Park to Baltimore Harbor and the Chesapeake Bay, has been home to four dams. These dams - Daniels, Union, Simkins, and Bloede- were used in the late 19th and early 20th centuries to power flour and textile mills as well as to generate hydroelectricity. However, these dams have become obsolete after industry moved on. Much of the river valley around the dams, within what is now the Patapsco Valley State Park, returned to a more natural state when the industries left. The dams remained, blocking passage for migratory fish, changing habitats, and creating hazards for swimmers and boaters.

Initial attempts to improve passage using fish ladders met with limited success. Anadromous fish such as blueback herring and hickory shad have been documented to use the river downstream of the dams during spring spawning runs. However, despite substantial cost and effort to maintain functionality, very few individuals of these species have been known to pass through the ladder on the downstream-most (Bloede) dam.

Beginning in 2009, American Rivers, Friends of the Patapsco Valley State Park, the Maryland Department of Natural Resources (DNR), and the National Oceanic and Atmospheric Administration (NOAA) teamed up to restore connectivity in this river by removing these dams. Work to remove Union Dam was completed in spring 2010, while removal of the Simkins Dam began later that same year in November 2010.

The removal of Simkins Dam was conducted using a passive sediment release, i.e., without first removing the sand and gravel from behind it. The potential existed for this sand and gravel to cover cobble and boulder habitats and fill in pools, thus influencing habitat for fish and insects downstream from the dam. Alternatively, the Union Dam removal was completed using an active sediment management approach where sediment behind the dam was removed prior to removing the dam. This active approach to sediment management also included recreating the natural channel dimensions in the process of removing material from behind the dam. Additionally, extra effort was made to prevent sediment from being released downstream while working in the river to remove the dam. Union Dam was also not a major barrier to fish because it was breached by Hurricane Agnes in 1972; however, during higher flows, fish passage was likely reduced due to velocity barriers and high turbulence found in the channel.

Simkins Dam (at 2.4 meters high and 45.7 meter long) prior to its removal, stored approximately 80,000 cubic yards of primarily sand and gravel. Unlike the Union Dam removal, the channel was not reconstructed manually. Instead the channel re-formed its shape through the natural sediment transport process. Despite the presence of a fish ladder, Simkins Dam was a complete barrier to the movement of most fish species. However, most migratory species (with the exception of American eel and sea lamprey), could not reach Simkins Dam due to the presence of Bloede Dam downstream.

Two additional dams remain on the Patapsco River – Daniels and Bloede Dams. Daniels Dam (at 8.2 meters high) is upstream of Bloede, Simkins, and Union dams and currently has a Denil fish ladder installed. Bloede Dam is currently the downstream-most dam on the Patapsco River. It is approximately 10-m-high and stores approximately 70,000 cubic yards of primarily sand and gravel. A Denil fish ladder was built on Bloede Dam in 1992. This fish ladder was shown to have a limited ability to facilitate upstream migrations of certain anadromous fishes soon after it was constructed. This fish ladder annually sustains substantial damage, leaving questions about its ability to effectively pass migratory fish. A feasibility study investigating the removal of the Bloede Dam was finalized in July 2012 with dam removal as a preferred alternative to provide fish passage, improve public safety, and reduce the financial burden on the State for reoccurring repairs and maintenance on the dam structure. With Bloede and Daniels Dams still in place, restoring ecological connectivity to the Patapsco River remains incomplete. However, the removal of Simkins and Union Dams provides an opportunity to examine the potential ecological advantages and disadvantages of dam removal. Since anadromous fish movements are still largely blocked from downstream and the removal of Simkins Dam was done using passive sediment management, we focused our ecological monitoring on the potential influence of sediment movement along with improved connectivity achieved by removing Union and Simkins Dams.

Chapter 2: Ecological Monitoring Methods

Ecological monitoring began in 2009 (two years prior to the removal of Simkins Dam). Standard sampling protocols at targeted sampling locations were used to explore potential changes in each aspect of monitoring. The majority of data were collected from 26 sites located on the mainstem and select tributaries, from the tidal portion of the river to upstream of Daniels Dam (Appendix 2.1: Fig. 2.2)). Each site was identified using a unique code. A summary of the sampling methods for each aspect of ecological monitoring and of sites sampled are described below. More detailed methods are described in Ciccotto et al. (2009) (Appendix 2.2).

Benthic macroinvertebrates

Benthic macroinvertebrate data were collected from 20 sites during 2009 and 21 sites (adding one site upstream from Daniels Dam) from 2010 to 2012 (Appendix 2.1: Fig. 2.3-2.7, Table 1). Maryland Biological Stream Survey (MBSS) sampling methods (Stranko et al. 2007) were used to collect benthic macroinvertebrate data. In brief, this method collects benthic macroinvertebrates from 20, 0.3 m^2 sub-samples of proportionally available optimal habitat using a 540 µm mesh D-shaped net. The 20 sub-samples were combined into one sample and sent to a laboratory where a minimum of 100 organisms were randomly selected and identified to either genus or the lowest practical taxonomic level. Typically, the MBSS sampling method requires that sampling take place only during March or April. However, sampling for this project was conducted during August in 2009 and during August and March in 2011 and 2012. In 2010, sampling took place only during March. The summer sampling was conducted in 2009 because the monitoring project did not begin early enough to allow spring (March or April) sampling. In 2011 and 2012, sampling was conducted in spring and summer so that comparable data were available from both periods during the pre- (2009 and 2010) and post- (2011 and 2012) dam removal periods.

Resident fish

Resident fish data were collected during the summer (June – September) from 11 of the 21 sites that were sampled for benthic macroinvertebrates. Ten sites were sampled during 2009 and 11 sites (adding the site upstream from Daniels Dam) were sampled from 2010 to 2012 (Appendix 2.1: Fig. 2.3-2.6, Table 1). MBSS protocols (Stranko et al. 2007) were used to collect fish data from these sites. In brief, MBSS sampling involved electrofishing a 75-m-long section of stream that was blocked off on the up and downstream ends with 6 mm mesh nets. All fishes collected from two electrofishing passes were weighed in aggregate, identified to species, counted, and released. The total lengths (mm) of all smallmouth and largemouth bass collected from the Patapsco River were also recorded, as is consistent with MBSS protocols of measuring the lengths of game fish.

American eels

Due to the importance of American eels to the Patapsco dam removal project, special emphasis was put on collecting data on this species. American eels were collected at all 21 sites sampled for benthic macroinvertebrates (Appendix 2.1: Fig. 2.3-2.7, Table 1). At the 11 sites sampled for resident fish, eels were counted and weighed in aggregate, separate from the other fishes, and eel abundance (number of eels collected per hour) was recorded. Additionally, eels were counted during a minimum of 600 seconds of electrofishing effort at the remaining 10 sites where benthic macroinvertebrate data had been collected but resident fish data were not. The number of eels collected at these 10 sites was recorded and standardized using the number collected per hour of sampling effort to provide data comparable to the data collected at the resident fish sampling sites.

Anadromous Fish

A total of five sites were sampled for anadromous fish approximately once per week in the spring of 2011 and 2012, during the spawning migration period (Appendix 2.1: Fig. 2.6-2.7, Table 1). In our study, this period spanned March to May. During both years, sampling was first conducted using boat electrofishing at the three downstream most sites to determine if anadromous fish had entered the river. Once anadromous fishes were collected at these sites, electrofishing continued at the fourth site directly downstream of Bloede Dam to determine whether those species were able to successfully migrate to the dam. Once anadromous fish were seen below Bloede Dam, the fifth site immediately upstream of the dam was sampled with a fyke net placed over the exit of the fish ladder. The net was used to collect any fish that were using the ladder to bypass Bloede Dam. This net was deployed for the entire time anadromous species were being collected by electrofishing at the site downstream of the dam. The duration in hours of each net set was recorded to allow for fish per hour calculations.

Freshwater Mussels

Freshwater mussel data (i.e., species presence) were primarily obtained by conducting an informal visual survey at each of the 11 sites that were also sampled for resident fish. Incidental observations of mussels were also recorded while sampling at the 10 sites sampled only for benthic macroinvertebrates and American eel abundance. When a mussel was encountered, it was identified to species and its condition (live or dead) was recorded. If a live mussel was found it was immediately returned to its approximate capture location. Valves from dead mussels were retained as vouchers. Additional mussel data were advantageously obtained because a rare species, Triangle floater, was known from the resident fish monitoring site downstream of Daniels Dam (Site 510). In 2009, a constrained area, timed searches snorkeling method (Strayer and Smith 2003) was used to assess the relative abundance (number of mussels per hour) at this site. In 2010, a two-phase survey (Strayer and Smith 2003) was used in an attempt to better understand the distributional extent and population size of mussels in the Patapsco River. The first phase involved timed snorkel surveys to determine the relative abundance and extent of mussels from Union to Daniels dams and from three river kilometers (rkm) downstream to the base of Bloede Dam. Site 510 was then re-sampled using a quantitative survey (excavation) to provide an estimate of the density and number of mussels in relation to their relative abundance at this site.

Physical Habitat

The quality of physical habitat available to fish and benthic macroinvertebrates was rated based on standard MBSS visual assessment protocols (Stranko et al. 2007). These ratings were used to track changes in habitat quality over time as the dams were removed and sediment from behind the dams was displaced. The physical habitat ratings were conducted at the same 11 sites where benthic macroinvertebrate, resident fish, American eel, and freshwater mussel data were collected (Appendix 2.1: Fig. 2.3-2.6, Table 1).

Water Quality

Data to investigate potential water quality changes in the Patapsco River following dam removal were provided by the Core/Trend Program conducted by the Maryland Department of Natural Resources. Two Core/Trend stations were monitored in the Patapsco River (Appendix 2.1: Fig. 2.9, Table 1). One of these stations is located downstream of Bloede Dam and the other station is located between Union Dam and Daniels Dam. The data collected at these sites consist of grab samples of water taken once each month. These samples are analyzed for a large number of chemical parameters including dissolved oxygen, specific conductance, chlorophyll a, ammonium, nitrate + nitrite, pH, phaeophytin a, phosphate, total suspended solids, particulate nitrogen, particulate carbon, particulate phosphorus, total dissolved nitrogen, total dissolved phosphorus, turbidity, and temperature. Data are available for most variables beginning in 1986, but we only used data from April 2005 through December 2011 (the most recently available data). Data from April 2005 to December 2010 were considered representative of pre-dam removal conditions and January to December 2011 represented a first look at post dam removal conditions.

Analyses

Each chapter presents analyses of ecological monitoring data that were conducted independently. The most appropriate analyses to test the hypotheses addressed in each chapter were used. In most chapters, the differences in values before and after dam removal were used to represent changes in ecological condition. These data are displayed in several cases using a graph showing the differences in site values for each site graphed by river kilometer (Fig. 2.1). These graphs also show the locations of existing and removed dams and often illustrate changes in river bottom substrate (e.g., cobble and gravel or sand dominated river bottom habitat). This information is useful to support conclusions about the influence of blockage removal and sediment dynamics in the Patapsco River as it pertains to changes in ecological conditions.

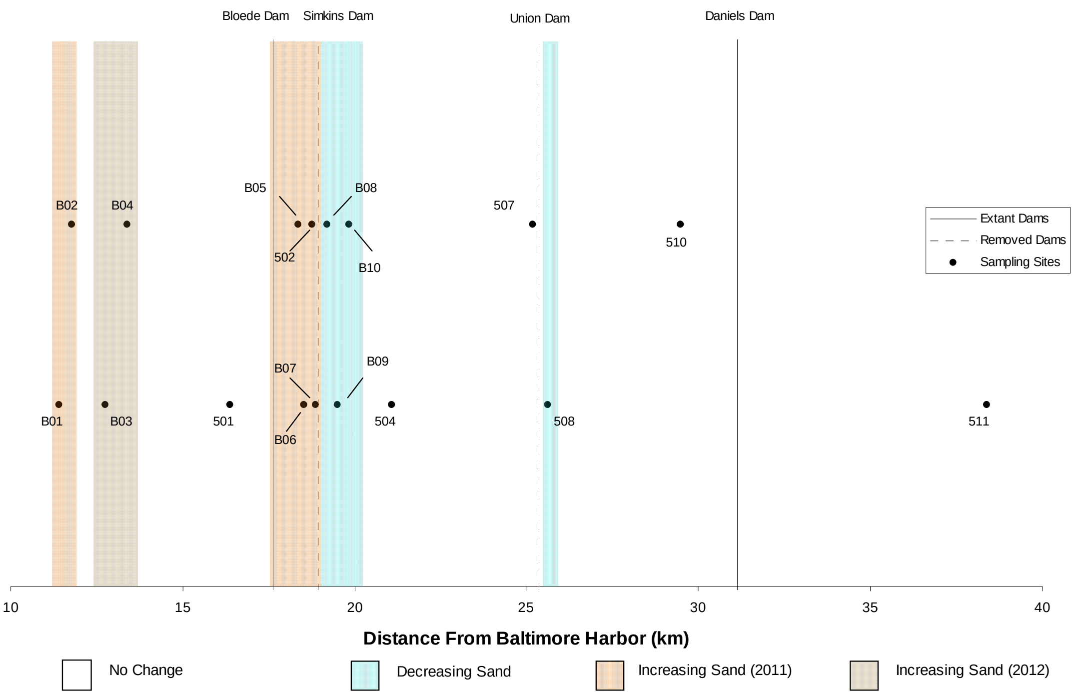

Figure 2.1: An example showing the way sites are displayed graphically in the ecological monitoring chapters. Shaded areas indicate areas of eroded or deposited (agraded) sediment after dam removal.

Literature Cited

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Appendix 2.1: Maryland Biological Stream Survey, Patapsco River Dam Removal Sampling Manual

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Purpose of Document

This document was prepared to provide written standard operating procedures for assessing the impacts of the removal of Simkins and Union dams in the Patapsco River. Due to the unique goals and timeframe of this project, Maryland Biological Stream Survey (MBSS) protocols have been adapted to answer a number of the proposed questions. For details on exact MBSS protocol procedures, please see the "Maryland Biological Stream Survey Sampling Manual." The purpose of this document is not to reiterate MBSS protocols, but rather 1) list what data will be collected using MBSS protocols and 2) describe the adapted methods used for portions of this project. It is imperative that the protocols used for every aspect of the MBSS be provided to guide progress throughout the dam removal monitoring and to ensure that the goals and objectives of the project are met. Therefore all crew members will have gone through MBSS training and be provided with copies of the MBSS manual during field sampling. These written protocols also provide information to anyone attempting to duplicate procedures used by the MBSS or to ensure comparability of data and results generated by the MBSS.

Patapsco Dam Removal Project Goal and Objectives

The goals of this assessment are to ascertain the impacts of the removal of Union, Simkins and Bloede dams on American eel and other diadromous fish distributions as well as benthic macroinvertebrate communities. The three main objectives of this project include:

1. Determine if American eels will utilize the river and tributaries upstream of Simkins and Union dams after their removal.
2. Quantify changes in the river following dam removal on fish and benthic macroinvertebrate communities upstream and downstream of dam locations.
3. Determine the presence and distribution of diadromous fishes in the Patapsco River.

Field Sampling

Overview

To assess the impacts of dam removal on the Patapsco River, MBSS sampling will take place during two index periods, spring and summer. The Spring Index Period extends from 1 March to 30 April, and the Summer Index Period extends from 1 June to 30 September each year. Four primary activities are conducted during the Spring Index Period: benthic macroinvertebrate, water chemistry for laboratory analysis, select physical habitat variable sampling, and vernal pool searches. During the Summer Index Period, seven primary activities are conducted: benthic macroinvertebrate, fish, reptile and amphibian, crayfish, invasive plant, and select physical habitat variable sampling.

Each site consists of the watered portion of the stream and an area 50 meters perpendicular to the stream and is 75 m in length. The 0m, 25m, 50m, and 75m portions of the site (beginning with 0m at the downstream-most end of the site) are to be flagged. The midpoint should also be flagged for the photodocumentation protocols. This project consists of 11 quantitative fish and benthic macroinvertebrate sites that follow standard MBSS protocols (herein referred to as "Fish Sites") and 10 benthic macroinvertebrate only sites (herein referred to as "Benthic Macroinvertebrate Sites").

1. Fish Sites. Full MBSS protocols are conducted during the Spring and Summer Index Periods (Additional benthic macroinvertebrate samples are taken during the summer visit following MBSS Spring benthic macroinvertebrate protocols).
2. Benthic Macroinvertebrate Sites. Full MBSS protocols are conducted during the Spring Index Period (see most recent version of "The Maryland Biological Stream Survey Sampling Manual"). The Summer Index Period is modified for these sites- benthic macroinvertebrate, select summer physical habitat data, and qualitative fish sampling are to be conducted.

Spring Index Period

At both Fish and Benthic Macroinvertebrate Sites, standard MBSS sampling methods from the Spring Index Period will be conducted during the index period, including:

1. Benthic macroinvertebrates
2. Placing temperature loggers
3. Photodocumentation
4. Water chemistry for laboratory analysis
5. Vernal pool searches
6. Spring habitat assessment
    a. Trash rating
    b. Distance of nearest road to site
    c. Riparian buffer width and riparian vegetation
    d. Adjacent land cover
    e. Buffer breaks and buffer break types
    f. Channelization
    g. Land use
    h. Stream gradient

A digital photograph monitoring record will provide documentation of visual changes at each site throughout the course of the project. Two photographs are taken from the mid-point of the site, mid-channel, one looking upstream and one looking downstream. These photographs are typically taken during the Spring Index Period and are used to depict the general appearance and conditions of the stream.

Summer Index Period

During the Summer Index Period, sampling methods will vary between Fish and Benthic Macroinvertebrate Sites. At the Fish Sites, sampling methods from the MBSS Summer Index Period will be conducted, including:

1. Quantitative fish- Aggregate American eel biomass is to be recorded separately from non-eel fish aggregate biomass.
2. Recovering temperature loggers
3. Invasive plant, reptile, amphibian, crayfish, and freshwater mussel searches
4. Summer habitat assessment
    a. Habitat assessment metrics
    b. Riffle embeddedness
    c. Shading
    d. Woody debris and root wads
    e. Stream character
    f. Maximum depth, wetted width, thalweg depth, and thalweg velocity
    g. Discharge
    h. Bank erosion
    i. Bar formation and substrate
5. Benthic macroinvertebrates (following Spring Index Period protocols)

At the Benthic Macroinvertebrate Sites, MBSS protocols have been adapted for the goals of this project. The following sampling methods will be conducted:

1. Benthic macroinvertebrates (following Spring Index Period protocols)
2. Qualitative fish
3. Select summer habitat assessment
    a. Stream character
    b. Wetted width, thalweg depth, and thalweg velocity
    c. Bank erosion
    d. Bar formation and substrate

Qualitative fish sampling at the Benthic Macroinvertebrate sites will consist of a minimum effort of 600 seconds with at least one backpack electrofishing unit throughout the site. The presence of all fish and other faunal groups observed, as well as the number of American eels and other migratory fish species collected, should be recorded on the MBSS Fish Data Sheet. The select habitat assessment parameters should allow major changes in stream morphology following dam removal to be documented and supplement geomorphological measurements to analyze the impacts of dam removal on macroinvertebrate communities.

Appendix 2.2: Sampling Locations and Data Collected on the Patapsco River

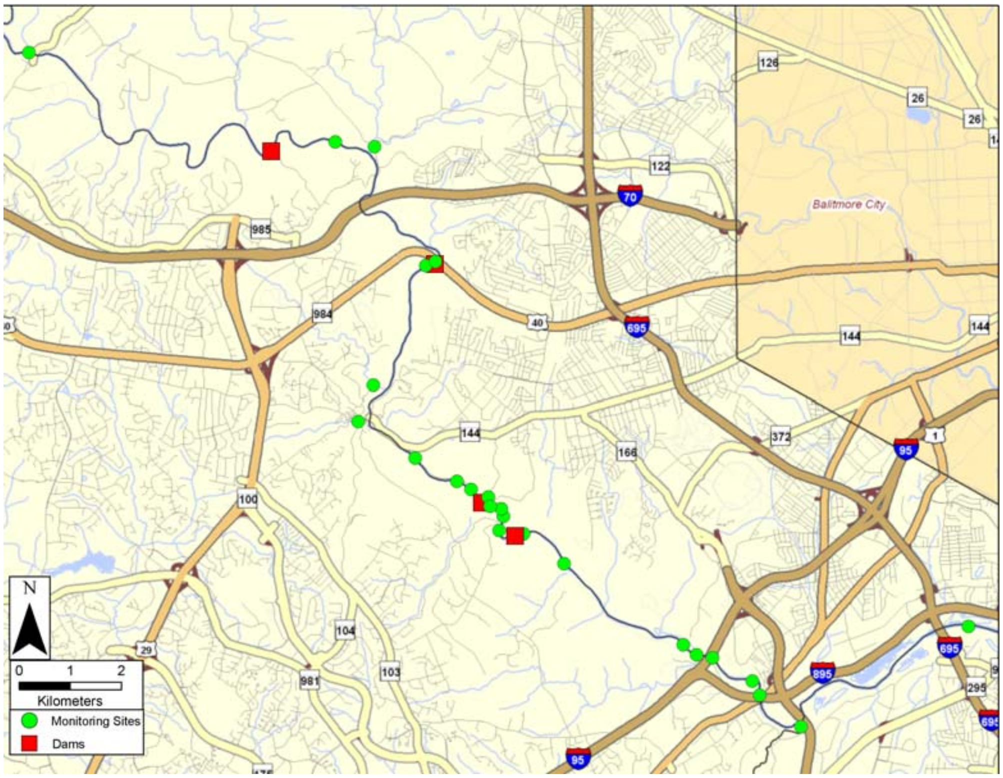

Figure 2.2: Locations of all sites sampled on the Patapsco River

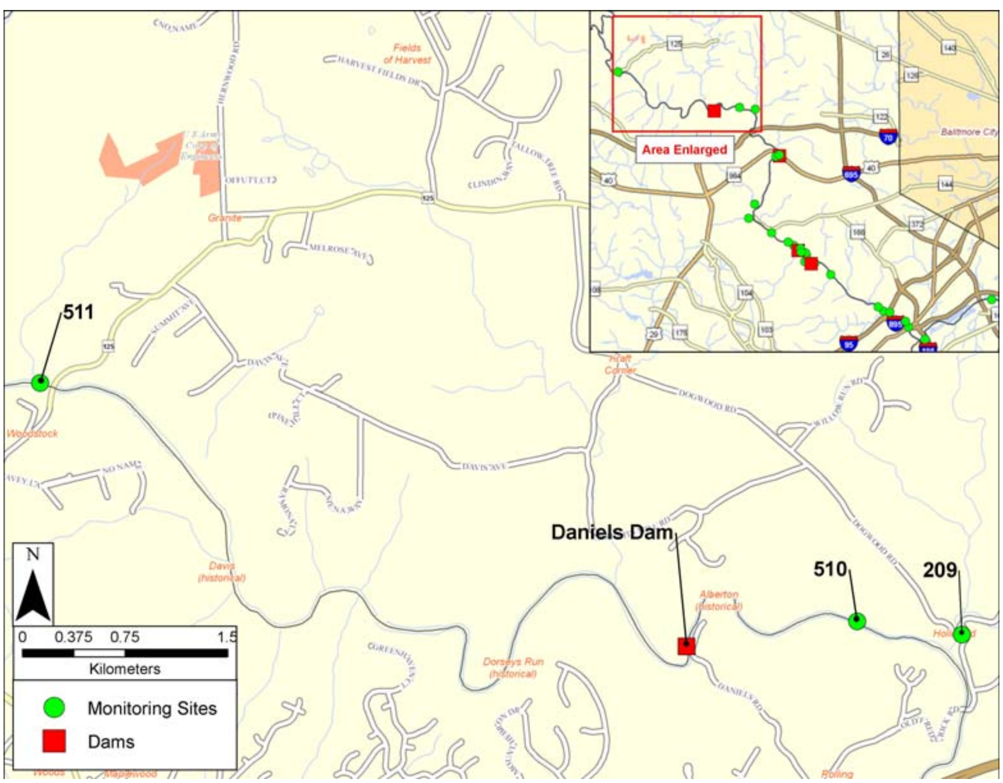

Figure 2.3: Sites Sampled in the vicinity of Daniels Dam

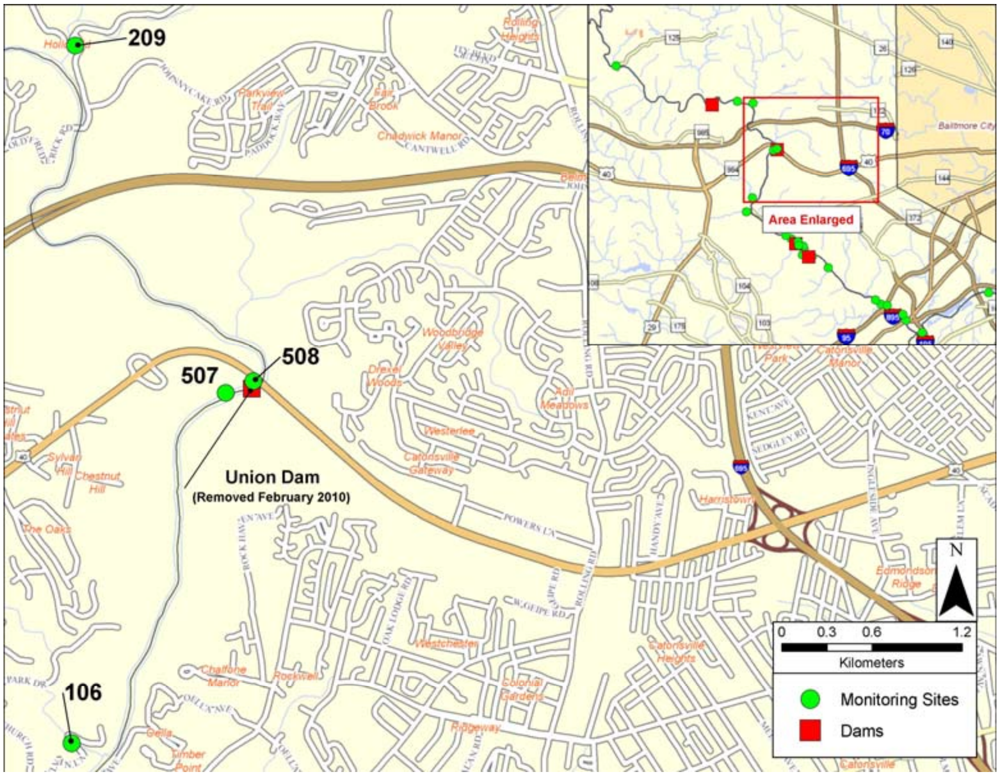

Figure 2.4: Sites Sampled in the vicinity of Union Dam

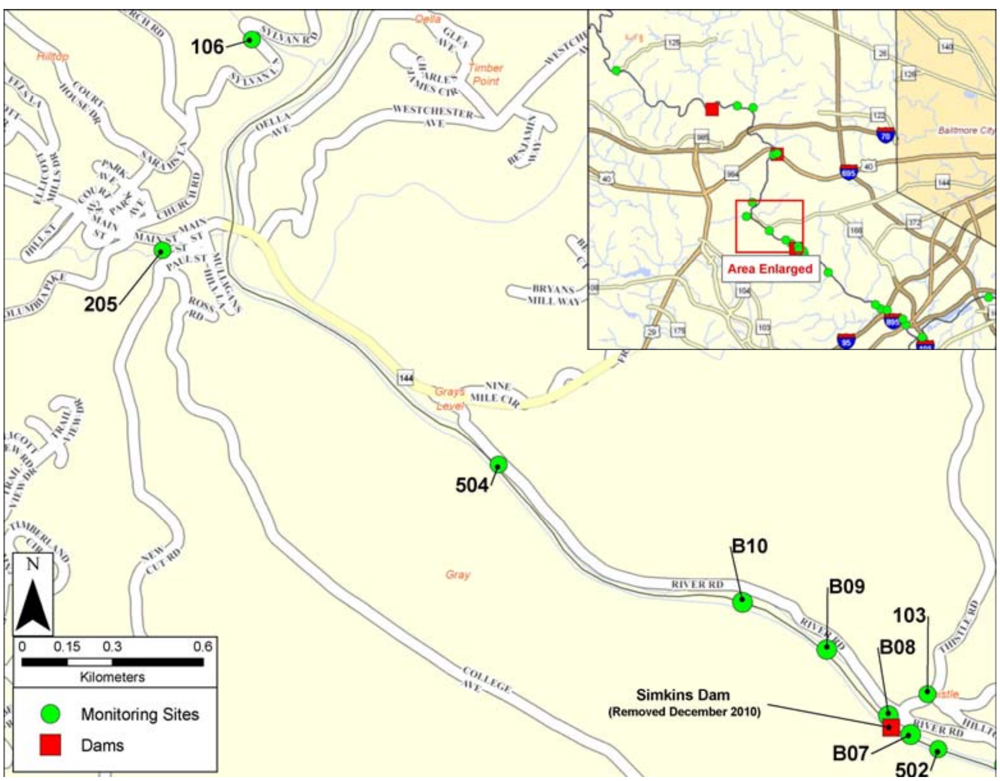

Figure 2.5: Sites sampled in the vicinity of Simkins Dam

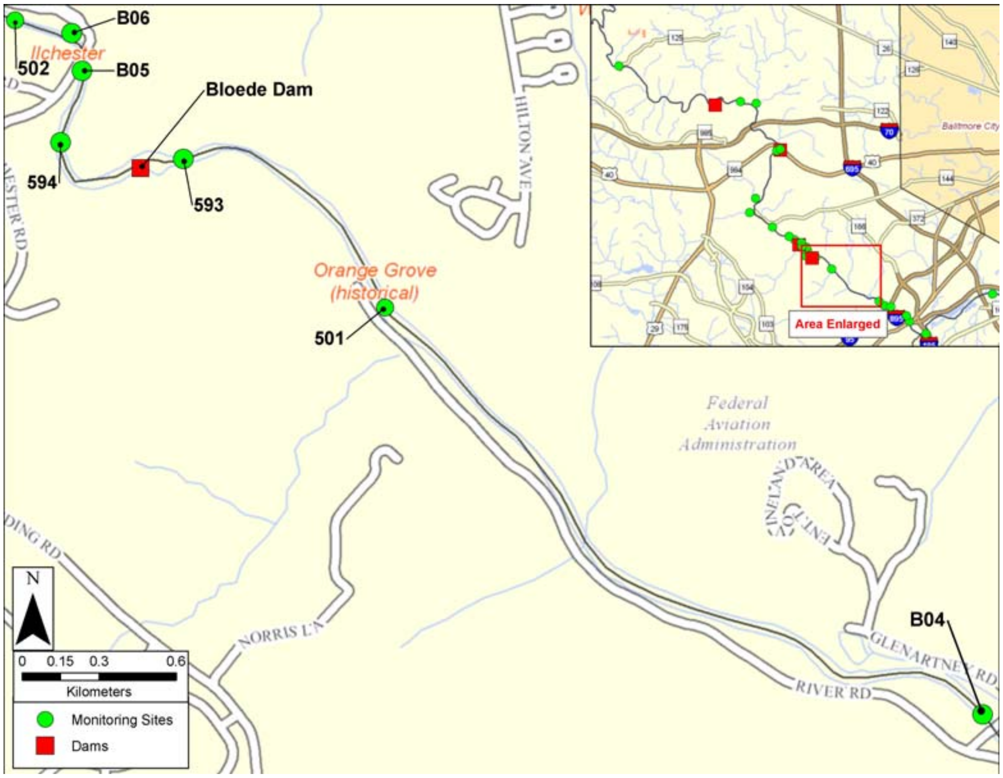

Figure 2.6: Sites sampled in the vicinity of Bloede Dam

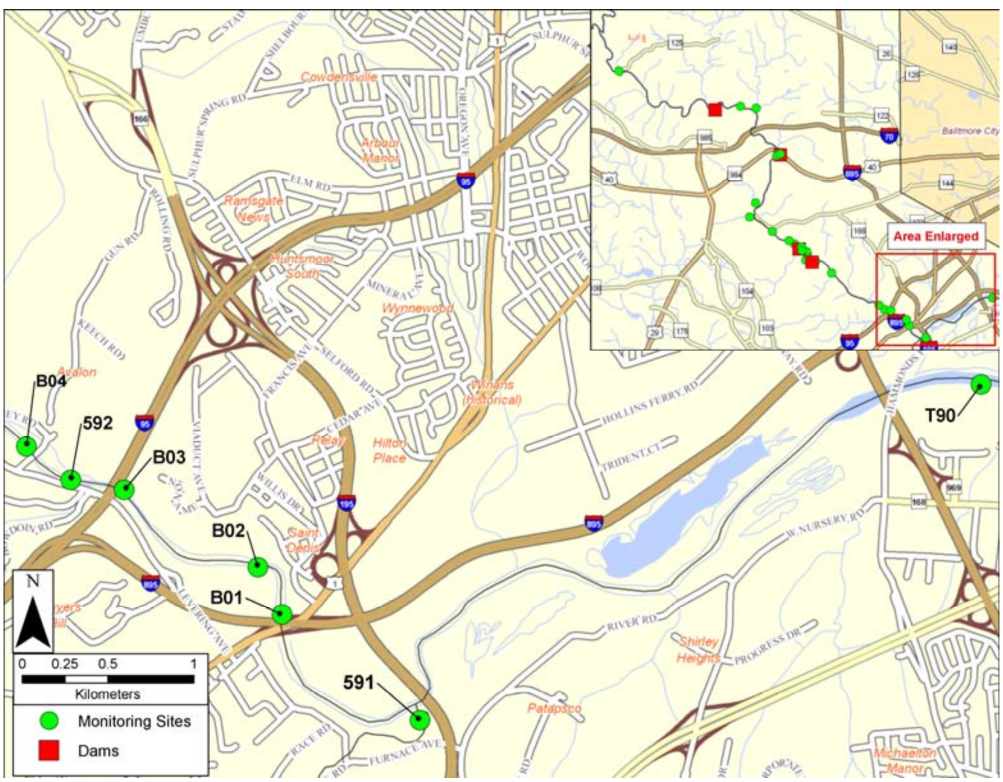

Figure 2.7: Sites sampled downstream of Bloede Dam

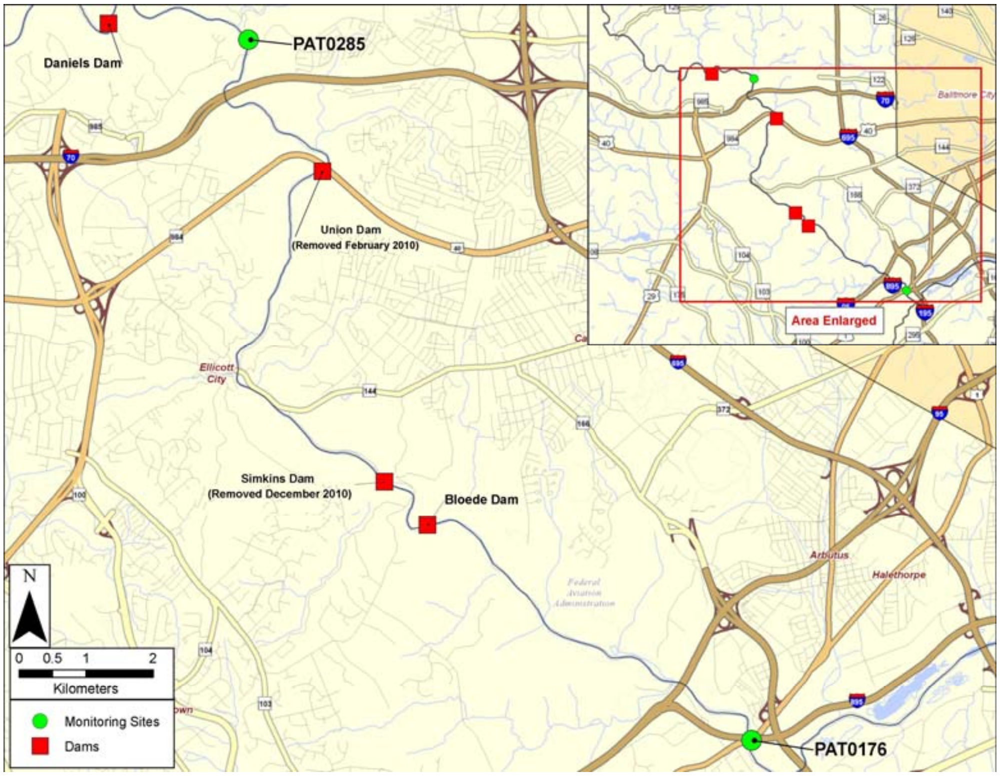

Figure 2.8: Locations of Core Trend sites on the Patapsco River

Table 2.1: Data collected at sites on the Patapsco River where sampling was conducted to assess changes due to the removal of Union and Simkins Dams

Site	Distance From Baltimore Harbor (rkm)	Anadromous Fish (weekly, March-May)		Water Chemistry (Monthly)				Benthic Macro-invertebrates						Physical Habitat (Summer)				Resident Fish (Summer)				Freshwater Mussels (Summer)				American Eels (Summer)			
		2011	2012	2009**	2010	2011	2012	2009 summer	2010 spring	2011 spring	2011 summer	2012 spring	2012 summer	2009	2010	2011	2012	2009	2010	2011	2012	2009	2010	2011	2012	2009	2010	2011	2012
T90	5.87	X	X																										
591	10.23	X	X																										
PAT0176	11.31			X	X	X	X																						
B01	11.41							X	X	X	X	X	X													X	X	X	X
B02	11.78							X	X	X	X	X	X													X	X	X	X
B03	12.75							X	X	X	X	X	X													X	X	X	X
592	13.06	X	X																										
B04	13.40							X	X	X	X	X	X													X	X	X	X
501	16.37							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
593	17.46	X	X																										
Bloede Dam	17.63																												
594	18.05	X	X																										
B05	18.35							X	X	X	X	X	X													X	X	X	X
B06	18.54							X	X	X	X	X	X													X	X	X	X
502	18.76							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
B07	18.87							X	X	X	X	X	X													X	X	X	X
Simkins Dam	18.93																												
B08	18.97							X	X	X	X	X	X													X	X	X	X
103*	18.98							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
B09	19.27							X	X	X	X	X	X													X	X	X	X
B10	19.59							X	X	X	X	X	X													X	X	X	X
504	20.58							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
205*	21.81							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
106*	22.51							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
507	25.17							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Union Dam	25.35																												
508	25.37							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
PAT0285	28.51			X	X	X	X																						
209*	28.76							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
510	29.48							X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Daniels Dam	31.14																												
511	38.44								X	X	X	X	X		X	X	X		X	X	X		X	X	X		X	X	X

*Sites 103, 205, 106, and 209 are on tributaries, all others on the Patapsco mainstem.
** Monthly samples from January 2000 were used in this study.

Chapter 3: Physical Habitat

Introduction

The removal of Union and Simkins dams is expected to provide benefits to the Patapsco River ecosystem by improving the connectivity of the river for the unimpeded movement of fishes and other river dwelling animals. However, because Simkins Dam was removed without first removing the sand and gravel from behind it, the potential exists for biological changes due to the displacement of this material from upstream to downstream of the dam. Additionally, two large storm events during the fall of 2011 (September and November) caused substantial movement of stream substrates. Stream bottom composition and changes to it are potentially key factors in determining both ecological conditions and the effect of sediment released from Simkin's removal.

The type of material that makes up the stream bottom plays an important role in determining the types, abundances, and condition of animals that inhabit a stream or river. A thin (< 1 cm) layer of fine sediment over a stream bottom that was previously dominated by coarse (gravel or cobble) substrate can be sufficient to fill interstitial spaces between rocks, thus displacing insects and benthic fishes that utilize these habitats. Fish and insects that prefer sand or silt substrates (e.g., burrowing insects) can benefit from the change in habitat and may subsequently increase. Likewise, if fine material is displaced (as is likely upstream from a dam after the dam is removed) uncovering larger substrates (e.g., cobble, boulder, gravel), organisms preferring the newly exposed coarser sediment are likely to increase.

Methods

To help analyze and interpret ecological data within the context of a changing stream bottom, crews collecting resident fish, American eel, and benthic macroinvertebrate data at 21 sites in the non-tidal river noted the primary substrate composition each time they sampled. This was done by recording the dominant substrate observed on the stream bottom within the site sampled for biology. Additionally, McCormick Taylor Inc. established cross-sections and conducted surveys of the river's substrate throughout the area affected by dam removal. Eight of these cross sections were conducted at the same location where the some of the ecological monitoring sites (Fig. 3.1) were sampled. This information was used to verify substrate composition observations recorded at those sites.

The quality of physical habitat to support fish and benthic macroinvertebrates in the Patapsco River was rated using standard MBSS habitat assessment methods (Stranko et al. 2007) at the seven mainstem non-tidal sites where resident fish, benthic macroinvertebrate, and American eel data were also collected. These rankings range on a scale of 0 to 20, with 0 being the poorest available habitat and 20 being the most optimal. These ratings were based on observations indicating the quality and stability of habitat within a sampling site. Separate habitat assessments were conducted for the quality of fish and benthic macroinvertebrate habitat. Biologists use extensive experience from hundreds of streams throughout Maryland, representing the total range of conditions, when conducting such assessments. Examples of habitat elements examined for ratings include the relative proportions of cobble, boulders, woody debris and other stable features that provide relief to the stream and habitat for biota.

The ratings were compared before and after dam removal to determine if changes to stream habitat quality (presumably as caused by the displacement of sand from upstream of dams to downstream) were detected.

Results

Prior to dam removal, the river bottom consisted primarily of sand at six of the 21 ecological monitoring sites. Four of these sites were located immediately upstream of Simkins, Union, or Bloede dams. The other two sites were the downstream-most sites on the Patapsco River and were near the head of tide. All other sites were composed primarily of either sand and gravel or cobble and gravel. After the dams were removed, the sites immediately upstream from Union and Simkins dams primarily consisted of cobble and gravel. Sand was the primary substrate at all sites downstream from Simkins Dam and upstream of Bloede Dam, indicating that a substantial portion of the river bottom had become covered with sand that was previously deposited behind Simkins Dam. Through the summer of 2011, the bottom substrate of sites below Bloede Dam did not change substantially. However, according to data from McCormick Taylor and supported by observations at DNR's ecological monitoring sites, the two large storms during the fall of 2011 resulted in short term deposition of substantial sand at the site immediately downstream from Bloede Dam and longer-term deposition at the next two sites downstream. These observations are presented in detail in Table 3.1.

While ratings were variable over time at all sites, physical habitat quality for fish and macroinvertebrates followed a temporal pattern similar to that observed in river bottom composition (Fig. 3.2). No consistent pattern in habitat quality was observed at sites in the vicinity of Union Dam, indicating that the active sediment removal utilized was successful at limiting major habitat changes in the river. A consistent decline in habitat quality was evident downstream of Simkins Dam the first year after the dam was removed. Presumably this was due to the effect of sediment transport from behind Simkins Dam and deposition in this area. Habitat quality improved the following year, possibly because at least some of the sediment had moved further downstream and out of this area. Habitat also improved upstream of Simkins Dam after it was removed (2011) and remained higher than prior to removal the next year (2012). The habitat quality at the downstream-most site where habitat was rated (downstream of both Simkins and Bloede dams) declined each year. This sequential decline may have been due to the continuous movement of sand from behind the dams to this area downstream. Presumably, habitat quality will improve throughout the non-tidal portion of the river as the sediment eventually moves further downstream.

Table 3.1: River bottom composition at 21 ecological monitoring sites sampled in the Patapsco River to assess the effects on dam removal.

	River Bottom Composition			
	Pre		Post	
Site	2009	2010	2011	2012
+ **B01**	Sand	Sand	Sand	Sand
+ **B02**	Sand	Sand	Sand	Sand
B03	Sand/gravel	Sand/gravel	Sand/gravel	Sand
+**B04**	Sand/gravel	Sand/gravel	Sand/gravel	Sand
+**501**	Sand/gravel	Sand/gravel	Sand/gravel	*Sand/gravel
Bloede Dam				
B05	Cobble/Gravel	Cobble/Gravel	Sand	Sand
B06	Cobble/Gravel	Cobble/Gravel	Sand	Sand
502	Cobble/Gravel	Cobble/Gravel	Sand	Sand
B07	Cobble/Gravel	Cobble/Gravel	Sand	Sand
Simkins Dam				
B08	Sand	Sand	Cobble/Gravel	Cobble/Gravel
103	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
+**B09**	Sand	Sand	Cobble/Gravel	Cobble/Gravel
+**B10**	Sand	Sand	Cobble/Gravel	Cobble/Gravel
+**504**	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
205	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
106	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
507	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
Union Dam				
+**508**	Sand	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
209	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
510	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel
Daniels Dam				
511	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel	Cobble/Gravel

*= Sand was added during September 2011 tropical storm and removed during November tropical storm
+ = Cross section data from McCormick Taylor was used to confirm visual observations

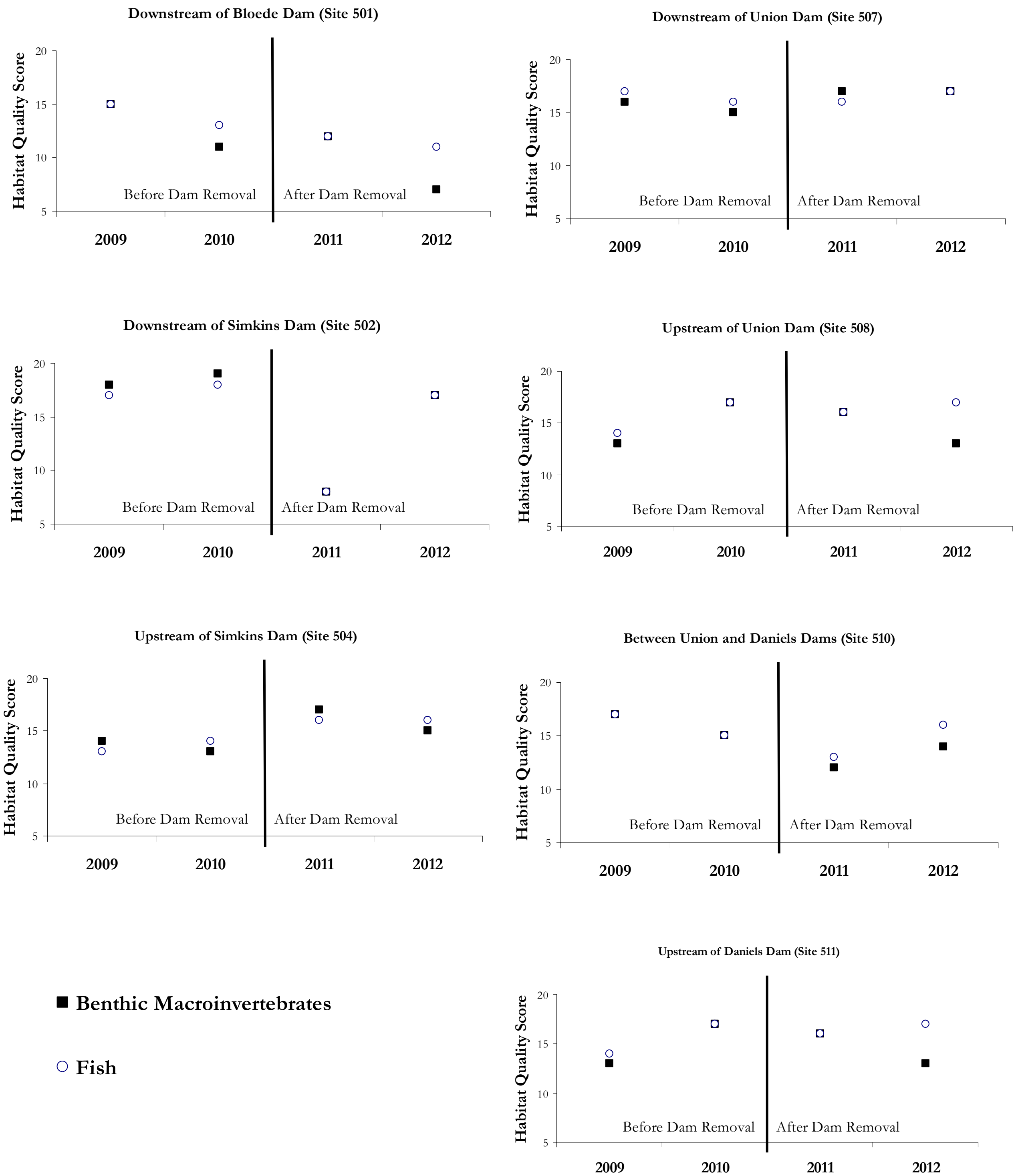

Figure 3.1: Habitat quality scores at sites sampled in the Patapsco River before and after dam removal.

Discussion

Physical habitat quality is integrally linked to biological condition in streams (Southerland et al. 2005). For example, habitat quality influences important biological indicators for the Patapsco dam removal project. Based on data collected throughout the Piedmont physiographic region during 2000 – 2011 by the Maryland Biological Stream Survey, EPT (mayfly, stonefly, and caddissfly) benthic macroinvertebrate taxa richness (Figure 3.2) and American eel density (number per square meter; figure 3.3) are correlated with physical habitat assessment scores. Thus, the changes to the physical habitat of the Patapsco River associated with dam removal, as described in this chapter, have the potential to influence the river's ecology.

Figure 3.2: Positive correlations between sensitive benthic macrovinvertebrate taxa and habitat scores in the Piedmont in Maryland.

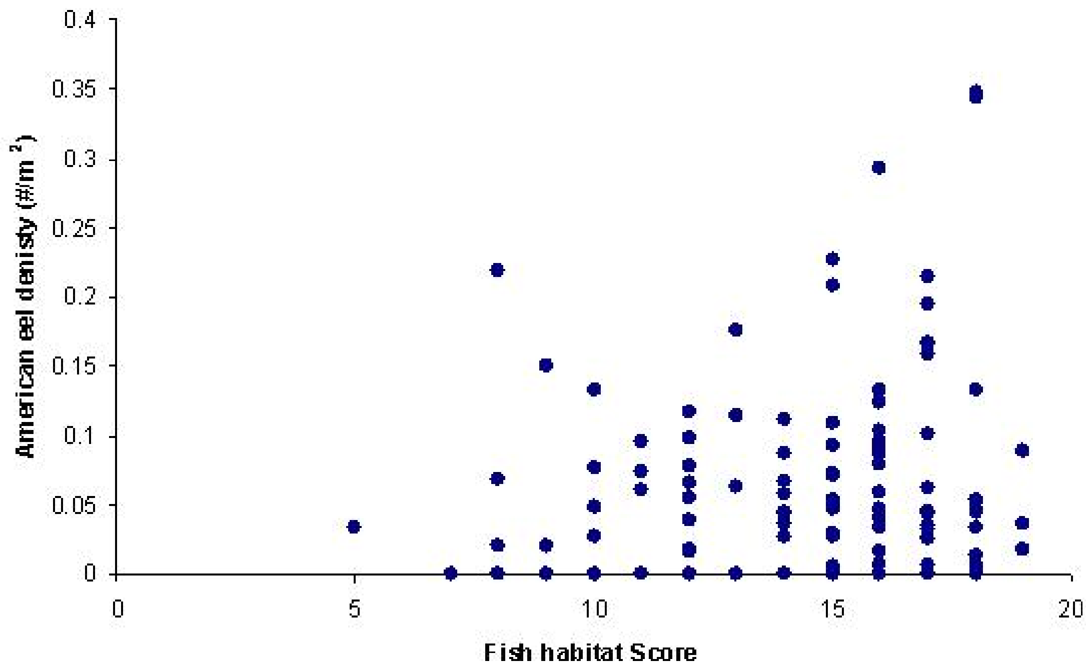

Figure 3.3: Positive correlations between American eel densities and habitat scores in the Piedmont in Maryland.

Physical habitat changes in rivers associated with dam removal have been shown to vary with dam size, removal techniques, the quantity and type of sediment stored, river gradient, , hydrologic conditions experienced, and other factors (Pizzuto 2002, Doyle et al. 2005). Most changes to the river channel occur within the first five years of dam removal. Improvements to the habitat for fish and benthic macroinvetebrates may be delayed in the Patpasco River by Bloede Dam because of its ability to attenuate sediment dispersal downstream from Simkins Dam. Despite this, our observations from two years of post-dam removal monitoring suggest that habitat is improving after the removal of Simkins Dam. As the habitat continues to improve, we expect concomitant improvements in fish and benthic macroinvertebrate habitat and consequent improvements biological communities.

Presumably the sand that was behind Simkins Dam will continue to move further downstream over time and removing Bloede Dam would likely expedite the movement out of the non-tidal portion of the river (Stillwater Sciences 2010, McCormick Taylor 2013). The active sediment removal and the existing breach at Union Dam seem to have limited the dispersal of sediment with little concomitant change in habitat quality in the vicinity of Union dam. The potential influence the stream bottom composition and changes to it had on river ecology is discussed within the ecological monitoring results chapters in this report.

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Monitoring Period June, 2012 to January, 2013. Prepared by McCormick Taylor. Baltimore MD. May, 2013. Pizzuto, J. 2002. Effects of dam removal on river form and process. BioScience 52:683-692.

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Stranko, S. A., D. M. Boward, J. V. Kilian, C. J. Millard, A. J. Becker, R. Gauza, A. Schenk, A. Roseberry-Lincoln, and M. O'Connor. 2007. Maryland biological stream survey sampling manual. Maryland Department of Natural Resources, Annapolis, Maryland.

Chapter 4: Water Quality

Introduction

Most of Chesapeake Bay and its tidal tributaries are listed by the Maryland Department of the Environment (MDE) and the Environmental Protection Agency (EPA) as "impaired" because of excess nitrogen, phosphorus and sediment (http://www.epa.gov/reg3wapd/tmdl/ChesapeakeBay/tmdlexec.html). In addition, the Lower North Branch of the Patapsco River (the portion of the river that this study focuses on) is listed by MDE as impaired by heavy metals, nutrients, suspended sediments, and fecal coliform bacteria (www.mde.state.md.us/programs/Water/TMDL/ApprovedFinalTMDLs/Pages/Programs/WaterPrograms/TMDL/approvedfinaltmdl/wqa_final_lnbpatapsco_metals.aspx). These listings establish a need for reductions within the Patapsco River and a need to limit these chemicals coming from the river to the estuary downstream. This listing was in place prior to the dam removals.

Recently, special focus has been paid to reducing nutrients and sediments to Chesapeake Bay (www.epa.gov/chesapeakebaytmdl) in hopes of restoring ecological integrity and fisheries productivity. Stream restoration, special agriculture practices, waste water treatment plant upgrades, and other management efforts are being applied throughout the watershed to reduce nutrients and other chemicals. Dams, like Union and Simkins, can potentially contribute to reduced loads to Chesapeake Bay simply by increasing retention time of water for the processing of nutrients (Caraco and Cole 1999). However, the impoundment areas upstream of Union and Simkins Dams were relatively small and shallow, thus reductions of nutrients within these former impoundments was probably negligible. The riparian buffer revegetation and eventual stabilization of soil along the banks in the riverine system where the impoundment above Simkins Dam once existed should enhance nutrient processing. As a result, the reductions produced by these buffer improvements should exceed those once achieved within the former impoundments.

The large majority of sediment particles (sand and gravel) found behind Simkins Dam is not considered to be the type that can effectively carry large quantities of nitrogen or phosphorus. However, it is possible that even a small proportion of the approximately 80,000 cubic yards of material behind the dam could carry enough nitrogen or phosphorus to show a measurable quantity downstream. Large quantities of fine sediment from larger dams have been shown to contain nitrogen and phosphorus nutrients (Stanley and Doyle 2002) and other materials bound to them that, when released, can temporarily influence water quality. Although Simkins Dam was substantially smaller and had relatively few fine particles upstream from it, released sediment from its removal could cause an acute pulse of nutrients downstream.

Methods

Two monitoring sites provided an opportunity to examine water quality changes associated with dam removal. Differences in water quality measurements between the two monitoring sites represent water quality differences along the 17 Rkm that separate these sites. Since Union and Simkins dams were between these two sites, we expect changes in the magnitude of differences in measurements from these sites following dam removal to be primarily associated with the dams having been removed. However, we recognize that it is possible for other factors (e.g., sewage leaks and water quality of tributaries) to contribute to differences in water quality between these sites. Monthly water quality data from ten years prior to dam removal and approximately one year after were examined from one site upstream of Union and Simkins Dam and one site downstream. Each monthly grab sample was analyzed for 17 water quality parameters using EPA approved laboratory techniques. We compared water quality measurements at the two sites over an approximately ten-year-period (pre-dam removal, January 2000 to November 2010). We then compared the range of differences from this period to differences measured between the sites from December 2010 through December 2011 (after Simkins and Union Dams were removed). We considered differences between measurements from the two sites post dam removal period that exceeded the differences measured during the pre dam removal period to indicate a possible water quality effect from dam removal.

Results and Discussion

There did not appear to be a change in most water quality parameters following dam removal (Appendix A, Fig. 4.1-4.17). Exceptions include phaeophyton a, particulate nitrogen, particulate carbon, particulate phosphorus, and total suspended solids. The differences in measurements for these parameters between the downstream and upstream monitoring sites were substantially greater during the post dam removal period than during the entire range of differences during the pre-dam removal period. The majority of the larger differences appeared to be due to measurements taken during large storms that occurred in the fall of 2011. Although sampling took place during the pre-dam removal period, the water quality grab sample taken during the post-dam removal period (especially in the fall of 2011) was taken during a flow that was higher than during any other sampling event. Figure 4.18 shows flows in the adjacent Gwynns Falls when samples were taken in the Patapsco. Gwynns Falls flow data were used because they cover the entire pre- and post- dam removal period when water quality grab samples were taken. Flow data from the Patapsco River were only available for part of that time.

Substantial sediment amounts that were previously behind Union and Simkins dams may have moved downstream during these storms. Nutrients and carbon within these sediments may have been released as it was mobilized. Certain algae may also have accompanied this material, as indicated by the increased difference in phaeophytin a following dam removal.

If releases of certain chemicals occurred when large storms mobilized material that had been trapped behind Union and Simkins Dams, the effect will likely diminish over time as the sediment and related material eventually leave the river. Monthly water quality sampling will continue at these two Core/Trend sites in the Patapsco River to determine if this is the case and to document the long-term water quality conditions of the Patapsco River. These data will also be useful in attempting to elucidate potential sources of differences between water quality measurements, and whether or not they related to dam removal.

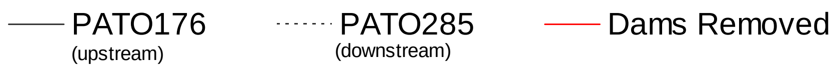
Literature Cited

Caraco, N.F. and J.J. Cole. 1999. Human impact on nitrate export: an analysis using major orld rivers. Ambio 28:167-170.

Stanley, E.H. and M.W. Doyle. 2002. A geomorphic perspective on nutrient retention following dam removal. BioScience 52:693-701.

Appendix 4.1: Water quality conditions at two sites in the Patapsco River before and after dam removals.

A solid line indicates water quality values measured upstream of Union and Simkins dams and a dashed line indicates values measured downstream.



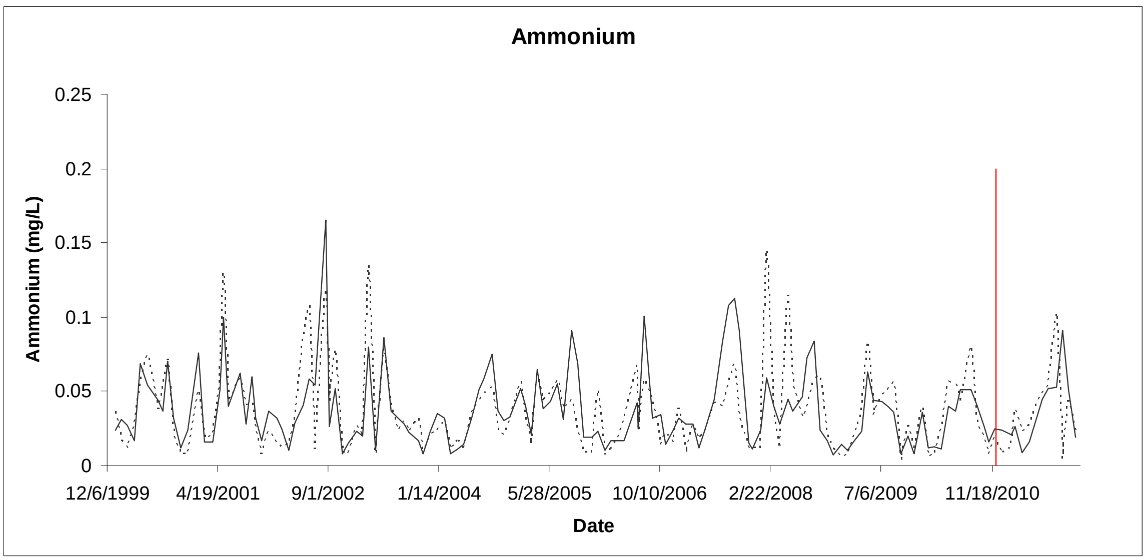


Figure 4.1: Ammonium at two sites in the Patapsco River before and after dam removals.

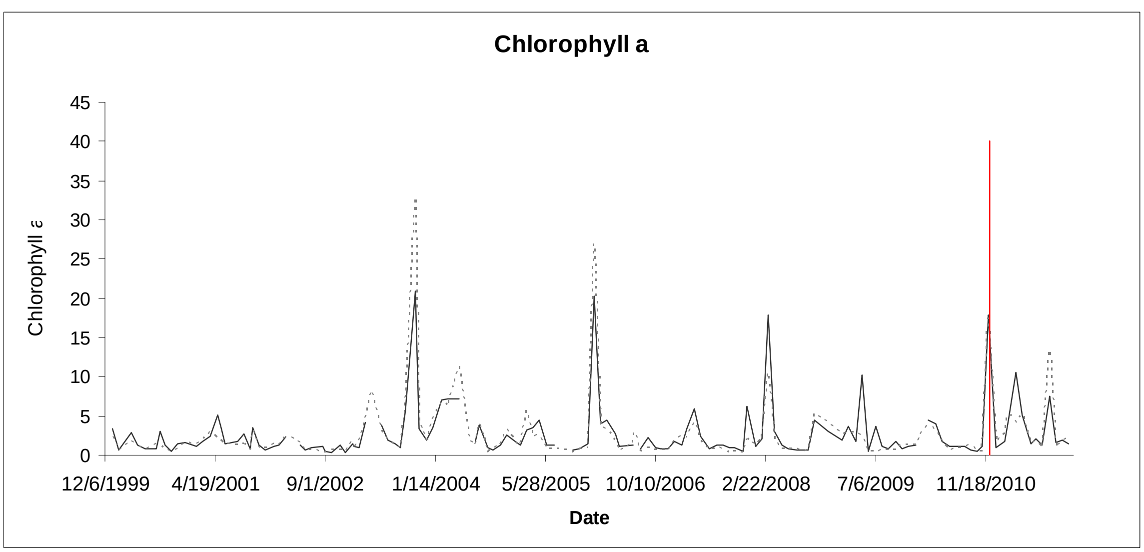


Figure 4.2: Chlorophyll a at two sites in the Patapsco River before and after dam removals.

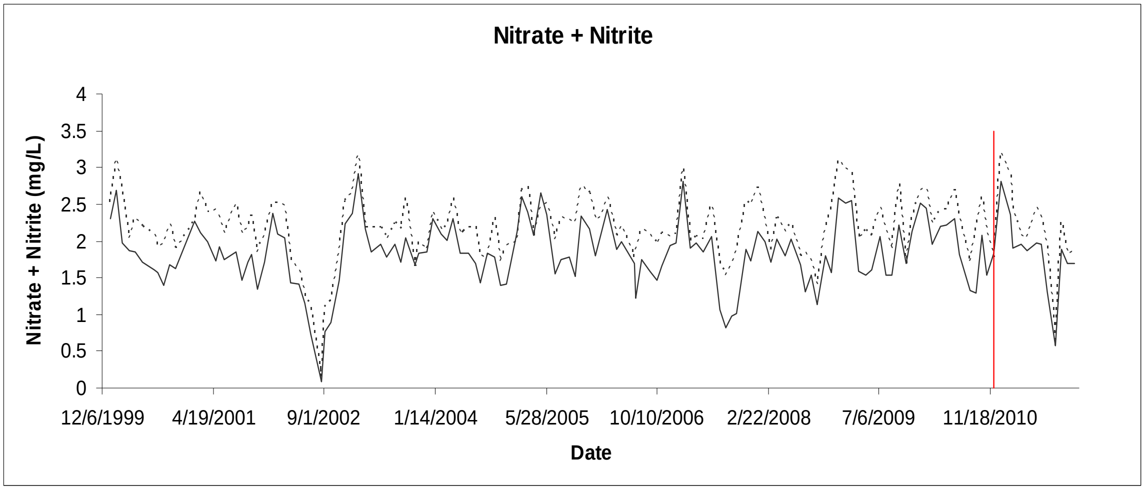


Figure 4.3: Nitrate + Nitrite at two sites in the Patapsco River before and after dam removals.

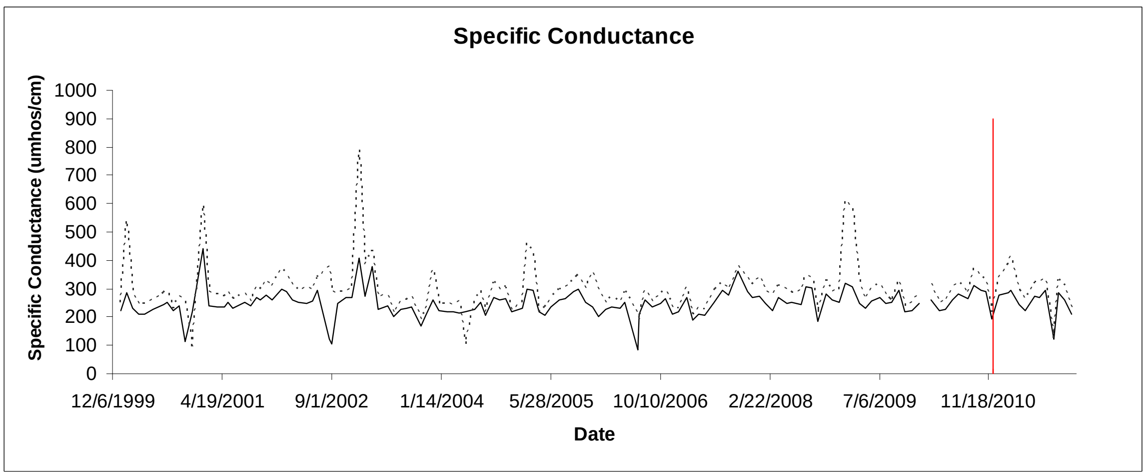


Figure 4.4: Specific Conductance at two sites in the Patapsco River before and after dam removals.

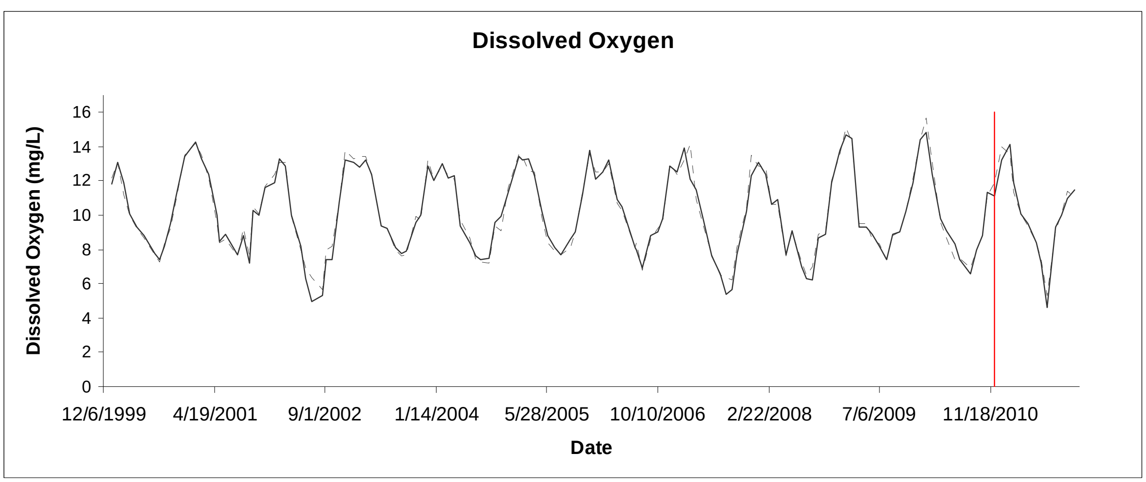


Figure 4.5: Dissolved Oxygen at two sites in the Patapsco River before and after dam removals.

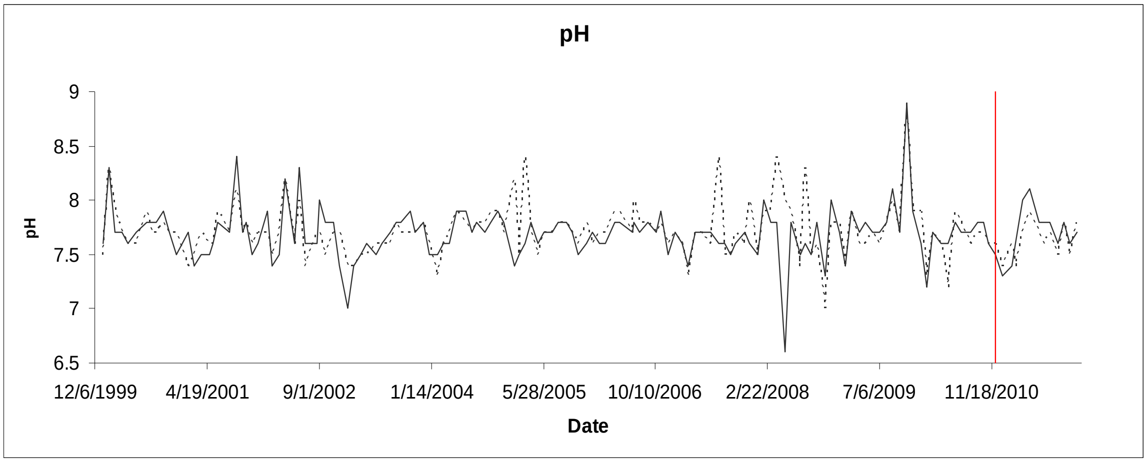


Figure 4.6: pH at two sites in the Patapsco River before and after dam removals.

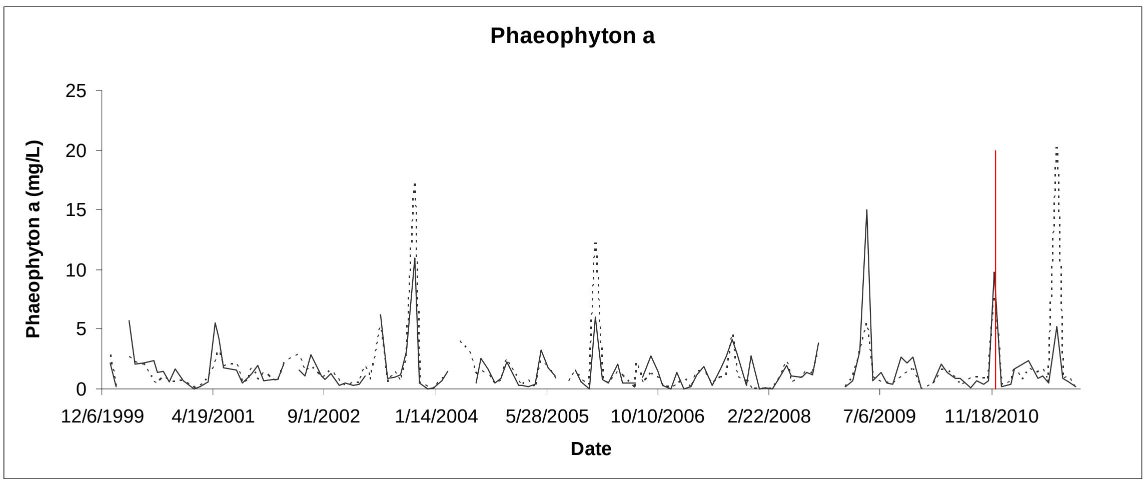


Figure 4.7: Phaeophyton a at two sites in the Patapsco River before and after dam removals.

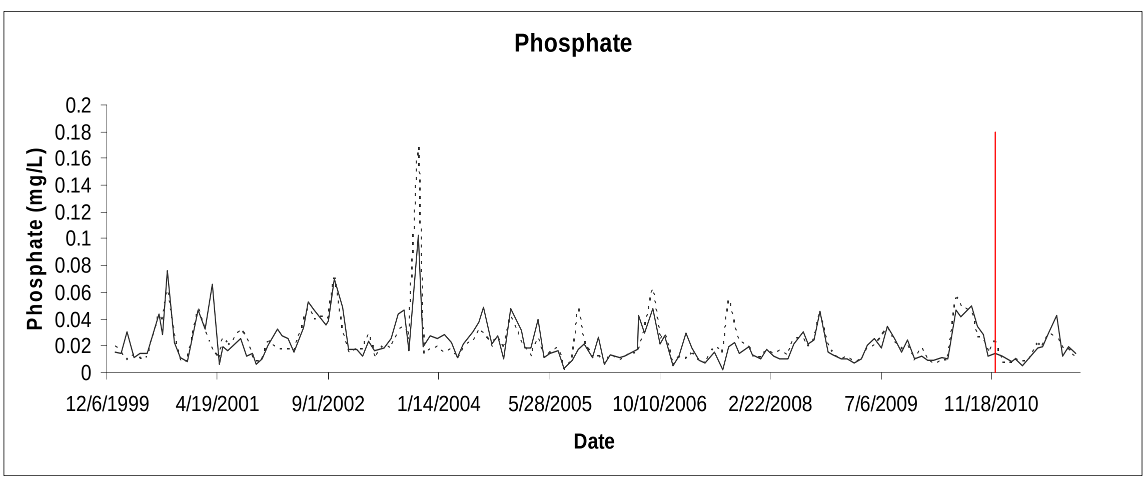


Figure 4.8: Phosphate at two sites in the Patapsco River before and after dam removals.

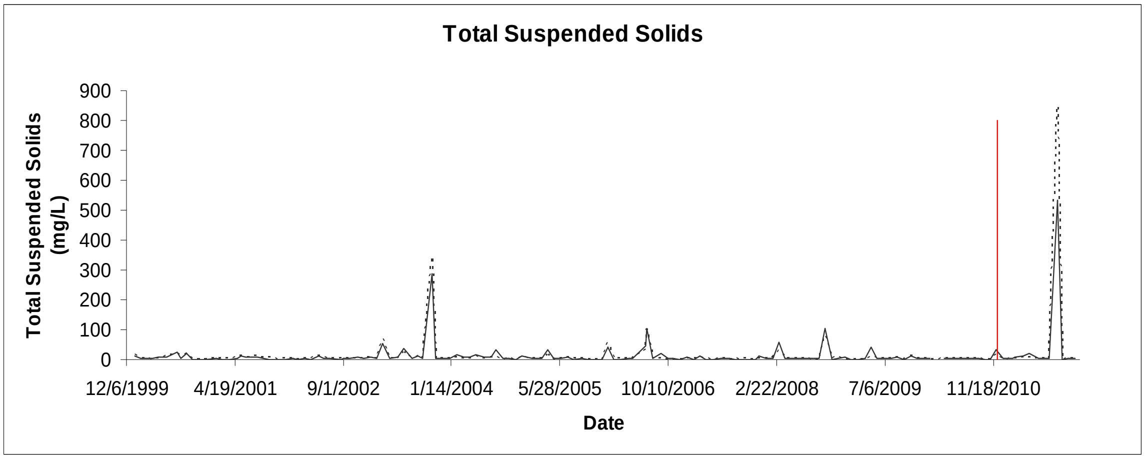


Figure 4.9: Total Suspended Solids at two sites in the Patapsco River before and after dam removals.

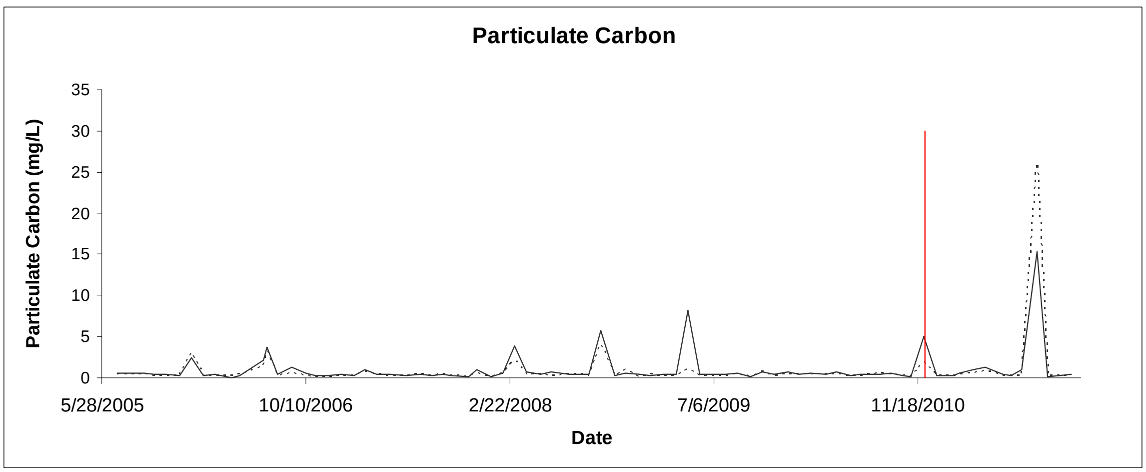


Figure 4.10: Particulate Carbon at two sites in the Patapsco River before and after dam removals.

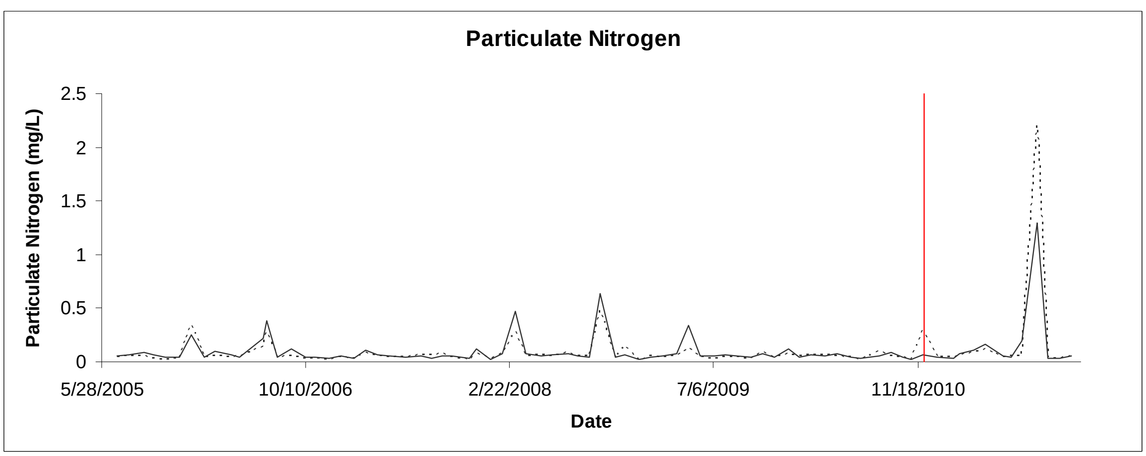


Figure 4.11: Particulate Nitrogen at two sites in the Patapsco River before and after dam removals.

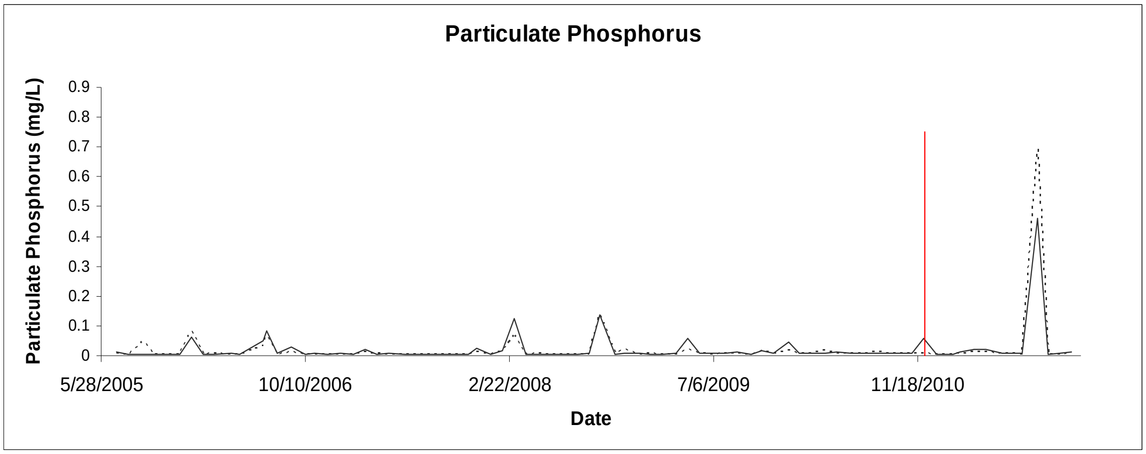


Figure 4.12: Particulate Phosphorus at two sites in the Patapsco River before and after dam removals.

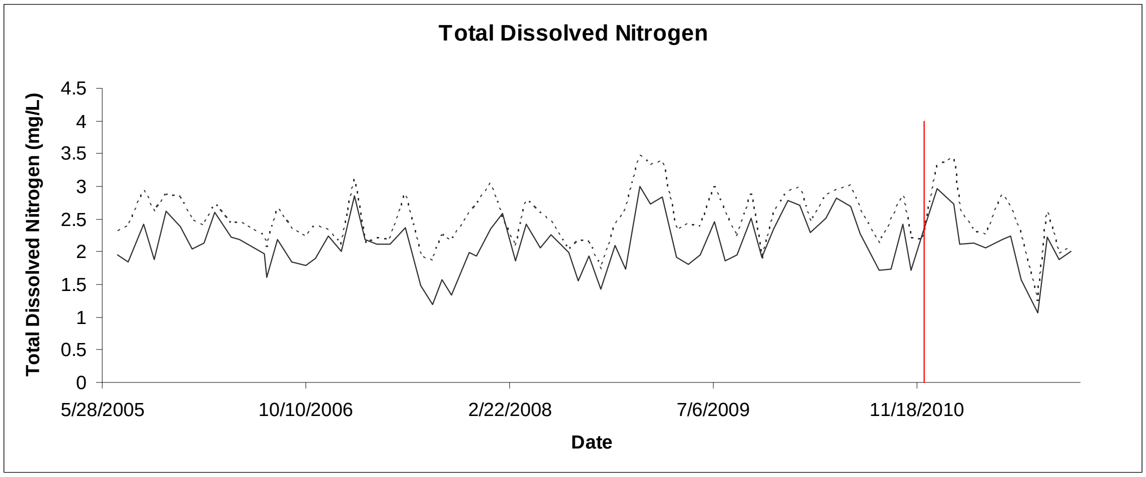


Figure 4.13: Total Dissolved Nitrogen at two sites in the Patapsco River before and after dam removals.

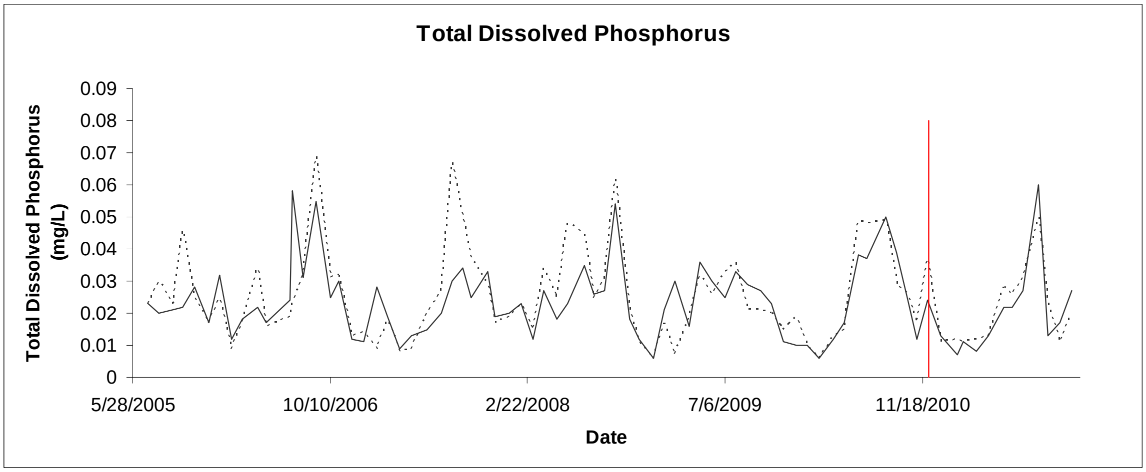


Figure 4.14: Total Dissolved Phosphorus at two sites in the Patapsco River before and after dam removals.

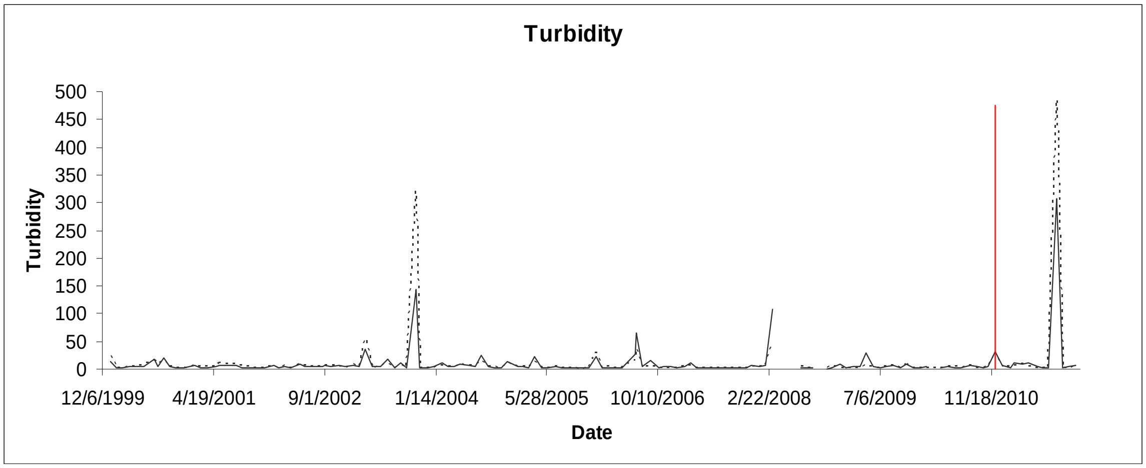


Figure 4.15: Turbidity at two sites in the Patapsco River before and after dam removals.

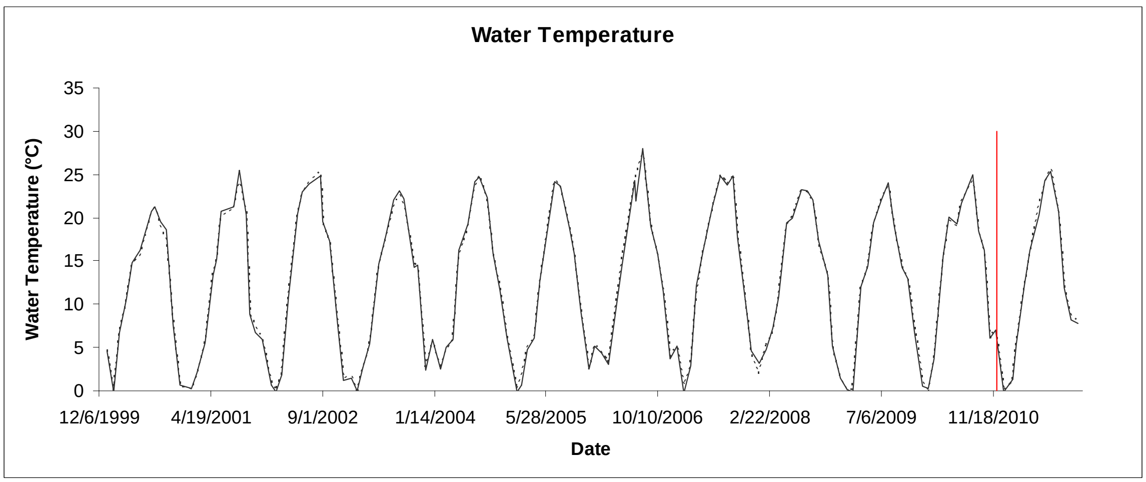


Figure 4.16: Water Temperature at two sites in the Patapsco River before and after dam removals.

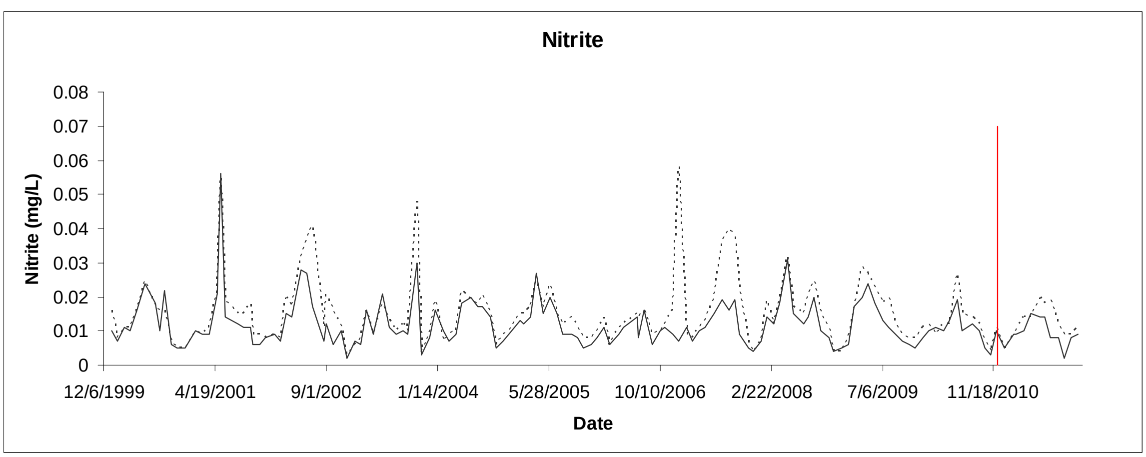


Figure 4.17: Nitrite at two sites in the Patapsco River preceding and following dam removals.

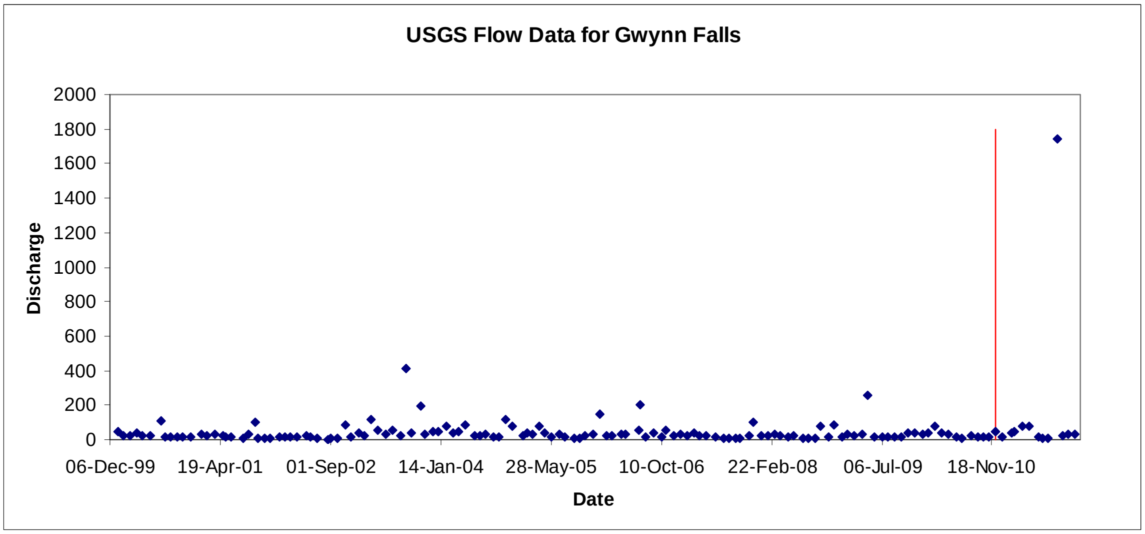


Figure 4.18: Flows in Gwynns Falls (adjacent to the Patapsco River) during the same times that water quality grab samples were taken.

Chapter 5: Anadromous Fishes

Introduction

Historically, several fish species migrated each spring from Chesapeake Bay to spawning habitat throughout the Patapsco River and its major tributaries. In the early 1900s, the construction of Daniels, Union, and Simkins dams to support industry, and Bloede Dam for power generation, eliminated access of these species to more than 27 kilometers of the Patapsco River mainstem. Although historic overfishing played a part, reduced stocks of these anadromous species in the river are largely the legacy of these major fish migration barriers.

Today, these migratory species still utilize the lower reaches of the Patapsco River for spawning; albeit in much reduced numbers. Based on fish surveys conducted in the mid-1970s and 1990s, anadromous species have been observed in the river downstream of the four dams (O'Dell et al. 1975; MDNR Fisheries Service, unpublished data). In 1992, a fish ladder was installed at the most downstream dam, Bloede Dam, to restore access to upstream reaches. This effort met with limited success. Of the eight species observed in the lower reaches of the Patapsco River, only American shad (*Alosa sapidissima*), sea lamprey (*Petromyzon marinus*), and blueback herring (*Alosa aestivalis*) had been documented using the fish ladder since its installation, and all were observed in low abundances. While some of this can be attributed to very limited sampling effort after the installation of the ladder, Bloede Dam nonetheless appears to remain a significant migration barrier to anadromous fishes.

The removal of Union, Simkins, and Bloede dams by the joint efforts of American Rivers, Friends of the Patapsco Valley State Park, the Maryland Department of Natural Resources, and the National Oceanic and Atmospheric Administration will re-open over 13 river kilometers of historical mainstem spawning habitats to anadromous fishes in the Patapsco River, and many more kilometers of tributaries. This effort provides a unique opportunity to document the effects of dam removal on migratory fishes. Presumably, the additional spawning habitat made available by removing these dams will improve spawning stocks over time. To measure this effect of the restoration effort, it is necessary to document the use of the river by these species prior to the removal of Bloede Dam and to monitor migrations of these fish once access to a larger portion of the Patapsco has been restored.

Our monitoring aims to determine the extent to which spring migrating fish species use the Patapsco River prior to and following the removal of Union, Simkins, and eventually Bloede dams. The monitoring effort is designed to answer the following management questions:

1. What fish species currently enter the Patapsco River during the spring?
2. What is the upstream extent of each anadromous species' distribution in the river?
3. Do any species currently reach Bloede Dam?
4. How much of a barrier is Bloede Dam to migration?
5. How do the distributions and abundance of spring migrating species change following dam removal?

Monitoring during 2011 and 2012 provided preliminary answers to the first four questions. Answering all five questions more completely will require multiple years of monitoring. We hope to continue monitoring the anadromous fish assemblage in the Patapsco River until Bloede Dam is removed. An additional two years or more of monitoring will be necessary to measure the effects of all three dam removals on anadromous fishes. This report documents the results of the first two years of pre-Bloede dam removal monitoring, during the springs of 2011 and 2012.

Methods

Sampling took place at five locations on the Patapsco River– three sites downstream of Bloede Dam in the tidal portion of the river, one site directly below Bloede in the dam's tailrace, and one site directly upstream of the dam (See Chapter 2, Appendix 2.1: Figures 2.6-2.7).

We used a boat mounted electrofisher downstream of Bloede Dam to sample fishes in the river below the dam and a fyke net fitted to the exit of the dam's fish ladder to trap fish passing the dam via the ladder. In 2011, we sampled weekly from 6 April until 12 May and in 2012 from 2 March through 17 May. Electrofishing was performed while moving downstream, with total shocking time, fish species observed, and abundance of migratory species recorded for each site. The fyke net was set for periods ranging from 45 and 192 hours, and was checked roughly every 24 hours during each set, with species collected, abundance, and time of deployment recorded.

Using the recorded time (either spent electrofishing or with the fyke net deployed) and the numbers of anadromous fish caught, we calculated catch-per-unit-effort (CPUE) in number of fish/hour as a measure of relative abundance for each species. CPUE is referred to as "abundance" hereafter. We did not record abundance of resident (non-migratory) fish species while electrofishing, keeping only a list of the species encountered. We recorded abundance of resident fish collected using the fyke net at the Bloede Dam fish ladder.

Results

We collected six species of anadromous and semi-anadromous fish including blueback herring, hickory shad (*Alosa mediocris*), sea lamprey, striped bass (*Morone saxatilis*), white perch (*Morone americana*), and yellow perch (*Perca flavescens*) by electrofishing at the four sites downstream of Bloede Dam. We collected all six of these species when we first began monitoring in 2011; but in 2012, we collected only blueback herring, sea lamprey, white perch, and yellow perch. We spent a total of 5.79 hours electrofishing between all four sites in 2011 and 6.77 hours in 2012.

In addition to the six species of anadromous fish we collected that are typically considered migratory, quillback (*Carpiodes cyprinus*), a large native sucker that superficially resembles a common carp, were also collected during the spring spawning run (Fig. 5.2). Although there are historic records of this species in the Patapsco River, quillback had not been previously documented during MBSS sampling. Quillback are a potadromous species-living in rivers and migrating to spawn in tributaries (Jenkins and Burkhead 1993). Quillback may live primarily in the tidal freshwater portion of the Patapsco River and move upstream to the non-tidal river to spawn during the spring.

Fig. 5.1: Quillback caught during spring 2011 on the Patapsco River.

In both 2011 and 2012, we collected 35 species of resident/non-anadromous fish by electrofishing at the four sites located downstream of Bloede Dam (Table 5.1).

Table 5.1: Resident fish species collected via electrofishing at four sites downstream of Bloede Dam during spring 2011 and 2012.

Species	2011	2012
banded killifish	X	X
black crappie	X	X
bluegill	X	X
bluntnose minnow		X
brown bullhead		X
brown trout	X	X
central stoneroller	X	
chain pickerel		X
channel catfish	X	
common carp	X	X
common shiner	X	X
fallfish	X	X
gizzard shad	X	X
green sunfish	X	X
inland silverside	X	X
largemouth bass	X	X
margined madtom		X
northern hogsucker	X	X
pumpkinseed	X	X
quillback	X	X
rainbow trout	X	X
redbreast sunfish	X	X
river chub	X	X
rock bass		X
satinfin shiner	X	X
shield darter	X	
smallmouth bass	X	X
spotfin shiner	X	X
spottail shiner	X	X
swallowtail shiner	X	X
tessellated darter	X	X
white sucker	X	X
yellow bullhead		X
Total	27	30

Of the six anadromous species collected, we only collected sea lamprey upstream of Bloede Dam. In 2011, we collected sea lamprey by electrofishing and caught no fish (resident or anadromous) in the fyke net. In 2012, following repairs to the fish ladder completed the previous winter; we collected 13 species of resident fish in the fyke net, but no anadromous fish species. Resident fish species included bluegill, brown trout, channel catfish, common carp, fallfish, gizzard shad, northern hogsucker, rainbow trout, redbreast sunfish, rock bass, smallmouth bass, white catfish, and white sucker. We deployed the fyke net deployed for a total of 115 hours during May 2011 and 689 hours between March and May 2012 (fewer hours were fished in 2011 because the net was not obtained until May of that year). We did not electrofish above Bloede Dam in 2012.

Five of the six anadromous species collected (blueback herring, hickory shad, white perch, yellow perch and sea lamprey) were observed as adults in spawning condition. In the case of sea lamprey, a single spawning adult was caught directly below the dam in 2011 and the rest were ammocoetes. The sea lamprey ammocoetes as well as the striped bass were assumed to have been residing in the river and not part of any spawning migration or activity.

Figure 5.2: Adult Sea Lamprey collected while electrofishing below Bloede Dam during Spring 2011.

Abundance of blueback herring, hickory shad, white perch, and yellow perch varied both by year and sampling location (Appendix 5.1: Table 5.2). Yellow perch was the only species collected in higher abundance in 2012 than in 2011, while all others declined. All four species were collected directly below Bloede Dam, in abundances lower in 2012 than 2011, with abundance of blueback herring declining dramatically (Fig. 5.2). None of these four species were collected upstream of the dam.

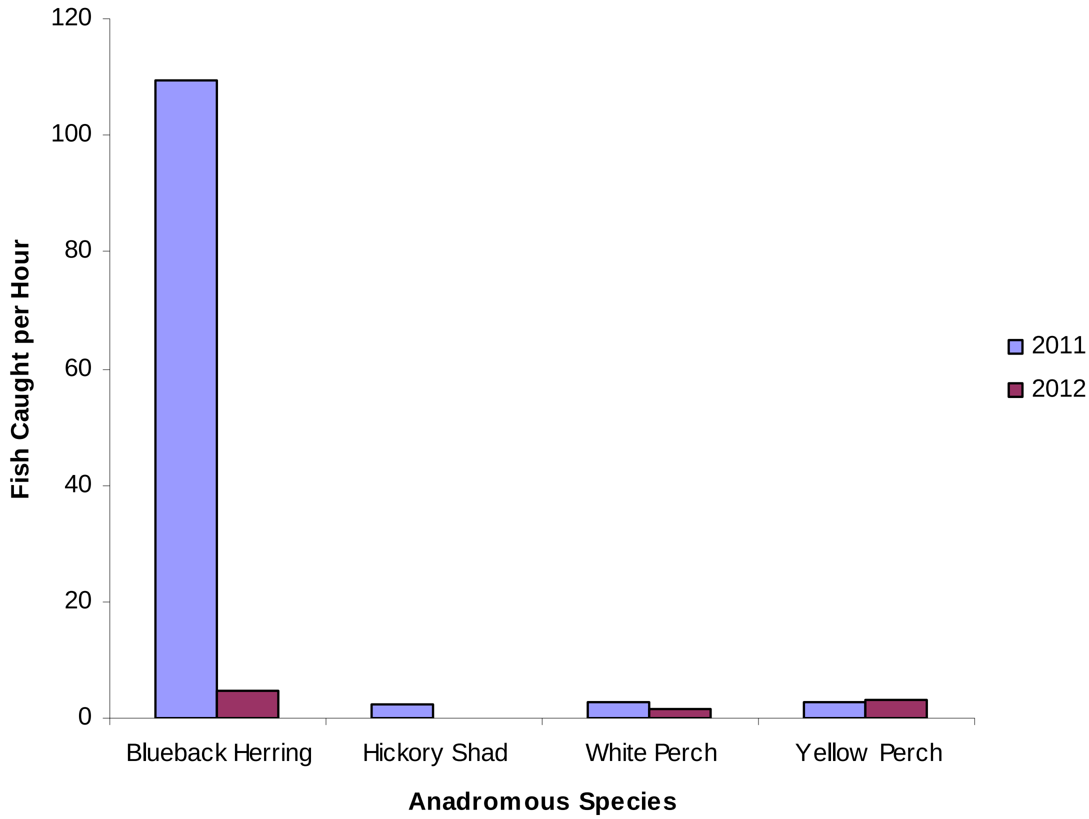


Fig. 5.3: Abundance (in CPUE) of anadromous fish captured by electrofishing in the Bloede Dam tailrace, March-May for 0.91 hours in 2011 and 1.88 hours 2012.

Discussion

Monitoring of anadromous fishes in the Patapsco River demonstrated that Bloede Dam remains a significant blockage to migratory species. Of the six species collected in 2011 and 2012, only sea lamprey was collected both above and below the dam. Results from our spring 2012 monitoring of anadromous fish around Bloede Dam also showed a dramatic decrease in abundance of blueback herring compared to 2011, while the other anadromous species collected at this site exhibited low and similar numbers during both years. For example, we collected blueback herring below Bloede Dam at a rate of 127/hour in 2011, but in 2012, we encountered them at a rate of just 5/hour. When we began seeing lower numbers of herring during 2012, we suspected that we were sampling too early in the spawning run, and would intercept the bulk of the herring spawning migration later in the spring (Harbold 2012). As the sampling period progressed, this never occurred and we continued to collect only small numbers of herring each time we sampled below the dam. Water levels were low while we sampled the three downstream sites throughout spring 2012, and we observed a large amount of sand filling in downstream habitat following the removal of Simkins Dam. While we suspected that the low spring flows and increased sedimentation might have limited the upstream movement of blueback herring (and likely other species) and resulted in the low numbers collected at the base of Bloede Dam, the reduced numbers of blueback herring in 2012 were apparently a region-wide trend.

According to the Maryland Department of Natural Resources Fisheries Service Juvenile Finfish Index, 2011 was the 15th highest year for recruitment of blueback herring in the in the upper Chesapeake Bay since records were first kept in 1959, with a geometric mean of 5.16 fish per seine haul. 2012, on the other hand, was the 11th lowest, with a geometric mean of just 0.13 fish per seine haul (Durell and Weedon 2012). Considering this, it is fair to conclude that observed declines in blueback herring numbers on the Patapsco River in 2012 were largely unrelated to dam removals, and were driven instead by wider-ranging factors affecting the entire Chesapeake Bay region.

We collected more yellow perch in 2012 than in 2011. This is likely due to the timing of our sampling. Yellow perch are one of the first migrants to enter the Patapsco River for spawning each spring (O'Dell et al. 1975). Average yellow perch abundance across the three downstream-most sites during the entire sampling season in 2011 was 2.2/hour, while the average abundance for the same sites sampled during the first quarter of 2012 (roughly one month earlier than monitoring in 2011) was 16.8/hour. By sampling earlier in 2012, we likely intercepted a spawning run of yellow perch that was largely complete by the time sampling began in 2011.

We did not collect anadromous fishes in the fyke net in either 2011 or 2012. While we collected no fish of any kind in the fyke net in 2011, we did collect resident species in 2012. In the second quarter of 2012, thirteen species of resident fish were collected in the fyke net after exiting the fish ladder and successfully bypassing Bloede Dam. There are records of American shad, sea lamprey, and blueback herring using the fish ladder soon after its installation at Bloede Dam in 1992 (MDNR unpublished data). Data we collected in 2011 suggested that the fish ladder was no longer functioning as it was originally designed. During the winter of 2011, repairs were made to the fish ladder. Debris was removed and several baffles (which block the flow and give fish places to rest in the current) were replaced. These actions seem to have improved the ladder's functionality to some extent, as our fyke net data from 2012 showed use of the ladder by at least 13 resident fish species.

What is not known is why these resident fishes used the ladder while anadromous fishes did not.

Some attribute of the fish ladder may make it unfavorable to anadromous fish species but not residents. Conversely, the apparent low to no use of the fish ladder by anadromous species may also be due to the low numbers of individuals traveling up the river and making it to the dam. While we may simply have missed a brief run due to sampling timing, in 2012 we found only four of the six species of anadromous fishes in the Bloede Dam tailrace that we collected by electrofishing the same areas in 2011. Those species that we did encounter were also less abundant than they were in 2011, blueback herring dramatically so. With so many fewer fish in the tailrace, there was certainly a lower potential for them to use the fish ladder. Despite the recent repairs made to the Bloede Dam fish ladder, two additional baffles could not be replaced and would require major effort to repair.

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Chapter 6: American Eel

Introduction

American eels (hereafter eels) are of special interest in the Patapsco River. They are an important commercial species and also an essential part of the aquatic ecosystem. Eels are a catadromous migratory fish, spending their adult lives in freshwater before returning to the ocean to spawn. Recently, eel abundance has declined throughout their range due to a variety of factors including habitat loss, pollution, overfishing, and migration barriers (Haro et. al 2000). Although pollution and habitat loss are certainly a concern for eels in the Patapsco River, we suspect that migration barriers, specifically those formed by four dams, are the most significant problem facing eels in this river.

Many studies have examined the distribution and demographics of eels in river systems with respect to their position in the watershed and the presence of dams or other barriers. Most of these studies indicate that, with increasing distance upstream, the numbers of eels (abundance, density, or catch-per-unit-effort) decrease (Smogor et al. 1995, Oliveira 1997, Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007). Where dams are present, eel numbers may be lower upstream of each successive barrier (Machut et al. 2007) and may appear inflated directly below each barrier as migrating eels are apparently concentrated by the blockage (Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007). Eel total length tends to increase with increasing distance upstream (Smogor et al. 1995, Krueger and Oliviera 1999, Goodwin and Angermeier 2003, Cairns et al. 2004, Machut et al. 2007) to a point where large adult eels may be the only individuals collected in the far upstream reaches of certain watersheds (Smogor et al. 1995, Goodwin and Angermeier 2003).

Restricting access of eels to the upper reaches of watersheds may have detrimental impacts on the entire population. Eel gender is likely environmentally determined, with juvenile individuals in high densities becoming predominantly male and juvenile eels in low densities becoming mostly female (Krueger and Oliviera 1999). Indeed, Machut et al. (2007) noticed that 73% of all the male eels collected in a Hudson River tributary were encountered in crowded conditions below a dam, while 76% of all females were collected in areas of lower eel densities upstream. Barriers to eel migration may not only create fewer females, but less fit females as well. Fecundity in eels is positively correlated with female size (Barbin and McCleave 1997), and eels tend to grow larger farther upstream in watersheds where tributaries support higher growth rates with better habitat and more abundant invertebrate food sources (Machut et al. 2007). By restricting upstream movement of eels, dams may create fewer, less fecund females, adding decreased reproductive capacity to an already growing list of threats to eel populations.

Recent examination of eels in the Rappahannock River, Virginia following the removal of Embrey Dam in 2004 suggests these impacts may be reduced when a barrier is removed. Hitt et al. (2012) looked at the response of eel abundance and size from 2004 through 2010 in streams up to 150 river kilometers (Rkm) upstream of the former dam. Over that time period, the numbers of eels increased while the size (minimum total length) of eels in the study area decreased, largely due to the immigration of smaller (<300 mm) eels into the sample area. If Embrey Dam had been slowing the migration of eels into streams in the study area, its removal as a barrier allowed an influx of new, smaller eels into tributaries far upstream and opened better habitats to a larger portion of the reproductive population.

With the removal of Simkins Dam in 2010, we have the opportunity to look for similar changes to the eel population in the Patapsco River. Although the spatial scale is smaller than in many of the previous studies (the Patapsco sampling locations are distributed over 27 Rkm, beginning 11.4 Rkm from the confluence with Baltimore Harbor), we expect the patterns in eel size and distribution to be similar to those seen in previous studies of North American rivers, impounded and otherwise.

In general, in the Patapsco River, we expect that:

1. Eel numbers will decrease with increasing distance upstream.
2. As numbers of eels decrease with increasing distance upstream, the average size of eels at a given site will increase.
3. Eel numbers may be inflated below dams due to their concentration behind the obstacle.

Now that Union and Simkins dams have been removed, we expect that the eel population will respond by:

4. Increasing in abundance above the former dam site and decreasing below as eel concentrations are released to the upper portions of the watershed.
5. Changing in abundance at sites around the former Simkins Dam outside the range of changes observed at non-impacted reference sites elsewhere in Maryland during the same period.
6. Decreasing in average size at sites above the former Simkins Dam as more smaller eels are able to move into the upper reaches of the watershed.

Methods

We collected eels annually in the Patapsco River during the summers from 2009 to 2012 following protocols described in Ciccotto et al. (2009) at 21 sites. At eleven sites eels were collected using standard two pass electrofishing in 75 meter reaches with block nets set at the upstream and downstream ends of each reach. Eel abundance, time spent electrofishing, and aggregate biomass were recorded at these sites. At the remaining ten sites, eels were collected by electrofishing the best available habitat for a minimum of 600 seconds, recording both abundance and time spent electrofishing.

The catch-per-unit-effort (CPUE) of eels (hereafter "abundance") was calculated using the time spent electrofishing and the total number of eels collected at each site. At the ten sites where aggregate biomass was measured, biomass was divided by the total number of eels collected to provide an estimate of the average individual body weight of eels at a given site, used in this report as a surrogate for eel size. We repeated these estimates of size and abundance using data from each of the four years. With these estimates, we calculated mean eel size and abundance for each site during the entire four year study period (two years prior to Simkins Dam removal and two years post-removal).

We calculated the difference between the post-removal and pre-removal means to determine changes in eel abundance and size at each site after Simkins Dam was removed. Changes in eel abundance observed in the Patapsco River were compared to data from

Maryland Biological Stream Survey sentinel sites (Prochaska et al. 2005). We used data from the 13 sentinel sites throughout Maryland where eels occur to determine the difference in eel abundance at those sites between the pre- (2009-2010) and post-dam removal (2011-2012) periods. Eels are collected at these sites using the same standard two pass electrofishing methods utilized at 11 of the 21 Patapsco sites. We calculated the mean of those differences to investigate the change in eel abundance outside of the Patapsco River. This represented a baseline reference condition for eel abundance in Maryland streams without dams or dam removals. Eels were not weighed separately from resident stream fish at sentinel sites, so there was no reference for eel size.

The mean eel abundance and size, as well as the differences in size and abundance following Simkins Dam removal for each site were plotted with respect to the site's distance in river kilometers (Rkm) from the Patapsco River mouth (confluence with Baltimore Harbor). We used the resulting graphs to look for trends in eel abundance and size throughout the river, as well as changes that occurred following dam removal.

Results

We collected eels at 20 of 21 sites during 2009 and 2010, and at all 21 sites in 2011 and 2012. Eels were absent only from Thistle Run, a tributary that enters the river immediately upstream of Simkins Dam, during 2009 and 2010. In 2011, the first sampling season after Simkins Dam was removed, eels were encountered at this site for the first time and persisted there through 2012.

In general, the abundance of eels at Patapsco River monitoring sites decreased with increasing distance upstream (Fig. 6.1), while eel size (represented by the average individual body weight of eels at each site) increased with increasing distance upstream (Fig. 6.2).

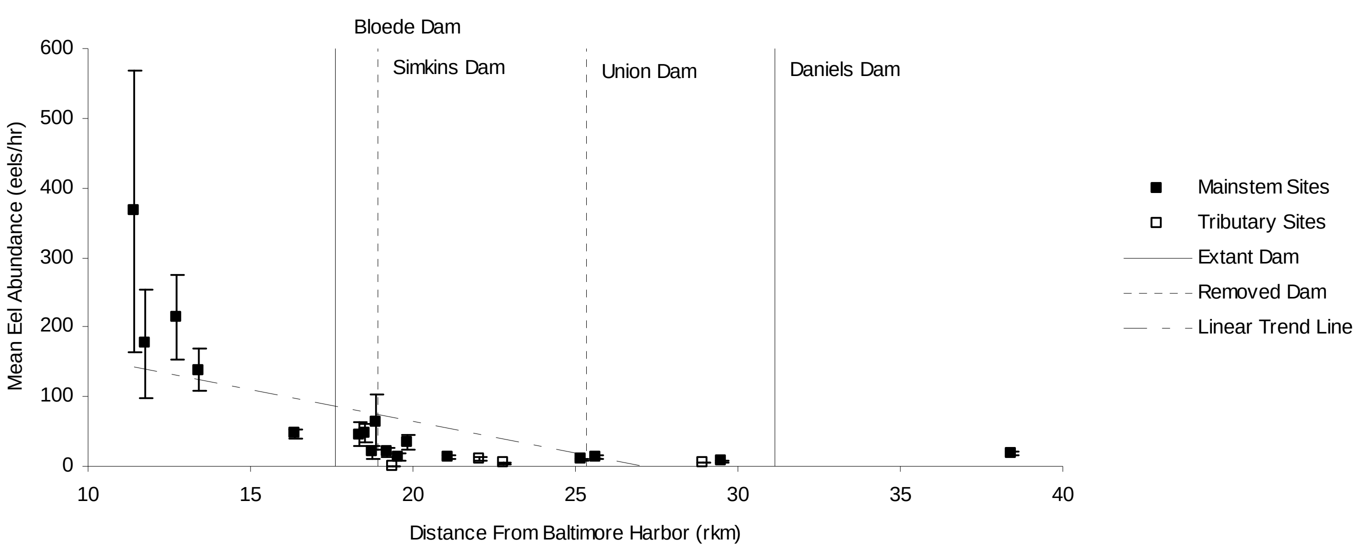

Fig. 6.1: Mean abundance (± 1 SE) of American eels at Patapsco River monitoring sites, 2009-2012.

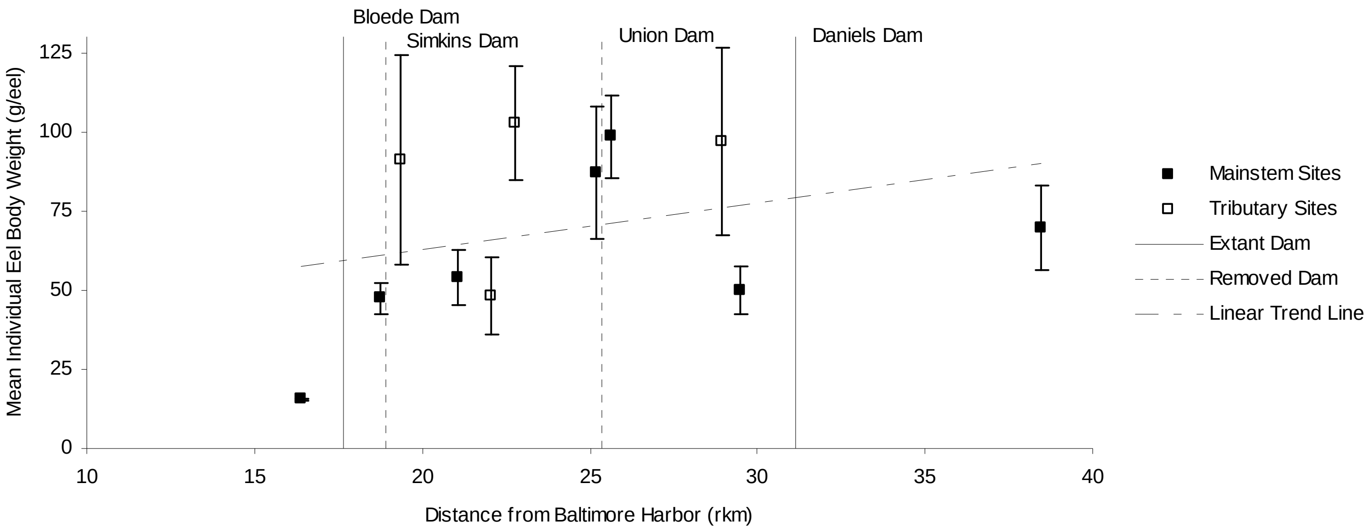

Fig. 6.2: Mean size (± 1 SE) of American eels captured at Patapsco River monitoring sites, 2009-2012.

While the highest eel abundances occurred at sites downstream of Bloede Dam, we also observed an apparent concentration of eels below Simkins Dam before it was removed. Eel abundance in 2009-2010 was higher at the site directly downstream of the Simkins Dam than at any of the subsequent sites between it and Bloede Dam downstream (Fig. 6.3).

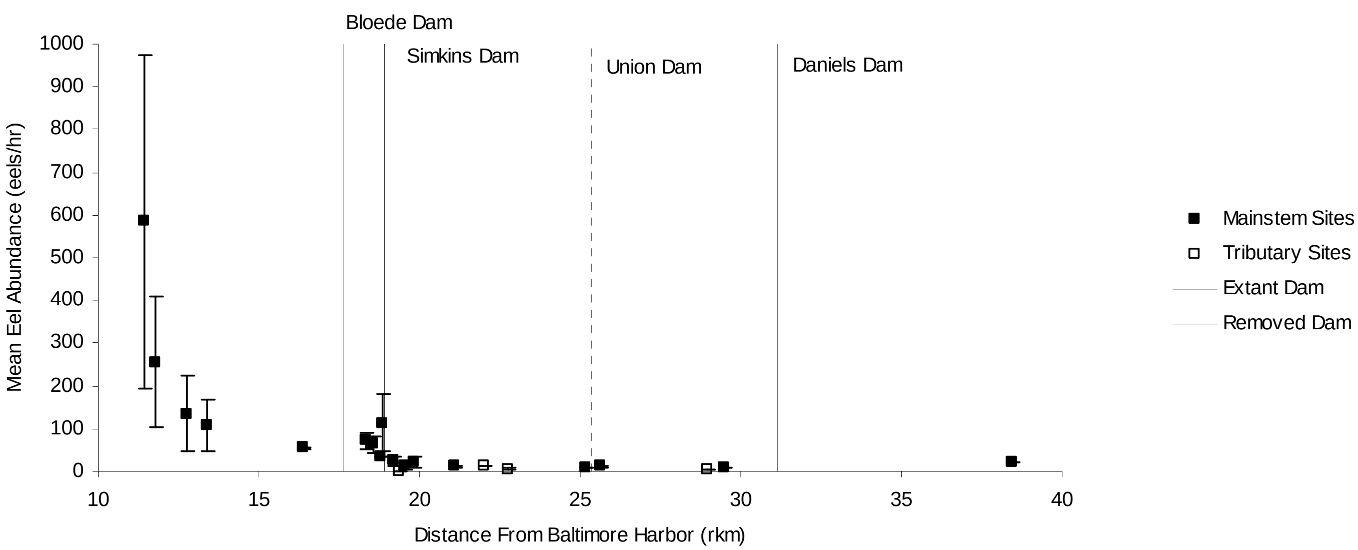

Fig. 6.3: Mean abundance (± 1 SE) of American eels at Patapsco River monitoring sites, 2009-2010.

Following the removal of Simkins Dam in 2010, eel abundance was somewhat constant at sites upstream of the former dam location, but we saw dramatic shifts (both positive and negative) at sites downstream of the dam (Fig. 6.4).

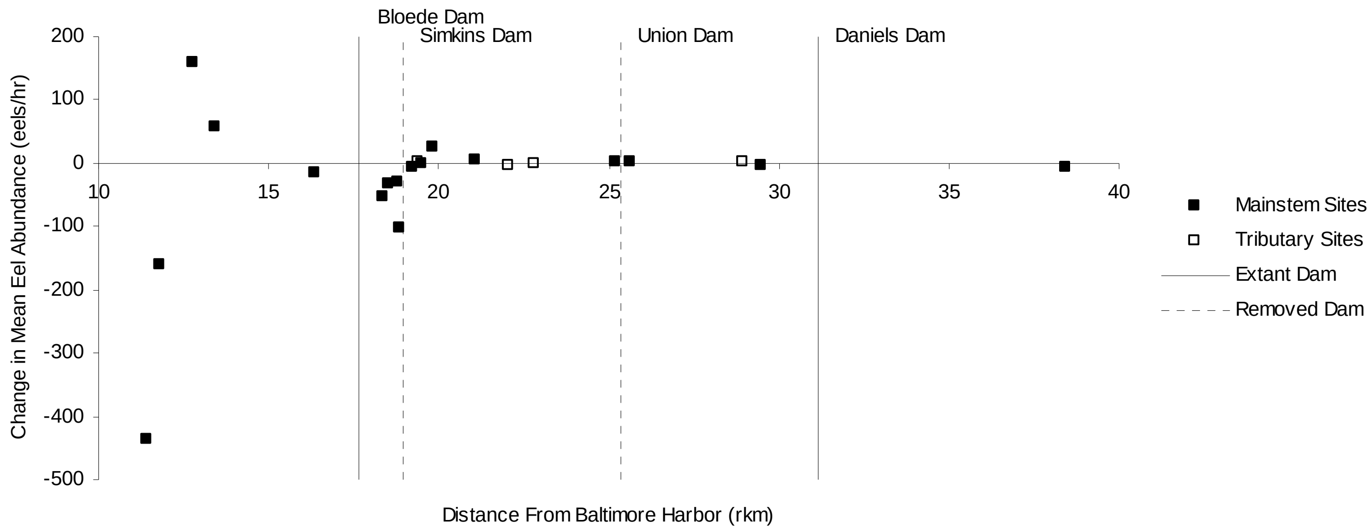

Figure 6.4: Change in mean American eel abundance at Patapsco River monitoring sites following the removal of Simkins Dam.

The average change in eel abundance across all 21 sites was -28.3 eels/hour, much greater than the 1.1 eels/hour change observed state-wide at MBSS sentinel sites. The change in eel abundance at MBSS sentinel sites is not visible when shown at the same scale at the Patapsco data, and as such has not been displayed.

At sites upstream of Simkins Dam, eel size decreased at six locations following dam removal, while it increased at only three. Eel size increased at the site between Simkins and Bloede Dams, and stayed essentially the same at the one site downstream of Bloede Dam (Fig. 6.5).

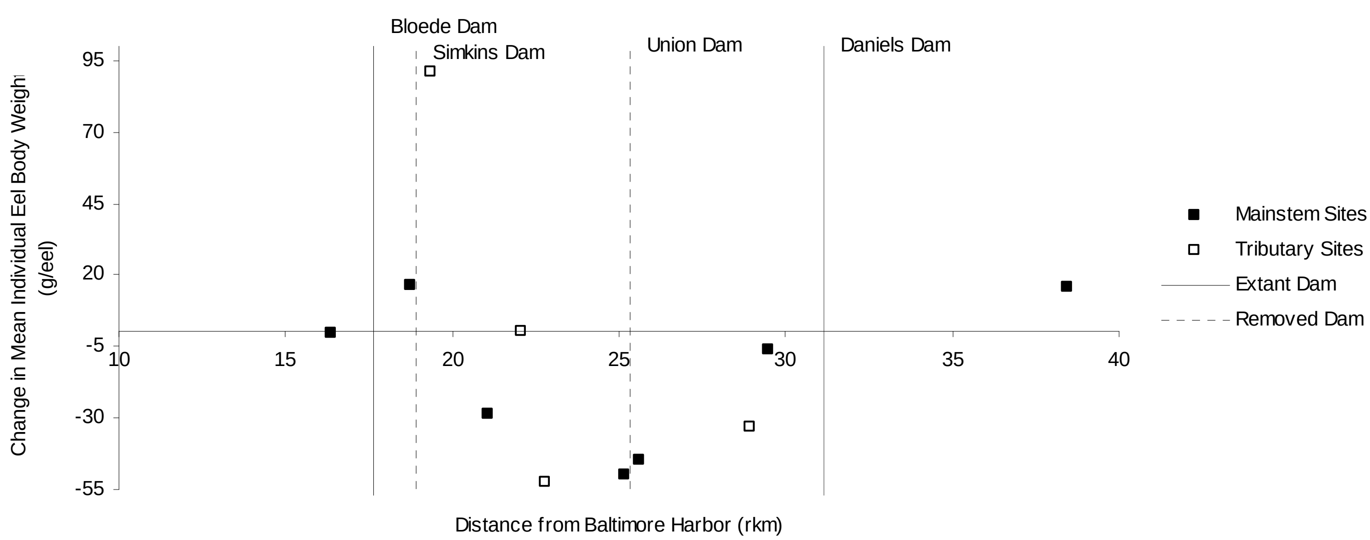

Fig. 6.5: Change in mean size of American eels at Patapsco River monitoring sites following the removal of Simkins Dam.

Discussion

Our findings on eel abundance and size in the Patapsco River irrespective of dam removal were consistent with those from other studies. Combining four years of data, eel abundance decreased and eel size increased with increasing distance upstream. Abundance was highest downstream of Bloede Dam. The ways in which these abundance and size patterns changed after Simkins Dam was removed are more difficult to interpret, due largely to the confounding influences of habitat change, sampling limitations, and the presence of additional migration barriers.

While we observed higher eel abundances at the site directly below Simkins Dam (pre-removal) compared to other sites between it and the next barrier downstream, this was the only dam on the Patapsco River where this pattern was evident. It is not clear why similar patterns were not observed at other Patapsco dams. Bloede and Daniels dams are equipped with Denil fish ladders, as was Simkins Dam before it was removed. While monitoring anadromous fish at Bloede Dam, we have seen that these types of structures can be limited in their effectiveness when damaged or allowed to fall into disrepair (see Chapter 5). It is possible that damage or neglect rendered the Simkins Dam ladder less passable for eels, in turn making Simkins Dam a more substantial barrier than either Bloede or Daniels dams, and offering one explanation for the higher eel abundance observed directly below Simkins Dam.

Eel abundance decreased directly below Simkins Dam following removal, but we did not observe a corresponding increase in abundance upstream. We assumed that if eels were concentrated below the dam as several previous studies observed (Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007), removing the blockage would allow a release of these eels upstream. Abundance should increase upstream as individuals dispersed into new habitats, and decrease in the areas below the dam from where these eels emigrated. Since this was not observed, another scenario may have occurred.

Changes in eel abundance around Simkins Dam may be a reflection of habitat change rather than of dispersal following the removal of a barrier. Dam removals are known to at least temporarily disturb habitats, specifically decreasing habitat for benthic and/or riffle dependent species in downstream areas via deposition of sediment (Bushaw-Newton et al. 2002, Maloney et al. 2008). In a study of Hudson River tributaries, Machut et al. (2007) found that benthic habitats with abundant interstitial spaces are preferred by eels. These habitats were covered after Simkins Dam was removed and a large amount of sand was released to downstream areas. Field observations confirm this. There was a change from a cobble and gravel substrate (preferred habitat) between Simkins and Bloede Dams pre-removal (2009-2010) to an all sand habitat post-removal (2011-2012) (G. Boardman, unpublished data). Given these changes, we can hypothesize that the decreases in eel abundance observed below Simkins Dam post-removal likely had more to do with temporary habitat disturbance rather than eel movements into new areas upstream of the dam.

In most previous studies of eel size, eels (or at least a subset of those collected) were measured individually (Oliviera 1997, Goodwin and Angermeier 2003, Machut et al. 2007, Hitt et al. 2012). We lacked the time and funding for this level of detail, and instead estimated eel size by averaging individual biomass from the aggregate eel biomass collected at each site. We predicted that when the dam was removed, the average eel size at sites above the former Simkins Dam would decrease as more and smaller eels were able to move into the upper reaches of the watershed. While we did observe a decrease in the size of eels above Simkins Dam post-removal, our sampling methods preclude firm conclusions. Knowing only the average size of eels at a given site makes it impossible to ascertain whether the observed decrease in size was due to the addition of numerous small individuals or the loss of a few large individuals in a given year. The only way to assure that observed eel size changes are due to more, smaller eels making it to sites above Simkins Dam after the removal would be to measure each eel individually and look at the distribution of sizes at each site over time, something we can do with additional funding and a continuation of our study.

Based on our results thus far, we speculate that changes we observed in the Patapsco River eel population so far are more the result of habitat disturbance caused by removing Simkins Dam than by restoration and expansion of eel habitat by removing a migration barrier. The decrease in eel abundance observed below the dam is likely due to temporary loss of preferred habitat caused by sediment movement- not emigration out of crowded habitats below the dam. The smaller average eel size at many sites above the former dam may be due to more small eels being able to access sites upstream, but the current monitoring protocols make this hypothesis impossible to verify. It is probably too early to see anticipated changes in eel abundance and distribution in the Patapsco River in response to the removal of Simkins Dam. Monitoring of eels following the removal of Embery Dam took six years to see changes (Hitt et al. 2012), while we have had only two years of monitoring on the Patapsco after Simkins Dam was removed. Also, and perhaps most importantly, the removal of all barriers to eel passage on the Patapsco River is far from complete.

While removing Simkins Dam has restored eel access to over 12 kilometers of river channel and many more within tributaries between it and the next barrier, Bloede Dam is still present just over one kilometer downstream. Bloede Dam is more than twice the height of Simkins Dam, and despite the presence of a fish ladder, is a significant obstacle for migrating eels. Indeed, the largest drop in eel abundances throughout our sampled reach was seen between the sites downstream of Bloede Dam and those between Bloede Dam and the former Simkins Dam. Until Bloede Dam is removed, we may not see any significant changes in eel abundance or size in the Patapsco River. Instead, for at least two or more years, we are likely to continue to see the effects of habitat disturbance as the river around Simkins Dam returns to a free-flowing state and impounded sediments move downstream.

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Chapter 7: The effects of Simkins Dam removal on resident fish assemblages of the Patapsco River

Introduction

The removal of small dams because of public safety, flooding, and wildlife concerns has increased over the past 30 years (Maloney et al. 2008), with over 450 dams removed since 1980 in the U.S. alone (American Rivers et al. 1999). In general, dam removal is viewed as beneficial to riverine ecosystems, restoring once impounded habitats to more natural, free-flowing systems. Despite the growing use of dam removal, very little research has been conducted to measure the ecological benefits associated with this restoration process. To date, arguments in support of dam removal have hinged on the benefits to anadromous species (American Rivers et al. 1999, Doyle et al. 2005). Few studies have examined the effects of dam removal on other components of the river ecosystem, including resident fish populations and assemblages (Doyle et al. 2005). Studies that focused on the monitoring of resident fish assemblages have documented both positive and negative changes resulting from dam removal.

The removal of a dam can cause dramatic changes to flow patterns, water temperature, channel geomorphology, riparian vegetation, substrate composition, and other physical and chemical properties of a river (Bednarek 2001, Doyle et al. 2003, 2005). These effects are most pronounced in adjacent reaches, areas immediately upstream and downstream, and decrease with distance from the dam (Doyle et al. 2005). Sediments previously stored in impounded areas erode, become mobilized, and are transported to downstream areas following dam removal. In general, this can have positive effects on upstream fish assemblages. Increased substrate size in previously-impounded reaches leads to improved fish cover and habitat quality (Kanehl et al. 1997). Fish assemblages change from those comprised of species more common in lakes and reservoirs to assemblages more characteristic of free-flowing rivers (Bushaw-Newton et al. 2002) as lotic fish species recolonize from adjacent reaches (Catalano et al. 2007, Gardner et al. 2011). Positive effects on upstream fish assemblages resulting from dam removal include increased biological integrity, species richness, abundance of native fishes, and abundance of game fishes (e.g., smallmouth bass) as impounded reaches revert to free-flowing areas (Kanehl et al. 1997, Catalano et al. 2007, Maloney et al. 2008).

Conversely, dam removal can have negative effects on downstream fish assemblages. Aggradation of sediment in downstream reaches can damage fish spawning habitats, reduce fish cover and prey availability (Bednarek 2001). Fish species that require clean, coarse substrate for spawning and feeding (i.e., lithophilic spawners, benthic riffle species) decline as a result (Bushaw-Newton et al. 2002; Maloney et al. 2008). Age and size-structure of important game fish populations can change directly downstream of dams following removal; with younger year classes declining in abundance (Doeg and Koehn 1994, Kanehl et al. 1997). Similarly, dam removal can decrease biological integrity, species richness, fish density and biomass in areas directly downstream of the removed dam (Kanehl et al. 1997, Catalano et al. 2007, Maloney et al. 2008). These adverse effects on downstream fish assemblages have been shown to be temporary (Doyle et al 2005). Species re-colonize habitats and fish assemblages tend to recover once geomorphic conditions stabilize and sediment moves through the river system. This recovery can occur within one to five years (Doyle et al. 2005; Catalano et al. 2007)

The removal of Simkins Dam on the Patapsco River, and subsequent changes observed in fish habitat quality and substrate composition (see Chapter 3), will likely cause changes to various aspects of the fish assemblages in upstream and downstream reaches. To document these changes, the Maryland Biological Stream Survey (MBSS) quantitatively surveyed stream fishes at seven sites in the Patapsco River for two years prior to and two years following the removal of Simkins Dam. In this chapter, we utilized these data to examine the response of fish assemblages to dam removal.

Specifically, we tested five hypotheses:

1. Fish Assemblage Stability: Environmental disturbances can affect stream fish assemblages by altering species composition and abundance (Freeman et al. 1988, Poff and Allan 1995, Grossman et al. 1998). Geomorphic and hydrologic changes associated with the removal of Simkins Dam will likely alter species composition and abundance. As a result, we expect fish assemblages at sites adjacent to Simkins Dam to be less stable in species composition and abundance over the four-year study period, in comparison to sites unaffected by dam removal.
2. Species Richness, Fish Density, and Biomass: Dam removal will likely alter the distribution of some fish species, with some displaced entirely from areas where they occurred prior to removal. Similarly, improved connectivity (i.e., more fish movement) among reaches following dam removal will likely increase dispersal of some species into areas where they were previously absent. Changes in fish distributions will be reflected in species richness (number of species) in upstream and downstream reaches following dam removal. Fish density (abundance/m^2) and biomass (g/m^2) in upstream reaches will likely increase following dam removal, while decreases in density and biomass will occur in downstream reaches where sedimentation is highest (Bushaw-Newton et al. 2002, Doyle et al. 2005, Catalano et al. 2007, Maloney et al. 2008, Gardner et al. 2012).
3. Biological Integrity and Ecological Composition: Biological integrity, as defined by Frey (1977), refers to the "capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of the region". In Maryland, biological integrity of a stream fish assemblage is measured using an Index of Biological Integrity (IBI) developed by Southerland et al. (2008). The IBI is an indicator of fish assemblage condition that integrates community attributes including fish abundance, species composition, species pollution tolerance, trophic composition, and reproductive function (Karr et al. 1986). By design, the IBI declines in response to environmental degradation and disturbance, reflecting fish communities altered by anthropogenic impacts (Karr et al. 1986, Lyons 1992). As observed in previous dam removal studies, we expect that the biological integrity of the fish assemblages will increase in upstream reaches after the removal of Simkins Dam due to improved fish habitat quality and stream connectivity (Catalano et al. 2007). Biological integrity is likely to decline in downstream areas due to shifts in species composition associated with sedimentation (Catalano et al. 2007, Maloney et al. 2008).
   Upstream erosion and downstream sedimentation will cause observable changes in the abundance of certain stream fish species altering the composition of

ecological guilds (hereafter referred as "ecological composition"). The abundance of species most sensitive to sedimentation and changes in substrate composition (e.g., lithophilic spawners, benthic riffle species) will likely decline as a result of dam removal(Poff and Allan 1995, Bushaw-Newton et al. 2002, Maloney et al. 2008). Similarly, the numbers of pollution-tolerant and intolerant species are likely to change in response to habitat disturbance associated with dam removal (Karr et al. 1986). The density of non-native fish species is also likely to decrease in affected reaches (Kanehl et al. 1997, Bushaw-Newton et al. 2002).

4. Fish Assemblage Similarity: Dams are known to reduce connectivity of fish populations resulting in loss of some species from upstream or downstream reaches and increased dissimilarity in species composition over time (Lienesch et al. 2000, Morita and Yamamoto 2002, Dodd et al. 2003). Dam removal restores connectivity within a river system, improving fish dispersal and leading to greater similarity among assemblages in adjacent reaches over time (Gardner et al. 2011). Fish assemblages at sites upstream and downstream of Simkins Dam should increase in similarity following dam removal as fishes re-colonize and geomorphic disturbance stabilizes over time.
5. Game Fish Populations: The removal of Simkins Dam is likely to alter the abundance and size structure of smallmouth bass in affected areas. Habitat quality for smallmouth bass is likely to improve in upstream impounded reaches following dam removal (Kanehl et al. 1997). Geomorphic and hydrologic changes in downstream reaches will likely negatively affect bass populations over the short-term, as reflected in reduced abundance and altered size structure (Doeg and Koehn 1994, Kanehl et al. 1997).

Methods

We conducted quantitative surveys of fish assemblages annually at seven sites (Fig. 2.1) in the Patapsco River mainstem from 2009 to 2012. Fish surveys were conducted during the summer (June – September) of each year following protocols described in Appendix A. With the exception of one site, all sites were surveyed twice prior to and twice following dam removal. Survey effort differed at only one site, 511. This site was surveyed only once (in 2010) prior to the removal of Simkins Dam. We compared pre- and post-dam removal fish data collected at each of these sites. To limit the effects of varying sampling effort on the analysis of data collected at site 511 (sampled only once prior to dam removal), we compared pre-dam removal data (i.e., 2010) to data collected from only one year following dam removal. We chose to use data collected in 2012 for comparison because these data reflected the most current conditions at the site at the time of our analyses.

Small sample size (i.e., only two years pre- and two years post-dam removal) precluded the use of robust statistical analyses to assess changes in fish assemblages resulting from dam removal. In place of these analyses, we used a combination of "control" and "treatment" sites to assess 1) natural changes in assemblages occurring throughout the Patapsco River over the study period, and 2) changes in assemblages most likely resulting from dam removal.

We examined changes in pre- (2009-2010) vs. post-dam (2011-2012) removal assemblages at two control sites to assess natural variability in assemblages occurring during the study period. Fish assemblages are affected by natural variability in environmental conditions (Poff and Allan 1995; Grossman et al. 1998). Natural variation in precipitation, river flow, water temperatures, etc. that occurred during our study undoubtedly influenced fish assemblages throughout the Patapsco River basin, including sites adjacent to Simkins Dam. To account for natural variability, we examined changes in assemblages at sites 510 and 511 in the Patapsco River mainstem (Fig. 2.3). We chose to use these sites as control sites in our analyses because 1) these sites were within the Patapsco River basin and, as such, were influenced by the same natural phenomena as sites adjacent to Simkins Dam; 2) these sites were located 10.6 and 19.5 river kilometer (Rkm), respectively, upstream of Simkins Dam and were unaffected by its removal; and 3) these sites were separated by an existing dam (Daniels Dam) and therefore served as ideal sites to compare to treatment sites separated by Simkins Dam. Variability in fish assemblages observed at these control sites during the study period (2009-2012) was assumed to be natural and unrelated to dam removal.

The three treatment sites used in our analyses were close to Simkins Dam, in areas most likely to be affected by dam removal (Doyle et al. 2005). Two treatment sites, 501 and 502, were located downstream of Simkins Dam within areas that experienced sedimentation following its removal (Fig. 2.6). Site 502, located 0.2 Rkm downstream of Simkins Dam experienced a dramatic shift from predominately coarse substrate to fine substrate following dam removal. Site 501, located 2.5 Rkm downstream of Simkins Dam, is the most downstream site surveyed quantitatively for fishes as part of this project. It is also located 1.3 Rkm below Bloede Dam. The presence of Bloede Dam between the two downstream treatment sites may limit our ability to detect changes in resident fish assemblages associated with Simkins Dam removal, especially at site 501, because it likely slowed the downstream movement of sediment originating from the Simkins Dam impoundment. Although we recognize the potential influence of Bloede Dam on our analyses of downstream effects, we chose to include 501 in our analyses because sedimentation did occur at this site, albeit to a lesser extent than what was observed at 502. The third site, 504, located 2.2 Rkm upstream of Simkins dam (Fig. 2.5), was upstream of the Simkins Dam impoundment and above areas that underwent significant erosion following dam removal (see Chapter 3). We compared pre- and post- dam removal fish assemblages at these treatment sites, and examined any observed changes in relation to natural changes observed at control sites. Changes in fish assemblages observed at treatment sites that exceeded in magnitude and/or were opposite the natural changes that were observed at control sites during the same period were attributed to dam removal.

Although the focus of this chapter is on fish assemblage response to dam removal at sites adjacent to Simkins Dam, we also report conditions observed at two additional sites (507 and 508) sampled in the Patapsco River mainstem. Site 507 was located immediately downstream and site 508 was located immediately upstream of the former Union Dam that was removed in 2009.

Fish Assemblage Stability:

We used pre- vs. post-dam removal changes in a Shannon-Weiner (SW) species diversity index as a measure of stability in fish assemblages (Krebs 1989, Gardner et al. 2011). The SW species diversity index is a measure of both the number of species in a community (richness) and the relative abundance of species in that community (evenness). This index can range from 0.0 to 5.0 (Krebs 1989). Disturbed, unstable ecosystems generally have lower index scores. Because the SW index responds to changes in species composition and abundances, we used this index as a measure of stability of assemblages over the study period. We calculated SW index scores for each site and for each year sampled. We calculated mean index scores for all sites prior to and following dam removal. We then calculated the change observed (post – pre) in SW index scores during the study period.

Fish Species Richness, Density, and Biomass:

Species Richness: We compiled a list of fish species collected at each site each year. We compared species lists for pre- and post-removal periods and identified the following: 1) species collected during both pre-and post-removal periods, 2) species collected before dam removal only, and 3) species collected after dam removal only. We examined changes in species richness at sites downstream and upstream of Simkins Dam, using pre- vs. post-dam removal species richness at control sites as a measure of natural change over the study period.

Fish Density and Biomass: Total catch (all species in aggregate) and total area of each site (75m long x mean width) were used to calculate total fish density (abundance/m^2) for all seven mainstem sites for each year sampled. Similarly, we calculated total fish biomass (g/m^2) for each site and for each year sampled. We examined pre- and post-removal changes in total fish density and biomass at sites adjacent to Simkins Dam in relation to changes observed at the two control sites.

Biological Integrity and Ecological Composition:

Biological Integrity: We calculated Fish Index of Biotic Integrity (IBI) scores for each site for each year sampled following Southerland et al. (2008). We calculated mean IBI scores for each site prior to and following dam removal. We then calculated the change observed (post – pre) in mean IBI scores at each site during the study period.

Ecological Composition: We tested five metrics commonly used in biological assessments that have been shown to change in response to dam removal or to stream disturbance in general (Karr 1981, Karr et al. 1986, Bushaw-Newton et al. 2002, Southerland et al. 2007, Maloney et al. 2008). The five metrics tested included: 1) number of benthic riffle species - species that reside on the stream bottom and are associated with riffle habitats and coarse substrate (following Bushaw-Newton et al. 2002); 2) number of lithophilic spawners - species requiring clean, coarse substrates for reproduction (Southerland et al. 2007, Maloney et al. 2008); 3) number of intolerant species – species known to be sensitive to anthropogenic stress (Karr 1981, Karr et al. 1986, Southerland et al. 2007); 4) number of tolerant species – species known to be tolerant to anthropogenic stress (Karr et al. 1986, Southerland et al. 2007); and 5) density of non-native species (Maloney et al. 2008). For number of benthic riffle species, number of lithophilic spawners, number of intolerant species, and number of tolerant species metrics, we summed the number of species from pre- and post-dam removal periods and calculated the net change (post – pre) of species at each site for each metric. For density of non-native species, we summed abundance of all non-native species caught at each site each year and calculated density (abundance/m^2) using total area (75 m x mean wetted width) sampled at each site. We calculated mean density for pre- and post-removal periods, and calculated change (post – pre) for each site.

Fish Assemblage Similarity:

We conducted two separate analyses to examine fish assemblage similarity at sites adjacent to Simkins Dam. For the first similarity analysis, we excluded anadromous and semi-anadromous species. These migratory species are limited in their upstream distribution by Bloede Dam – separating the downstream treatment sites 501 and 502. By excluding these species, we were able to look at the effects of Simkins Dam removal alone. We used Sorensen's Similarity Index (following Krebs 1989, Gardner et al. 2011) to evaluate assemblage similarity in space (i.e., similarity among assemblages adjacent to Simkins Dam) and time (i.e., changes in similarity at these sites from pre- to post-dam removal periods). Sorensen's index scores range from 0.0 for assemblages that are completely dissimilar to 1.0 for assemblages that are identical. For the three treatment sites adjacent to Simkins Dam, we calculated Sorensen's index scores comparing the two downstream sites (501 × 502) and each of these sites to the site upstream of Simkins Dam (504) for each year sampled. We then examined changes in mean similarity index scores from pre- to post-dam removal periods.

Our second similarity analysis was aimed at examining the continued effects of Bloede Dam on fish assemblage composition. For this analysis, we included migratory species and repeated the procedures listed above. For both similarity analyses, we examined concurrent changes in similarity between the two control sites over the study period.

Smallmouth Bass Populations:

We compiled data on smallmouth bass collected from the seven mainstem sites from 2009 to 2012 and examined downstream to upstream patterns in smallmouth bass abundance and size structure. Size classes used in our analysis were 1) Young-of-Year (<90 mm), 2) Stock (180–305 mm, considered "catchable" in size, but not meeting minimum legal size requirement; and 3) Harvestable (>305 mm, meeting state minimum size requirement). The Young-of-Year size class was defined using a length-frequency analysis of Smallmouth Bass data collected as part of the statewide MBSS from 2000 to 2011. Stock and Harvestable classes were defined following Gabelhouse (1984).

We also examined pre- and post-removal changes in bass abundance and size structure at downstream and upstream treatment sites in relation to changes observed at the control sites during the study period.

Results

Fish Assemblage Stability:

Shannon-Weiner index scores were variable at all sites. Index scores ranged from 1.80 (site 504) to 2.69 (site 510) during the study period. Pre- to post-dam removal changes in SW index scores ranged from 0.06 to 0.63. SW index scores increased at all but one site (508) after dam removal. This site experienced a decline in SW scores during the study period. The largest changes in SW scores, reflecting the highest instability in species composition and abundance, were observed at sites immediately downstream and upstream of Simkins Dam (502 and 504, respectively). Changes observed at these sites exceeded changes observed at control sites during the study period (Fig. 7.1).

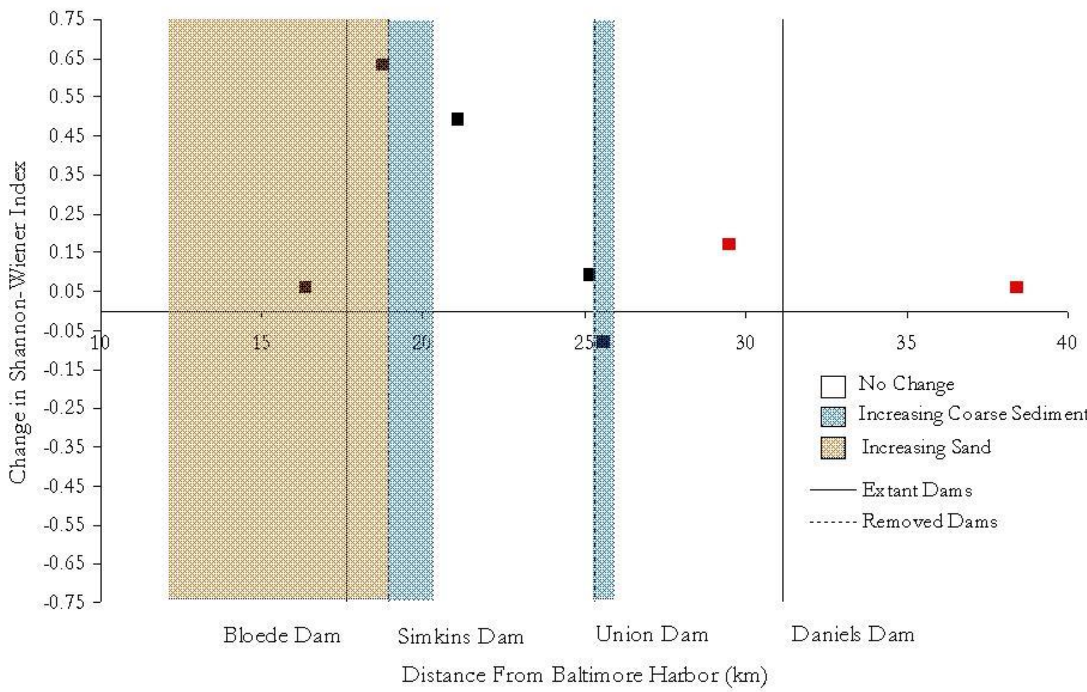

Figure 7.1: Pre- vs. post-dam removal changes in fish assemblages at Patapsco mainstem sites. Black boxes depict sites in vicinity of Simkins and Union dams. Red boxes depict control sites.

Fish Species Richness, Density, and Biomass:

Species Richness: Over the course of this study, we collected 42 species and 18,149 individual fish in the Patapsco River mainstem. Of the seven sites surveyed, the most downstream site 501, had the highest total species richness (35) collected over the four years. Four of these were anadromous and semi-anadromous species (i.e., striped bass, sea lamprey, white perch, yellow perch) found only in portions of the Patapsco River below Bloede Dam. Excluding these migratory species, richness at this site was 31. Twenty-eight and 27 species were collected at site 502 and 504, respectively. Twenty-nine and 27 species were documented at control sites, 510 and 511, respectively, over the study period.

Species richness changes were observed at downstream, upstream and control sites following dam removal (Table 7.1). Downstream sites experienced a greater species loss - species only collected prior to dam removal - than upstream and control sites. Three species (i.e., Blue Ridge sculpin, common carp, and shield darter) and two species (i.e., central stoneroller and shield darter) collected at sites 501 and 502, respectively, prior to dam removal, were not found at these sites after the dam was removed. No species loss was observed at the upstream site (504). One species, blacknose dace, was documented prior to but not following dam removal at the control site, 510. Species additions – species only collected after dam removal - were observed at all downstream, upstream, and control sites (Table 7.1). Following dam removal, seven species were collected at 502, located directly below the dam, that were not observed prior to dam removal.

Table 7.1: Pre-vs. post-dam removal changes in species richness observed at sites downstream and upstream of Simkins Dam and at control sites in the Patapsco River mainstem. Species collected both prior to and following dam removal are not shown.

Site Location	Site	Species collected prior to Dam Removal Only	Species collected following Dam Removal Only
Downstream	501	Blue Ridge sculpin	blacknose dace
		common carp	creek chub
		shield darter	yellow perch
	502	central stoneroller	banded killifish
		shield darter	creek chub
			golden shiner
			green sunfish
			rosyface shiner
			spottail shiner
			yellow bullhead
Upstream	504		banded killifish
			spottail shiner
Control	510	blacknose dace	spottail shiner
	511		blacknose dace
			creek chub
			spottail shiner
			satinfin shiner
			largemouth bass
			bluegill

Fish Density: Total fish density at all Patapsco River sites sampled during this survey ranged from 0.17 to 0.86 fish/m^2. Fish density values were highest at the furthest downstream site, 501, throughout the study period. Following the removal of Simkins Dam, fish density declined at downstream sites from a mean of 0.53 to 0.43 fish/m^2 (Fig. 7.2). Fish density at the upstream site increased from a mean of 0.35 to 0.65 fish/m^2 following dam removal. Changes in fish density observed at downstream and upstream sites exceeded or were opposite to that observed at control sites during the same period. Mean fish density at the downstream control site (510) during the study period increased from 0.32 to 0.55 fish/m^2. Mean fish density at the upstream control site (511) increased from .40 to .41 fish/m^2.

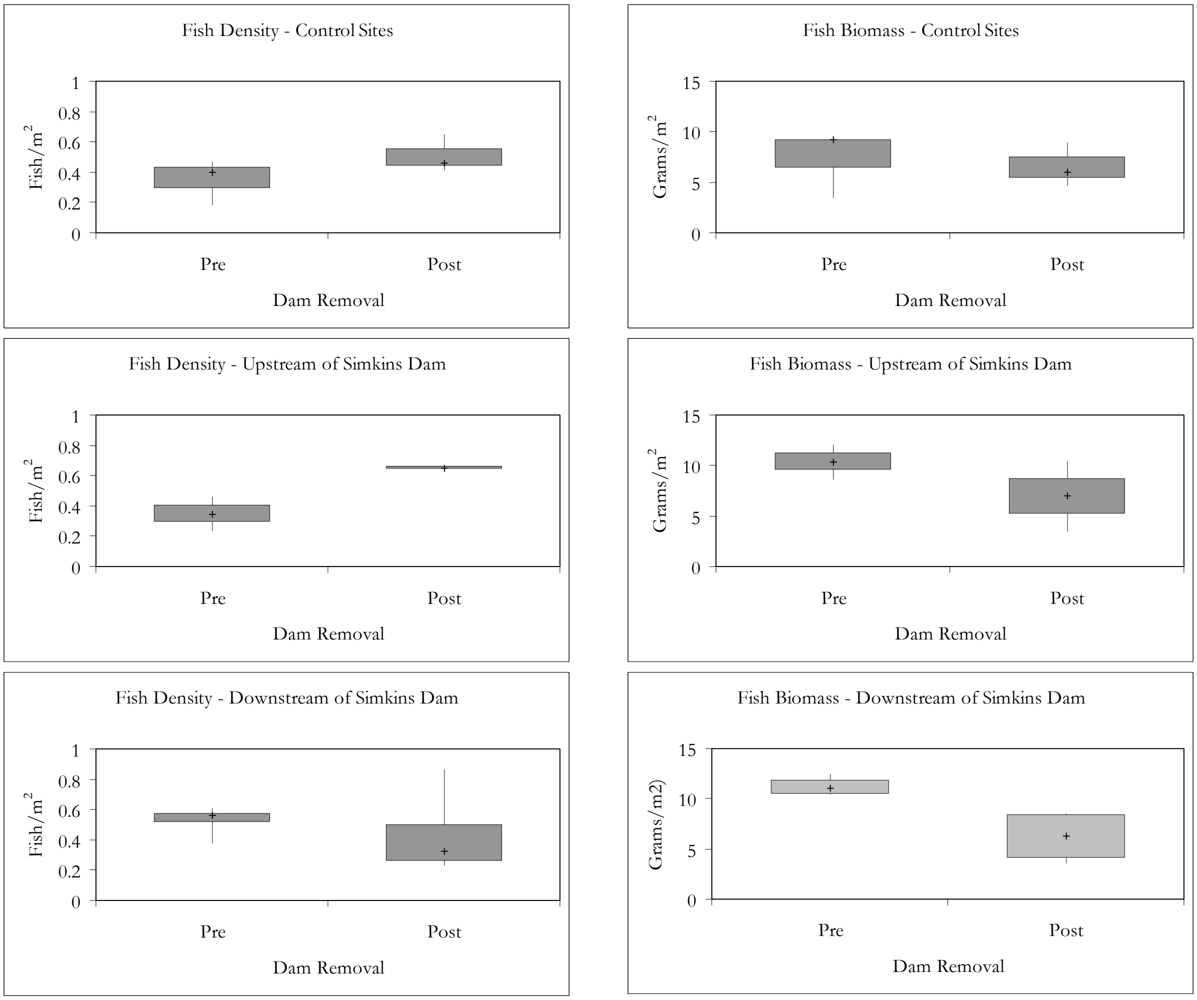

Figure 7.2: Pre- and post-dam removal fish density and biomass at sites upstream of Simkins Dam (N=1), downstream of Simkins Dam (N=2) and control sites (N=2).

Fish Biomass: Total fish biomass at all Patapsco River sites sampled during this survey ranged from 3.44 to 16.72 g/m^2. Fish biomass varied considerably at all sites with no obvious downstream to upstream longitudinal pattern in the river. The highest biomass recorded during the survey was from site 508, upstream of the former Union Dam site.

Following the removal of Simkins Dam, fish biomass declined at downstream sites from a mean of 11.25 to 6.15 g/m^2 (Fig. 7.2). Despite increases in fish density observed at the upstream site, fish biomass declined from a mean of 10.35 to 6.95 g/m^2 following dam removal. Changes in fish biomass observed at downstream and upstream sites exceeded that of control sites during the same period. Mean fish biomass at the downstream control site (510) during the study period decreased from 6.34 to 5.33 g/m^2. Mean fish biomass at the upstream control site (511) decreased from 9.22 to 8.93 g/m^2.

Biological Integrity and Ecological Composition:

Biological Integrity: Fish IBI scores at the seven sites in the Patapsco were quite variable through the course of this study, ranging from Poor (2.0-2.99) to Good (4.0-5.0). Most sites saw a general increase in fish biotic integrity during the study (Fig. 7.3). Following the removal of Simkins Dam, IBI scores at both control sites, 510 and 511 increased by 0.66 and 1.33, respectively. Positive increases in IBI scores were also observed at the site upstream of Simkins Dam (504), and at 502, the site immediately below Simkins Dam. However, increases observed at these sites were not higher in magnitude than natural increases observed at both control sites. One downstream site (501) showed a substantial decrease from 3.66 (Fair) to 2.33 (Poor) following dam removal. This decrease was greater in magnitude than decreases in fish IBI scores observed at sites 507 and 508 near Union Dam and opposite to that observed at the two control sites (Fig. 7.3).

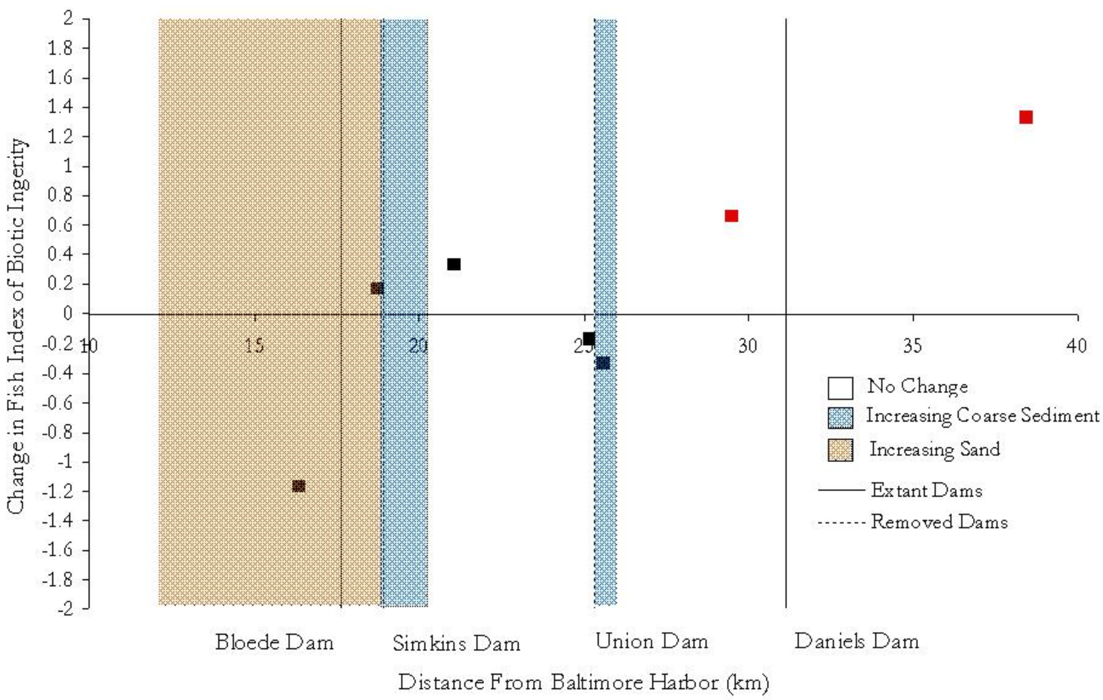

Figure 7.3: Pre- vs. post-removal changes in fish index of biotic integrity at Patapsco mainstem sites. Black boxes depict sites in vicinity of Simkins and Union dams. Red boxes depict Control sites.

Ecological Composition: The number of benthic riffle species declined at downstream sites following dam removal (Fig. 7.4). This change was driven by the loss of shield darter and Blue Ridge sculpin (as reported previously). A decline in benthic riffle species was also observed at 507, the site downstream of the former Union Dam. This metric remained unchanged at the site upstream of Simkins Dam and at both control sites.

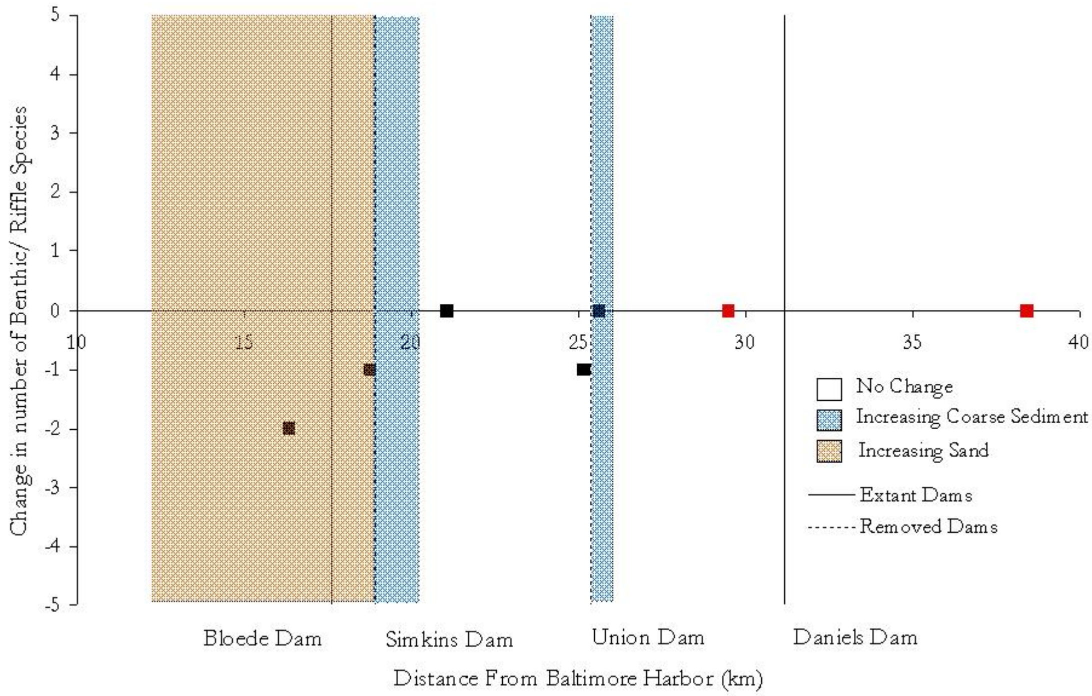

Figure 7.4: Pre- vs. post-removal changes in number of benthic/ riffle fish species at Patapsco mainstem sites. Black boxes depict sites in vicinity of Simkins and Union dams. Red boxes depict Control sites.

Lithophilic spawning species declined after dam removal at only one site, 501, located downstream of Simkins Dam (Fig. 7.5). Changes observed at the other sites were the same or smaller than that observed at control sites and were attributed to natural variability in this metric during the study period.

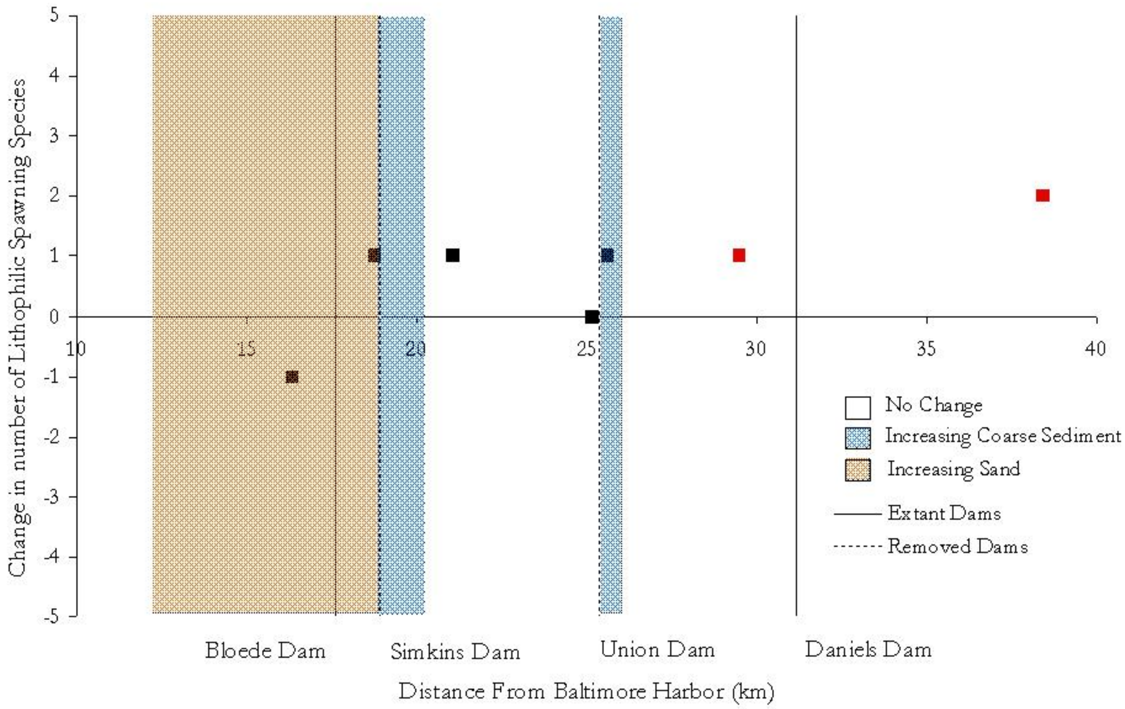

Figure 7.5: Pre- vs. post-removal changes in number of lithophilic spawning fish species at Patapsco mainstem sites. Black boxes depict sites in vicinity of Simkins and Union dams. Red boxes depict Control sites.

We documented a decrease in the number of intolerant species at the two sites downstream of Simkins Dam following its removal (Fig. 7.6). Changes in this metric reflect loss of shield darter and central stoneroller from these downstream sites. The number of intolerant species remained unchanged at the sites upstream of Simkins Dam and adjacent to Union Dam. This metric increased at both control sites during the study period.

Although the number of tolerant species varied at sites adjacent to Simkins Dam, this variability was similar to that observed at control sites and was thus considered to be natural and probably not related to dam removal.

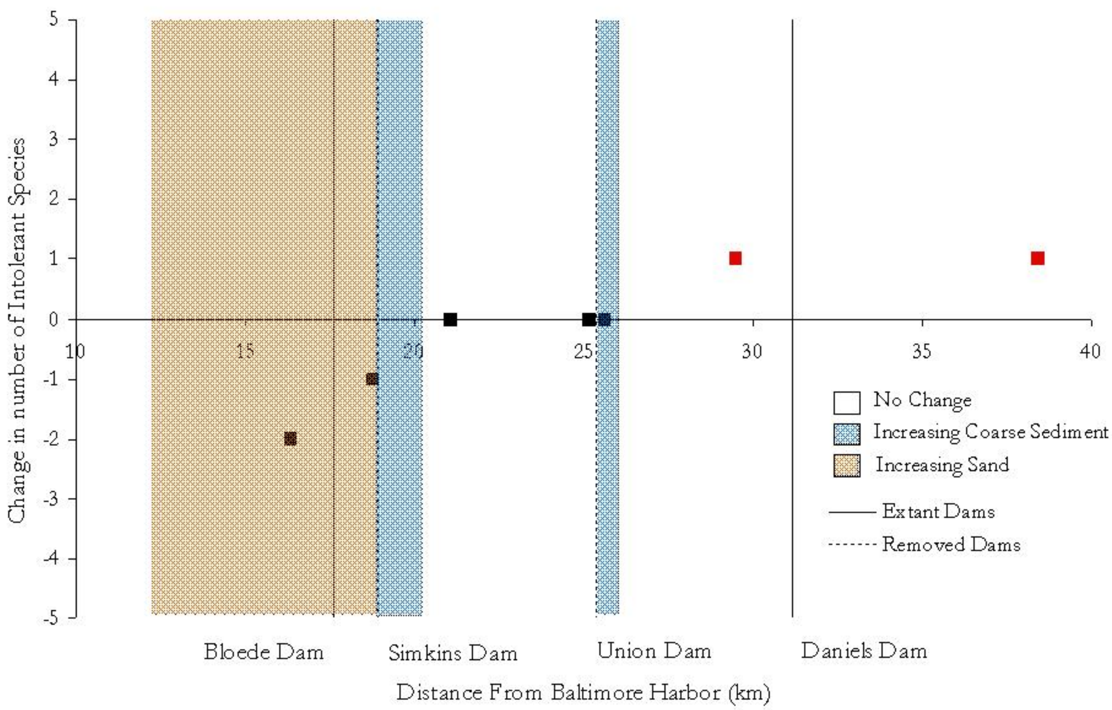

Figure 7.6: Pre- vs. post-removal changes in number of intolerant fish species at Patapsco mainstem sites. Black boxes depict sites in vicinity of Simkins and Union dams. Red boxes depict Control sites.

Densities of non-native species were variable at most sites over the study period (Fig. 7.7). Natural change at control sites ranged from -0.02 to <0.01 individuals/m². Non-native species density declined by 0.025 and 0.030 at sites immediately downstream (502) and upstream (504) of Simkins Dam, respectively. These changes were driven mostly by declines in density of rock bass at 502 and green sunfish and rock bass at 504 following dam removal. Variability observed at all other sites was within the range of natural changes observed at control sites.

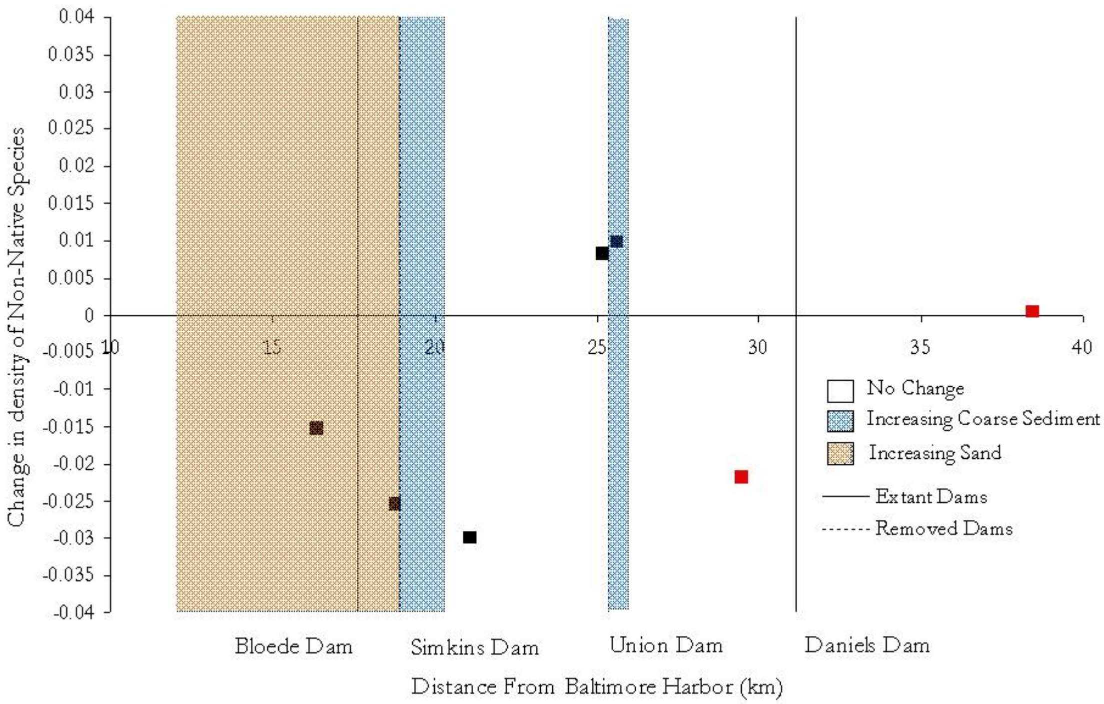

Figure 7.7: Pre- vs. post-removal changes in density of non-native fish species at Patapsco mainstem sites. Black boxes depict sites in vicinity of Simkins and Union dams. Red boxes depict Control sites.

Fish Assemblage Similarity:

When anadromous and semi-anadromous species were excluded from the analysis, assemblage similarity increased following the removal of Simkins Dam (Fig. 7.8). Sites 501 and 502 increased in similarity by 0.09 (9%). Sites 501 and 504 also increased in similarity by 0.09 (9%) following dam removal. Of all three site comparisons, the two sites closest to and separated by Simkins Dam (502 and 504) showed the greatest increase in similarity (12.8%) following dam removal. Overall, control sites were more similar to one another than what was observed at sites adjacent to Simkins Dam. Pre- to post-dam removal similarity at control sites varied little, changing from a mean of 0.91 to 0.88.

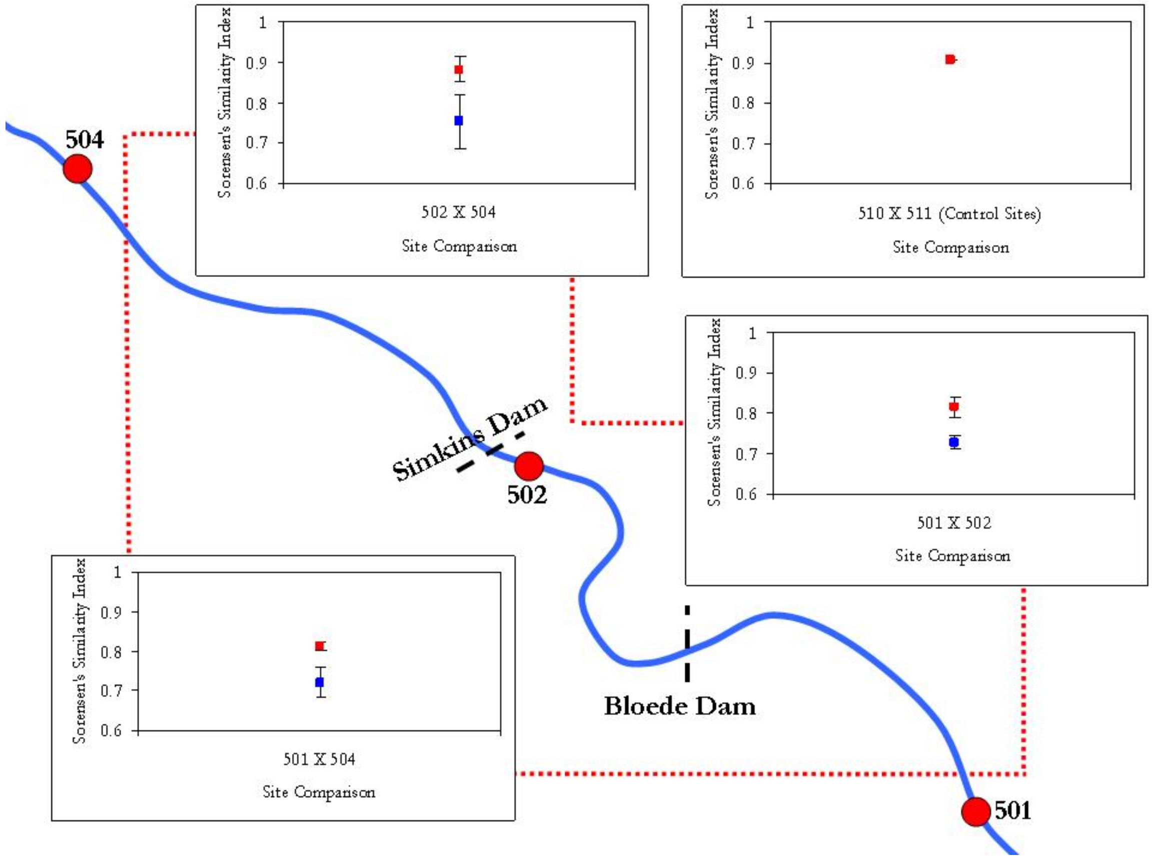

Figure 7.8: Pre- and post-dam removal fish assemblage similarity (excluding anadromous and semi-anadromous species) at sites adjacent to Simkins Dam. Blue squares depict pre-dam removal assemblage similarity. Red squares depict post-dam removal assemblage similarity. Error bars in all graphs represent the range of index scores over the study period. Note: The blue (pre-dam removal) square in the control site graph is obscured by the red (post-dam removal) square.

Analysis of similarity that included migratory anadromous and semi-anadromous species demonstrated the continued effect of Bloede Dam on fish assemblages in the lower Patapsco River (Fig. 7.9). The pattern of increased similarity among all sites following the removal of Simkins Dam noted in the analysis of non-anadromous species was consistent in this analysis. However, an increased dissimilarity was apparent among comparisons between 501 (located downstream of Bloede Dam) and all other sites (located upstream of Bloede Dam). Increased dissimilarity reflected the absence of anadromous and semi-anadromous species at sites above Bloede Dam.

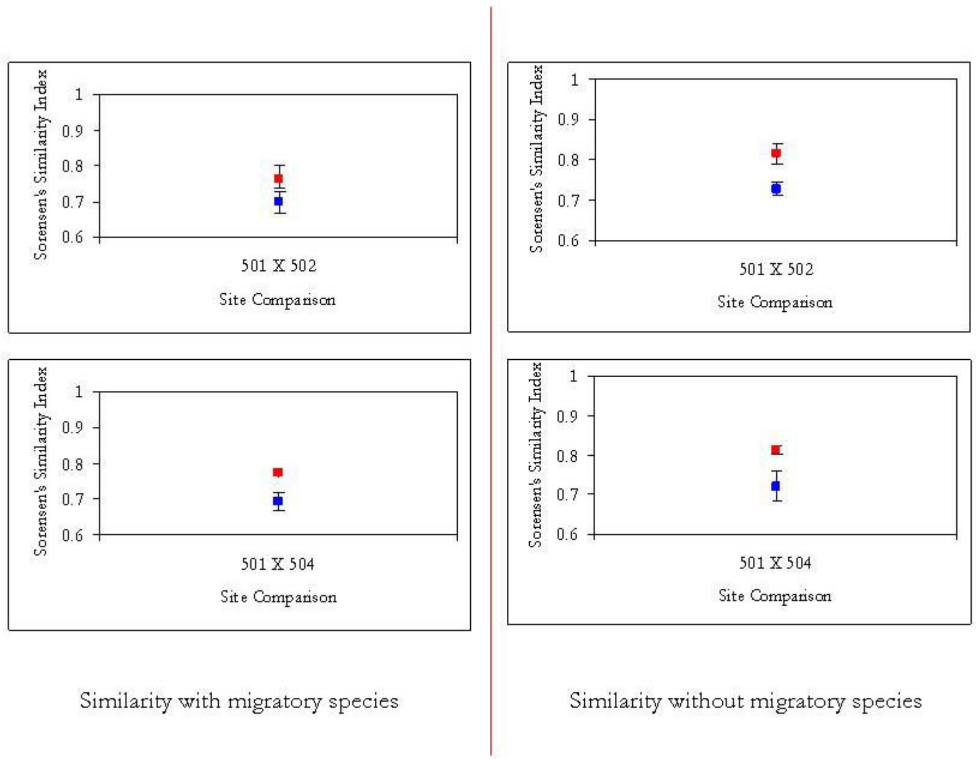

Figure 7.9: Comparison of pre- and post-dam removal fish assemblage similarity between site 501 (below Bloede Dam) and sites 502 and 504 with anadromous and semi-anadromous species included (left graphs) and removed from analysis (right graphs). Blue squares depict pre-dam removal assemblage similarity. Red squares depict post-dam removal assemblage similarity. Error bars in all graphs represent the range of index scores over the study period.

Smallmouth Bass Populations:

Smallmouth bass abundance generally increased in an upstream direction in the Patapsco River. Populations at each site were comprised predominately of YOY individuals followed by bass within the Stock size class. Very few bass within the Harvestable size class were collected during the study period (Fig. 7.10).

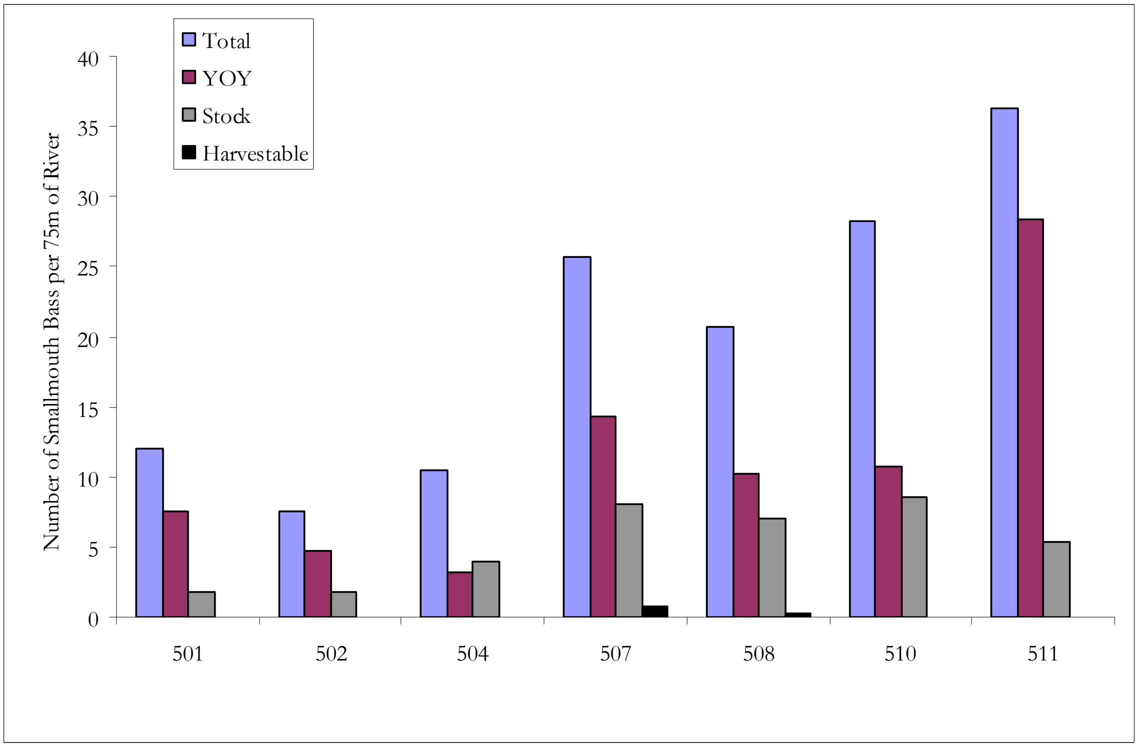

Figure 7.10: Smallmouth bass mean total abundance and abundance per size class at Patapsco mainstem sites from 2009 to 2012.

Smallmouth bass total abundance and abundance within each size category were quite variable at all sites sampled in the Patapsco River during the study period, including the two control sites 510 and 511 (Table 7.2). Mean total abundance and stock-sized bass decreased following dam removal at all sites (i.e., downstream, upstream, and control). Mean YOY bass abundance declined at downstream sites following dam removal. This pattern was opposite to that observed at upstream and control sites where increases in YOY bass were observed during the same period. No smallmouth bass were collected at site 502, located immediately downstream of Simkins Dam in 2012 (Fig. 7.11). Patterns of smallmouth bass abundance at site 504, located upstream of Simkins Dam, were similar to that of control sites during the study period.

Table 7.2: Pre- vs. post-dam removal comparisons of Smallmouth Bass total abundance and size classes at sites upstream and downstream of Simkins Dam and at mainstem control sites. Mean values are calculated from two sampling events – two prior to dam removal and two following dam removal.

Sites	Category	Pre-Removal Mean	Post-Removal Mean
Downstream	Total Abundance	12 (2.9)	7.5 (3.1)
	YOY (<90 mm)	8 (3.2)	4.3 (3.1)
	Stock (180-305 mm)	2.5 (0.7)	1.0 (0.4)
	Harvestable (>305 mm)	0	0
Upstream	Total Abundance	12.0 (3.0)	9.0 (4.0)
	YOY (<90 mm)	3.0 (3.0)	3.5 (0.5)
	Stock (180-30 5mm)	6.5 (4.5)	1.5 (0.5)
	Harvestable (>305 mm)	0	0
Control	Total Abundance	34.0 (8.9)	30.0 (2.6)
	YOY (<90 mm)	16.3 (10.4)	19.8 (4.0)
	Stock (180-305 mm)	11.0 (1.5)	4.3 (0.5)
	Harvestable (>305 mm)	0	0

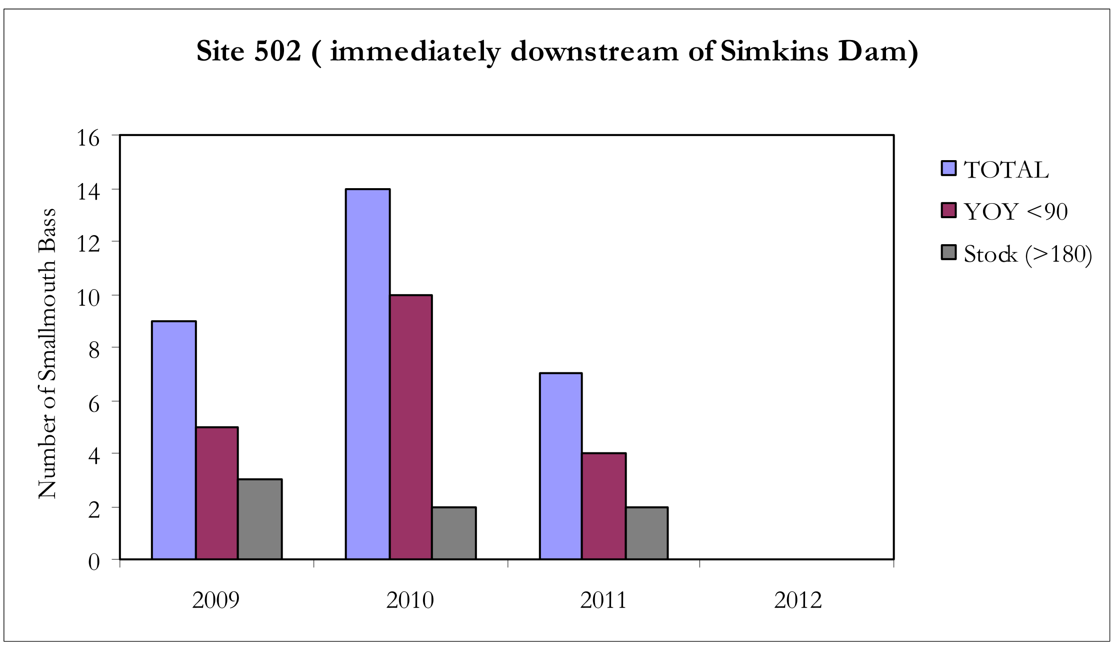

Figure 7.11: Smallmouth Bass mean total abundance and abundance per size class at site 502, located immediately downstream of Simkins Dam, from 2009-2012. No bass within the Harvestable size class were collected during the study.

Discussion

Assessing changes that occurred at control sites, unaffected by dam removals, in response to annual variation in precipitation, river flow, etc. provided a comparative benchmark that allowed us to look for responses in fish assemblages to dam removal in the Patapsco River, as represented by changes that were greater in magnitude than what occurred naturally in the river during the study period. The responses of fish assemblages to dam removal differed in upstream and downstream reaches and were, in general, similar to that documented in previous dam removal studies in other riverine systems.

Changes in fish assemblages were most pronounced in reaches downstream of Simkins Dam where fish habitat quality and quantity changed substantially as the river bottom shifted from predominantly coarse (e.g., cobble, boulder) to finer substrate (mostly sand). Fish assemblage instability was highest at site 502, located directly below the dam, than at all other sites sampled in the river. This instability reflected changes in species composition both from the addition of seven species collected only after dam removal and the loss of three species following dam removal. Species additions observed at both downstream sites (501 and 502) may reflect increased connectivity and dispersal potential resulting from the removal of Simkins Dam. However, similar species additions were observed at control sites during the study period. Downstream sites had a higher proportion of species loss - specifically the loss of shield darter, Blue Ridge sculpin, and central stoneroller - than other sites sampled in the river. The loss of these species is significant in that all three are species that are bottom-dwelling, utilize crevices and interstitial spaces in substrate as refuge, require coarse substrate free of fine sediment for successful reproduction, or in the case of the central stoneroller feed only on algae attached to rocks and other large debris (Jenkins and Burkhead 1994). The dependence of these species on coarse substrates likely made them sensitive to the influx of sand into downstream sites following dam removal. It is unclear if the loss of these species represents direct mortality or displacement. We also detected declines in total fish density and biomass at both downstream sites following dam removal. Observed changes in species richness and abundance affected the ecological composition of fish assemblages in downstream reaches. As documented in previous dam removal studies (Bushaw-Newton et al. 2002), downstream sedimentation had the greatest affect on benthic riffle fishes that need clean, coarse substrate as refuge and for feeding. Intolerant species – those sensitive to disturbance – also declined following removal of Simkins Dam. Although we documented a decline in lithophilic spawning species at one downstream site, this reproductive guild was unchanged at 502, a site where sedimentation was most evident. Although this reproductive guild may have remained unaffected by dam removal at this site, the lack of change could reflect a delayed response in these fishes. The effect of dam removal on these species may take more than two years to detect (Maloney et al. 2008). Although we expected biological integrity of fish assemblages to decline at both sites downstream of the dam, we documented decline at only one site – 501. Despite loss of species and changes observed in several ecological metrics, biological integrity at the other downstream site, 502, appeared to be unaffected by dam removal.

In addition to loss of species, declines in density and biomass, and shifts in the ecological composition of fish assemblages, we documented changes in smallmouth bass populations in reaches downstream from the former Simkins Dam site following its removal. We documented a decline in the young-of-year size class in bass populations. Although the causes for this decline are unknown, juvenile fishes tend to be more susceptible to increased sedimentation than adults (Doeg and Koehn 1994; Waters 1995). Sedimentation of downstream habitat following dam removal may have reduced juvenile survival. Conversely, this decline may simply reflect displacement of juvenile bass from this portion of the Patapsco River. Although stock and harvestable-sized bass abundance did not change in downstream areas during the study, we failed to collect bass of any size at 502 in 2012, even after bass habitat seemingly improved at this site. This may indicate a prolonged response of smallmouth bass populations to dam removal in this portion of the river.

Dam removal can improve fish habitat quality and increase biological integrity, species richness, and abundance of fishes and game fishes in upstream reaches (Kanehl et al. 1997; Catalano et al. 2007; Maloney et al. 2008). With the exception of increased fish density, we detected no change in the fish assemblage at our upstream site, 504, related to the removal of Simkins Dam. Variability in biological integrity and species richness observed at this site was similar to that observed at control sites during the study period. Our inability to detect further changes to upstream fish assemblages, those changes documented in other dam removal studies, was likely due in large part to the location of site 504. Unfortunately, we could not sample in the Simkins Dam impoundment – the area most altered by dam removal – due to river depth. This precluded the use of our standard quantitative fish sampling protocols (those used at all other sites). Site 504 was located upstream of the Simkins Dam impoundment and was, therefore, upstream of areas where fish assemblages were likely to be most affected by dam removal. Although we did not sample in the impoundment during our survey, there were noticeable improvements to fish habitat that occurred following dam removal as this portion of the river reverted to more natural, riverine conditions. These changes likely had positive affects on fish assemblages similar to those documented in previous studies, but were not detected by our study.

Dam removal is an important restoration tool that reverses years of habitat alteration, population fragmentation, and altered species distributions caused by the damming of riverine ecosystems. Although the long-term effects of dam removal are generally viewed as positive, dam removal is not without short-term, less positive consequences. As documented in this and other studies, the dam removal process causes immediate geomorphologic and hydrologic changes, especially in areas adjacent to the dam. These changes elicit subsequent responses in fish assemblages, some of which (e.g., declines in density and biomass, loss of native species, declines in recreationally important game fishes) are negative ecologically and may run counter to restoration goals (Gardner et al. 2011). However, the negative ecological effects of dam removal are usually short-lived (Doyle et al. 2005). Adjustments in channel geomorphology that drive much of the ecological changes associated with dam removal usually occur within the first five years (Doyle et al. 2005). Fish assemblages can recover rapidly following geomorphic stabilization. Initial declines in species richness and biological integrity have been shown to recover within one year following dam removal (Catalano et al. 2007). We expect that fish assemblages in the Patapsco River will recover when the remaining sediment behind the former Simkins Dam is transported out of the study area and geomorphic conditions stabilize. Increased connectivity among reaches should increase fish dispersal and hasten re-colonization of downstream sites by shield darter, Blue Ridge sculpin, and central stoneroller. Similarly, we expect fish density and biomass, the ecological composition of fish assemblages, and smallmouth bass populations to return to pre-dam removal levels (or better) as habitat conditions improve.

Despite some short-term negative impacts associated with the dam removal process, the removal of Simkins Dam will benefit fish assemblages in the Patapsco River over the long-term. Some positive effects of dam removal on fish assemblages were detectable even within the short time period of our study. We documented a decrease in the density of non-native fishes both upstream and downstream following dam removal; a pattern consistent with other dam removal studies (Kanehl et al. 1997; Bushaw-Newton et al. 2002). Although the decline of these species in this portion of the Patapsco River may be temporary, it likely reduced competition and predation pressures on populations of native fishes already stressed by other dam removal impacts. An additional positive effect of dam removal observed during our study was the increase in fish assemblage similarity in the river. Simkins Dam, as a fish blockage, impacted the distribution and abundance of fishes by altering hydrologic patterns and fragmenting habitat (Pringle 1997; Gardner et al. 2011). Its removal restored connectivity within a larger portion of the river ecosystem allowing for fish dispersal between upstream and downstream reaches. As a consequence, fish assemblages at sites adjacent to Simkins Dam became more similar in species composition. Sites previously separated by Simkins Dam showed the greatest increase in similarity following its removal.

Although fish assemblage similarity increased following the removal of Simkins Dam, the influence of Bloede Dam on assemblage composition was apparent. As documented in our similarity analysis that included anadromous and semi-anadromous species, Bloede Dam continues to disrupt the longitudinal connectivity of fish assemblages in the lower Patapsco River, leading to higher dissimilarity in species composition among adjacent reaches. Full restoration of fish assemblages in the river will require the removal of Bloede Dam.

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Chapter 8: Benthic Macroinvertebrates

Introduction

Benthic macroinvertebrates have been used as indicators of water quality for more than 60 years. They are especially useful in investigating physical and chemical disturbances in lotic systems. Macroinvertebrates are frequently used in these studies because they are relatively easy to collect using inexpensive and time-efficient methods and are a relatively large and taxonomically diverse group of organisms displaying an array of ecological traits. Several macroinvertebrate biotic indices and commonly used bioassessment metrics have been developed to assist biologists and environmental managers in understanding the effects of ecosystem disturbance.

Macroinvertebrate taxa have been assigned to groups, or guilds, based on their morphological adaptations. Three such groups commonly used in biological assessments are Functional Feeding Groups (FFGs), Habit groups and Habitat groups. Taxa are placed into FFGs based on morphological adaptations of the mouthparts (Cummins and Klug 1979, Cummins et al. 2008). For example, taxa placed in the scraper FFG have mouthparts designed to scrape hard surfaces for periphyton - their primary food source. Habit groups categorize a taxon based on its mode of existence (or general habits). Habit groups are based on mode of locomotion, attachment, or concealment (Cummins et al. 2008). For example, burrowers, as the name suggests, burrow into substrate - typically fine sediments. Taxa are also placed into Habitat groups such as lotic-erosional (running waters/riffles), lotic-depositional (running-water/pools and margins), and lentic (standing water). These groups represent the habitat(s) in which a taxon is most frequently observed or those that provide more suitable or optimal conditions. These groups are frequently used in biological assessments to describe the macroinvertebrate composition of an area.

Substrate composition has been found to be a major factor driving the macroinvertebrate distribution and abundance in streams and rivers (Cummins and Lauff 1969, Hynes 1970). For example, sprawlers and burrowers are generally more abundant in slow-moving, depositional areas dominated by sand since they are better adapted to these conditions. Other organisms, such as clingers and scrapers, are typically more abundant in lotic-erosional areas with exposed rock surfaces, as this habitat is more suitable to their morphological adaptations, i.e., body shape, feeding mechanism, and mode of existence. Optimal habitats are also more likely to provide an abundant food source. For example, sprawlers that feed on leaf litter are more likely to be found in depositional areas where large quantities of leaf litter are more likely to accumulate. A change in the substrate composition of an area will likely be followed by a shift toward organisms better adapted to the new conditions.

Impoundments, such as dams, that alter the natural flow regime of a stream or river can impact these systems by altering sediment transport and biotic processes (Ward and Stanford 1979, Poff et al. 1997, Pizzuto 2002). These impacts can affect macroinvertebrates in a number of ways. One such way is by the alteration of preferred habitat and available food sources for certain organisms, creating a change in macroinvertebrate composition (Doeg and Koehn 1994). Despite the removal of more than 400 dams during the last century (Hart et al. 2002), over 100 of which were small dams in the United States (Born et al. 1998, American Rivers et al. 1999), few studies have examined the effects dam removal has had on the pre- and post- removal macroinvertebrate communities. The majority of the literature examining the effects of dam removal on macroinvertebrates tends to focus on the changes in communities as it relates to the changes in fine sediment condition and substrate composition before and after the removal of the dam.

Wood and Armitage (1997) provide a summary of the effects fine sediment (sand, silt, and clay) and sedimentation can have on macroinvertebrates. In general, sedimentation alters the composition of the substrate, increasing the suitability for some taxa (e.g., burrowers) while reducing that for other taxa (e.g., clingers). Sedimentation affects macroinvertebrates in a number of ways: clogging interstitial spaces, thereby reducing available habitat for certain macroinvertebrates such as clingers and net-spinning caddisflies; the accumulation of sediment particles in the nets of filter-feeders which affects the nets' ability to collect food particles; smothering or scouring of periphyton communities which serve as a food source for scraper taxa; or increasing drift downstream of certain taxa. Several studies have shown that a reduction in macroinvertebrate densities and diversity downstream of removed dams appears to be associated with an increase in sediment (Gray and Ward 1982, Doeg and Koehn 1994, Renofalt et al. in press, Tiemann et al. 2004, Thomson et al. 2005, Orr et al. 2008,).

Sediment-laden impoundments are often dominated by lentic taxa while assemblages in free-flowing habitats have a higher abundance of lotic-erosional taxa. A frequently observed result of dam removal is the shift from an assemblage with more lentic taxa in impounded areas (pre-removal) to an assemblage with more lotic taxa (post-removal) (Hart et al. 2001, Bushaw-Newton et al. 2002, and Stanley et al. 2002). Such shifts in composition often follow a post-dam removal shift in substrate composition from fine sediments to a more heterogeneous mixture of different rock sizes; habitat more suitable to the lotic-erosional taxa than the lentic taxa. Few studies have examined the long-term changes in the macroinvertebrate community following dam removal.

We investigated macroinvertebrate communities at sites in the Patapsco River before and after the removal of Simkins and Union Dams to observe potential changes over this period. The results were analyzed and the major observations are reported here. We anticipate that information will be useful in future predictions and assessments examining the changes in macroinvertebrate communities that occur as a result of the removal of a dam.

Methods

The Macroinvertebrate Fauna Collected from Mainstem Patapsco River Sites

Macroinvertebrates were collected from 16 mainstem Patapsco River sites (Fig.8.1) from 2009-2012 following MBSS protocols (Stranko et al. 2007). Macroinvertebrates were collected twice before the removal of Simkins Dam (summer 2009 and spring 2010) and four times after the dam was removed (spring and summer 2011 and 2012). Individuals were identified by DNR taxonomists to genus-level, if possible, or to the lowest taxonomic level if genus-level identification was not possible (Boward and Friedman 2000).

Change in the Macroinvertebrate Communities

Pre-dam removal data were combined to form a "Pre" dataset. 2011 data were combined to form a "Post1" dataset and 2012 data were combined to form a "Post2" dataset. The following metrics were calculated using these datasets: Functional Feeding Group proportions (%Scraper, %Collector-gatherer, %Filter-feeder, %Predator, and %Shredder), Habit proportions (%Clinger, %Climber, %Sprawler, %Burrower, and %Swimmer), Non-insect individuals, %Non-insects, %EPT (Ephemeroptera, Plecoptera, and Trichoptera), EPT taxa richness, %Trichoptera, %Ephemeroptera, %Chironomidae, total taxa richness, total individuals, and BIBI (Benthic Index of Biotic Integrity). "Pre" data for each metric were subtracted from both the "Post1" and "Post2" data to determine the change in each metric from 2009/2010 to 2011 ("Post1 - Pre") and 2009/2010 to 2012 ("Post2 - Pre"). Explanations of selected metrics can be found in Appendix 8.1.

Control Site

Control site data were recorded during the same sampling periods as the treatment sites (2009-2012) to assess potential changes in the macroinvertebrate community not associated with dam removal. Data were not collected from the control site prior to the beginning of this study (2009). Two major events that occurred during this study that could have created substantial changes other than the removal of two dams were storm-created flooding events that occurred in September and November 2011. Substantial changes observed during the Post1 - Pre period are more likely to have occurred due to the removal of Union and Simkins than those observed during Post2 - Pre. During the Post2 – Pre period, both the dam removals and flooding events occurred, making it difficult to determine which changes were associated with which event. Changes at the control site indicate that the 2011 storms had a major influence on the macroinvertebrate community. Any changes observed at sites around Union or Simkins would need to exceed control site changes to suggest an association between dam removal and macroinvertebrate community changes at the dam sites.

Comparison of Macroinvertebrates Collected at a Free-flowing Site and an Impounded Site Prior to the Removal of Simkins Dam

Stanley et al. (2002) compared the macroinvertebrate community at free-flowing reference sites to that found at sites impounded by a dam. Their study focused on the differences in EPT and clingers and found these groups were higher at the free-flowing reference site. Stanley et al. (2002) also analyzed dominant taxa comprising the fauna at free-flowing sites and impounded sites. In their study, *Cheumatopsyche* and *Ceratopsyche*, naidid worms, *Orthocladius*, and *Maccaffertium/Stenonema* (heptageniid mayflies) accounted for 56% of individuals collected at the free-flowing sites. Tubificid worms, *Chironomus*, and *Polypedilum* (chironomids) comprised 59% of the individuals at the impounded sites.

We examined EPT richness, %EPT, number of EPT individuals, clinger richness, %Clinger, and number of clinger individuals at an impounded site immediately upstream of Simkins (B08) and a free-flowing site further upstream of Simkins (510). The dominant taxa at the free-flowing and impounded sites were also examined.

Change in Lentic and Lotic-erosional Taxa at Two Upstream Impounded Sites Before and After the Removal of Simkins Dam

Merritt et al. (2008) provides a summary of habitats that each aquatic insect family and genus inhabit. Lotic-erosional, lotic-depositional, and lentic are among the habitat categories used that pertain to this study. Some taxa are grouped into several categories, making it possible for the habitat of a taxon to be listed as both lotic-erosional and lentic. The information found in Merritt et al. (2008) was used to categorize the habitat for each taxon collected during this study.

Macroinvertebrates collected at two sites immediately upstream of Simkins, B08 and B09, during Pre, Post1, and Post2 periods were analyzed. The number of taxa and individuals within these taxon known to inhabit lotic-erosional and/or lentic habitats were counted and summed. The same analysis was used to determine the lotic-erosional and lentic taxa at a control site, 510.

Shannon Wiener Diversity Index

A Shannon Wiener diversity index was used to observe the temporal changes in macroinvertebrate diversity at sites during the Post1 - Pre and Post2 –Pre periods. All amphipod taxa (i.e., *Caecidotea*, *Crangonyx*, *Gammarus*, and taxa identified to the ordinal level of Amphipoda) were combined into a single amphipod group. Taxa collected at a site that were identified to order, family, subfamily, or tribe and were determined by taxonomists to not be a new taxon were deleted from the dataset in a consistent manner. The deleted individuals accounted for 3% of the total number collected during this study.

Results

Macroinvertebrate Fauna Collected from Mainstem Patapsco River Sites

During the study period (2009-2012), 11,385 individuals were analyzed. The most collected orders were Diptera (4,492 individuals), Amphipoda (2,787), Trichoptera (1,451), Ephemeroptera (1,227), and Coleoptera (750). The most abundant genera in these orders are as follows: Diptera (*Orthocladius* and *Hydrobaenus*), Amphipoda (*Gammarus*), Trichoptera (*Chimarra* and *Cheumatopsyche*), Ephemeroptera (*Baetis*), and Coleoptera (*Stenelmis*). Non-insect individuals accounted for 3,248 individuals.

Change in the Macroinvertebrate Communities

Figures in this section contain shaded portions representing major shifts in sediment condition that occurred after the removal of Union and Simkins dams. Figure 8.1 shows the sites sampled, the locations of extant and removed dams, and areas in which a sediment shift was observed and measured. Shaded areas represent a shift from sand to cobble/gravel (eroding), from cobble/gravel to sand (agrading). In some areas a shift in dominant habitat from cobble/gravel to sand did not occur until 2012.

Several taxa contributed to the increase or decrease in a parameter(s) at sites upstream or downstream of Simkins Dam (Upstream: B08, B09, B10, 504; Downstream:

B07, 502, B06, B05). Numerically, all of these taxa were relatively abundant and, as such, the largest increases and decreases in the number of individuals of any taxon at a site over a period of time typically were observed with these "driver" taxa. Most macroinvertebrate taxa are included in a number of different FFG and Habit groups and can be included in more than one category within each group (e.g., inclusion in both clinger and swimmer categories within the Habit group), allowing certain taxa to drive multiple parameters at one site. The taxa that drove each parameter throughout the Patapsco sampling sites were analyzed and the results are included here. In this section, a superscript 1 refers to the period Post1 – Pre and a superscript 2 refers to Post2 – Pre. These superscript numbers appear after the site number. Results from the metrics examined here are presented below. Not all of the metrics that were analyzed are discussed.

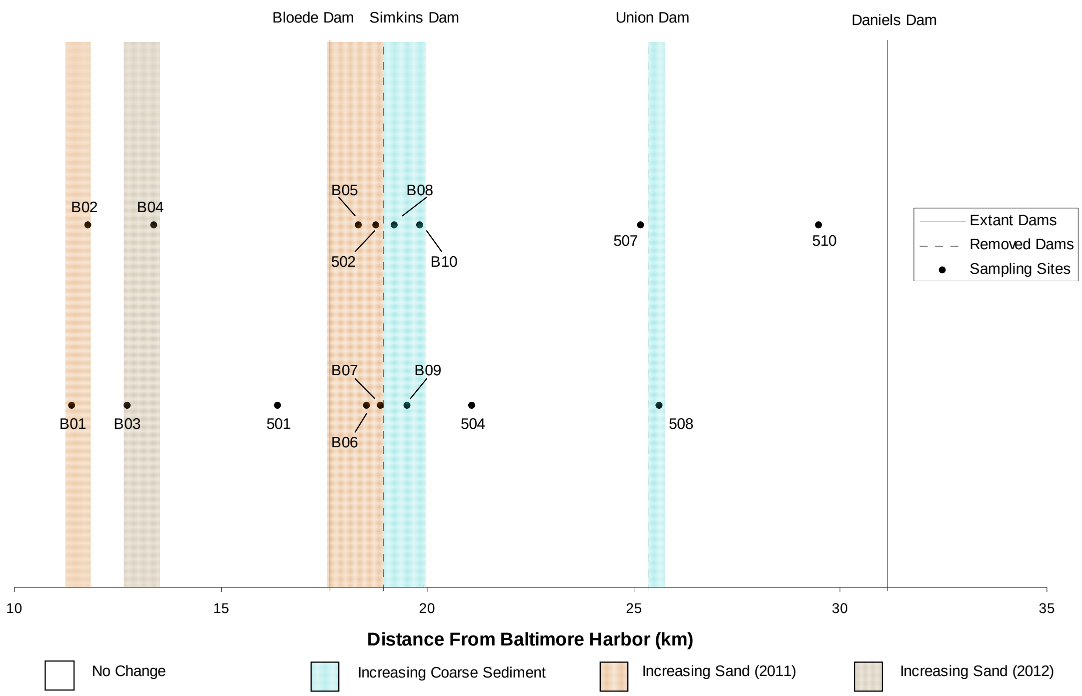

Figure 8.1: Mainstem Patapsco River sites sampled from 2009-2012.

Control Site

In the graphs below, the control site (510) is the furthest upstream site, or furthest site to the right (on the graphs). During both Post1 – Pre and Post2 - Pre, the following metrics increased at the control site: %Clinger, EPT richness, %EPT, %Filter-feeder, and BIBI. Decreases in %Burrower, %Non-insect, %Shredder, and %Sprawler were also recorded at the control site.

Benthic Index of Biotic Integrity

The BIBI score decreased at only two sites between Simkins and Bloede dams (Fig.8.2). Increases in 1 and 2 BIBI ratings (i.e., an increase in 1 rating would change the BIBI at a site from Poor to Fair) occurred at numerous Patapsco sites from Pre to Post1 and Post2 periods. The majority of these increases occurred upstream of Simkins Dam. All but one of the sites where no BIBI change was recorded were downstream of Simkins Dam.

BIBI scores increased at the control site during both periods. For the sites upstream of Simkins, the BIBI increase was equal to or greater than the increases at all but one site, 504, during the Post2 – Pre period. All but one site (B02) downstream of Simkins Dam had lower BIBI scores than the control during both periods.

%Non-insect, %Sprawler, and %Shredder

Decreases in the %Non-insect metric, primarily >20%, occurred at all but three sites and were greatest at sites upstream of Simkins and Union dams (Fig.8.3). Compared to the control site, %Non-insect decreases were greater at the sites upstream from Simkins and Union dams and the increases were greater at two sites below Simkins Dam. The largest decreases, from 21 to 59%, were observed at sites around Simkins Dam and one site below Bloede Dam. %Sprawler and, to a lesser extent, %Shredder followed a pattern similar to %Non-insect (Figs. 8.4 and 8.5). Increases in %Shredder occurred at more sites than the other parameters, and the largest increases (observed at Simkins Dam sites) were substantial deviations from the control site values. Site B10, upstream of Simkins Dam, recorded the only %Shredder decrease greater than that observed at the control site. These were due to increases in amphipods and a chironomid, *Cricotopus*, from the previous year. The largest deviations in %Sprawler from the control site were observed at sites around Simkins Dam and at B03 downstream of Bloede Dam. Similar to %Non-insect and %Sprawler, %Shredder generally increased at sites from Post1 – Pre to Post2 – Pre.

Amphipods (non-insect crustaceans), the second most numerically abundant taxonomic group collected in this study with 2,787 individuals, dominated the increases and decreases in %Non-insect (Fig.8.3), %Sprawler (Fig.8.4), and %Shredder (Fig.8.5). This group consisted mostly of individuals of *Gammarus* as well as a few *Caecidotea* and *Crangonyx* individuals. Large decreases (>45 individuals) in amphipods decreased %Shredder at four Simkins Dam sites (504[1], B10[1,2], B09[2], and B06[1,2]), %Sprawler at five Simkins Dam sites 504[1], B10[1,2], B09[1,2], and %Non-insect at five Simkins Dam sites (504[1], B10[1,2], B09[1], B08[1,2], B07[1,2], B06[1,2]). Large increases (>45 individuals) led to %Shredder and %Non-insect increases at two downstream sites, B05[1] and 502[2], and a %Sprawler increase at B05[1]. Both *Hydrobaenus* and *Orthocladius* assisted amphipods in driving decreases in %Sprawler at B09[1,2] and B07[1,2]. During the periods of Post1 – Pre and Post2 – Pre, the number of amphipod individuals substantially increased at 502[2] (+48) and B05[1] (+59). All other Simkins Dam sites during these periods observed large declines in amphipods.

EPT Taxa Richness and %EPT

A similar pattern was observed for EPT richness and %EPT. Increases for both were highest upstream of Simkins Dam, post-removal, with the largest decreases occurring at sites between Bloede and Simkins dams (Figs. 8.6 and 8.7). The majority of sites below Bloede Dam also recorded increases in both parameters. An increase in %EPT was observed at all sites upstream of Simkins Dam and an EPT richness increase was observed at all but one site, 507, immediately downstream of Union Dam. The largest %EPT increases occurred at sites upstream of Simkins and Union dams. The decreases in EPT richness were all less than three and occurred directly downstream of Union Dam, at two sites between Bloede and Simkins dams, and two sites downstream of Bloede Dam, with the most notable having occurred between Bloede and Simkins dams. An increase in EPT richness was observed during Post2 - Pre at one of these sites downstream of Bloede Dam. There were no decreases in %EPT upstream of Simkins Dam during either Post1 - Pre or Post2 - Pre.

At the control site, EPT richness and %EPT increased during both Post1 – Pre and Post2 – Pre. EPT richness increases were greater than that observed at the control at the first three sites upstream of Simkins during Post1 – Pre where a shift from sand to cobble/gravel occurred. Five sites downstream of Union, Simkins, and Bloede recorded substantially lower EPT richness during both periods than the control sites. Compared to the control site, increases in %EPT were greater at one downstream and two upstream Simkins sites and decreases were greatest at two sites immediately downstream of Simkins and sites B03 and B04 below Bloede.

Changes in EPT richness and %EPT were largely driven by four caddisfly taxa and two mayfly taxa. The caddisfly genera *Chimarra*, *Cheumatopsyche*, *Hydropsyche*, and *Ceratopsyche* were primarily responsible for decreases in %EPT at 502[1,2] and increases at B09[2] and B06[1]. Decreases in *Heterocloeon*, a baetid mayfly characterized as a swimmer/clinger and scraper that inhabits lotic-erosional habitat, and *Baetis* factored into declines in %EPT at B05[1]. Increases in %EPT due to *Baetis* occurred at B09[1], B06[2], and B10[1].

%Burrower

During this study, there was an increase in %Burrower at sites downstream of Simkins Dam and a decrease at sites upstream of Simkins Dam (Fig.8.8). In observing this change during the two time periods, there was a greater increase in %Burrower at the sites downstream of Simkins Dam and a greater decrease at sites upstream of Simkins Dam during Post2 – Pre than Post1 – Pre. The largest decreases in the first year after dam removal occurred at the two sites immediately upstream of Simkins Dam. While %Burrower increased during Post2 – Pre at these two sites, this group decreased at every site upstream of these sites during this period. The decrease that occurred at B08 upstream of Simkins Dam during Post1 – Pre was the only large deviation (decrease) from the value recorded at the control site. Compared to the control site, the largest increases in %Burrower occurred at sites downstream of Simkins and Bloede dams.

Individuals within the families Pisidiidae and Chironomidae were responsible for changes in %Burrower (Fig.8.8). Pisidiids, or fingernail clams, are bivalve mollusks that burrow into sediment where they filter-feed. The loss of all 73 pisidiid individuals at B08 following removal of Simkins Dam drove a %Burrower decrease at this site. Tube-building chironomid taxa, primarily *Orthocladius*, and to a lesser extent, *Cricotopus,* drove %Burrower at five Simkins Dam sites. The decrease caused by these taxa occurred at 504[1]. The increases occurred at B08[1,2], 502[1], B06[2], and B05[2].

%Clinger

%Clinger increased at all but three sites, all of which were downstream of Simkins Dam (Fig.8.9). The largest increases occurred in the year following the removal of Simkins Dam (Post1 – Pre) and were observed at sites upstream of Simkins and Union dams. These increases were notably greater than those observed at the control site. An increase of 21% or more was observed at six of the eight Simkins Dam sites during this period. From Post2 – Pre, %Clinger decreased from Post1 – Pre levels at all but two sites around Union Dam and one site downstream of Bloede Dam. The majority of these Post2 – Pre decreases were substantial deviations from the Post2 – Pre control site value.

Caddisflies, mayflies, and elmid beetles were the primary drivers of the %Clinger metric. Species in the caddisfly genera *Chimarra*, *Cheumatopsyche*, *Hydropsyche*, and *Ceratopsyche* led to %Clinger increases at B10^{1,2}, B07^{2}, and B06^{1}. *Baetis* was the most abundant of the nine baetid genera collected during this study, accounting for 51% of all mayflies, and was the primary factor in an increase at B06^{2}, and assisted in an increase with the four caddisfly taxa above at B10^{1}. Decreases in *Heterocloeon* and *Baetis* drove declines in %Clinger at B05^{1}. Large changes in elmid beetle numbers factored into an increase in %Clinger at 504^{1,2}.

%Filter-feeder

In general, the %Filter-feeder metric increased at most sites with the exception of two sites around Simkins Dam (Fig.8.10). The declines at a site downstream of and one site upstream of Simkins Dam were expected as filter-feeders have been found to be abundant in impounded areas (Petts 1984 and Schlosser 1992). These declines were notably different from the %Filter-feeder values observed at the control site. With the exception of a few sites, the change in %Filter-feeder from Post2 – Pre to Post1 – Pre was minimal. The increase in burrowing filter-feeding taxa downstream of Bloede Dam and filter-feeding net-spinning caddisflies around Simkins was the cause of the increases in %Filter-feeder in these areas.

Rheotanytarsus, a tube- and net-building filter-feeding chironomid that primarily inhabits lotic-erosional habitat, contributed along with trichopterans in %Filter-feeder increases at two sites, B10^{1,2} and B06^{1}. *Chimarra*, *Cheumatopsyche*, *Hydropsyche*, and *Ceratopsyche* are caddisfly genera that are filter-feeding clingers that contributed to a %Filter-feeder decrease at 502^{2} and %Filter-feeder increases at B09^{1} and B07^{1,2}. The decrease in pisidiids at B08^{2} also drove the decrease in %Filter-feeder observed at this site.

%Scraper

%Scraper decreased at all but five sites during this study, with the majority of decreases occurring downstream of Simkins Dam (Fig.8.11). Compared to the %Scraper values at the control site, notable declines were observed at sites around Simkins Dam and downstream of Bloede Dam while notable increases occurred upstream of Union and Simkins dams. Decreases occurred at all but two sites downstream of Simkins Dam where cobble/gravel habitat was replaced by sand habitat as the dominant habitat. With the exception of the first two sites upstream of Simkins Dam, %Scraper increased at sites in which cobble/gravel replaced sand. Increases were also observed at two sites downstream of Bloede Dam (B03 and 501) where substrate was composed of sand/gravel. These increases along with those at 504 and the site upstream of Union Dam (508) were the highest recorded during this study. The decreases at the two sites upstream of Simkins Dam were unexpected and were due to major losses in individuals of *Hydrobaenus*. Increases in *Hydrobaenus*, mayflies, and, primarily, elmid beetles caused the %Scraper increases at two sites downstream of Simkins Dam. The loss of numerous elmid individuals at one of these sites, B03, caused a decrease in %Scraper from Post1 – Pre to Post2 – Pre.

Changes in %Scraper were primarily due to *Hydrobaenus*, elmid beetles, and a mayfly, *Heterocloeon*. Large decreases (>29 individuals) in *Hydrobaenus* led to %Scraper decreases at three sites (B08^{2}, 502^{2}, and B05^{1,2}). Large changes in elmid beetle numbers factored into an increase in %Scraper at 504[1]. Decreases in *Heterocloeon* and *Baetis* factored into declines in %Scraper at B05[1].

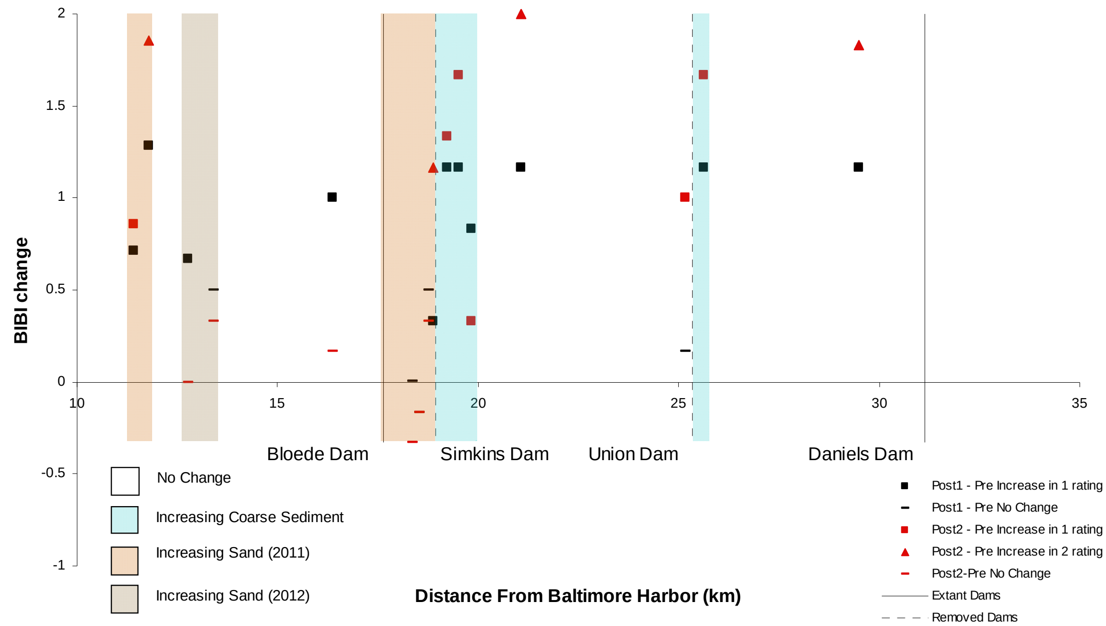

Figure 8.2: Change in BIBI scores (numerical) and ratings (categorical) at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre. Changes in BIBI ratings (i.e., Good, Fair, Poor, and Very Poor) are represented by a square symbol (increase in 1 rating category), a triangle (increase in 2 rating categories), or a hyphen (no change in rating category).

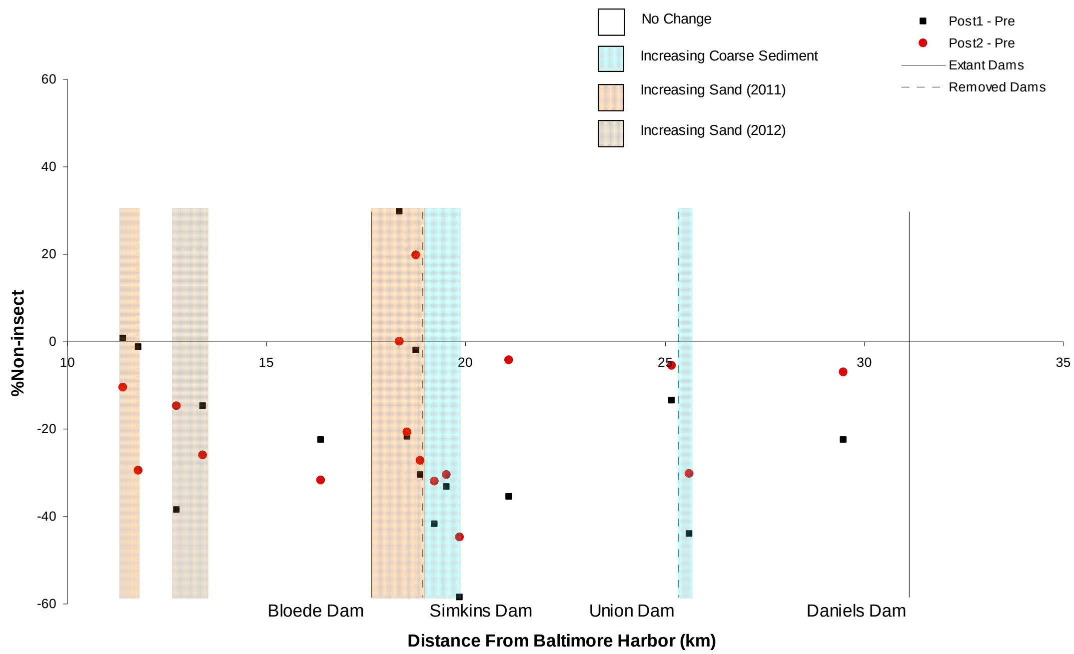

Figure 8.3: Change in %Non-insect at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

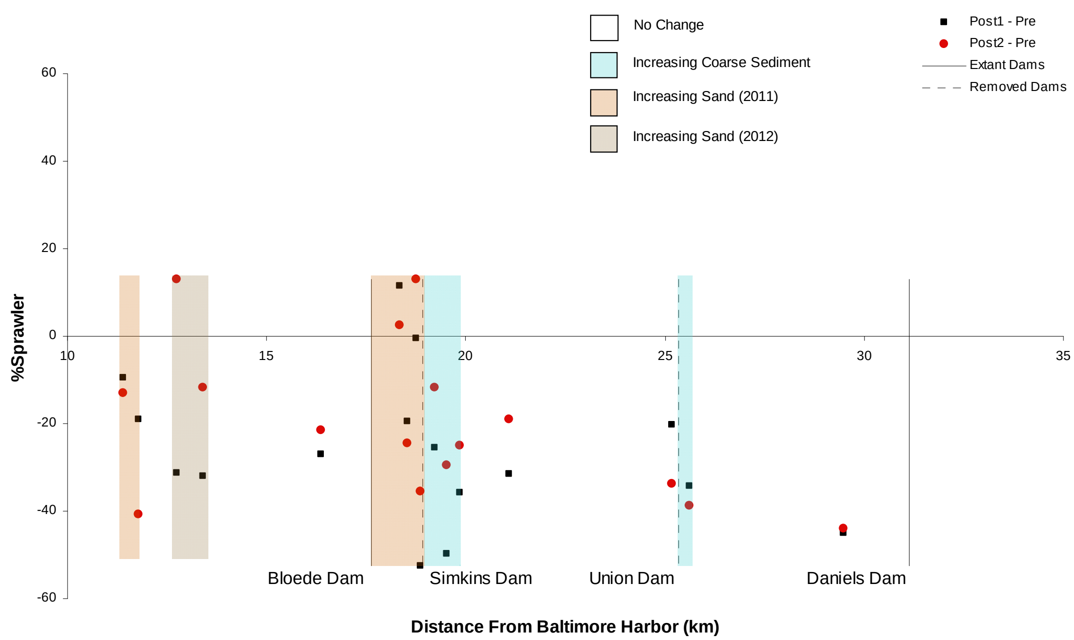

Figure 8.4: Change in %Sprawler at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

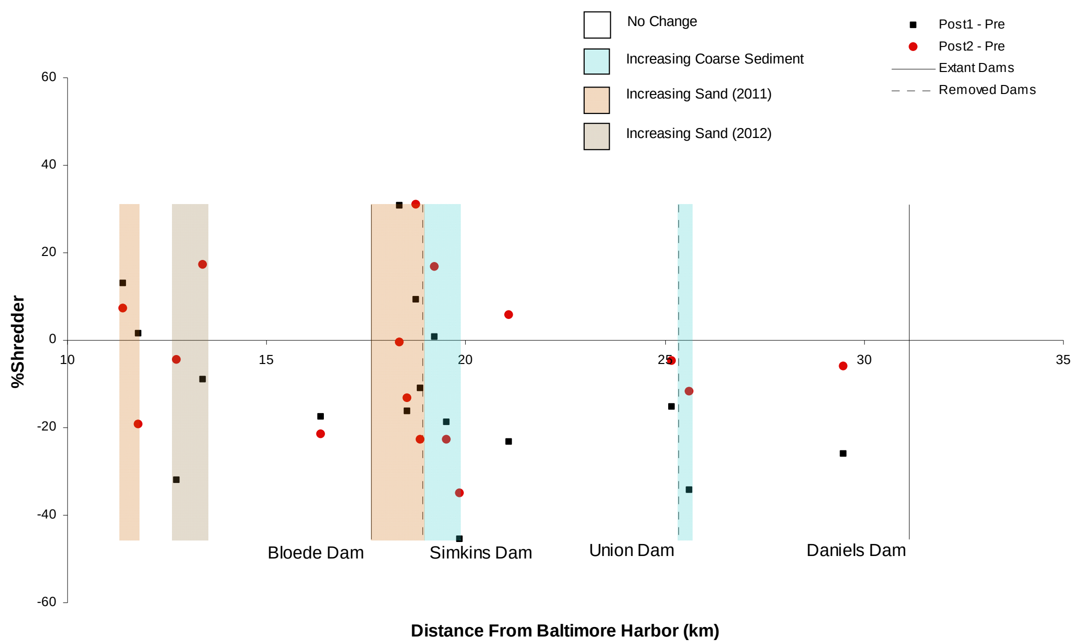

Figure 8.5: Change in %Shredder at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

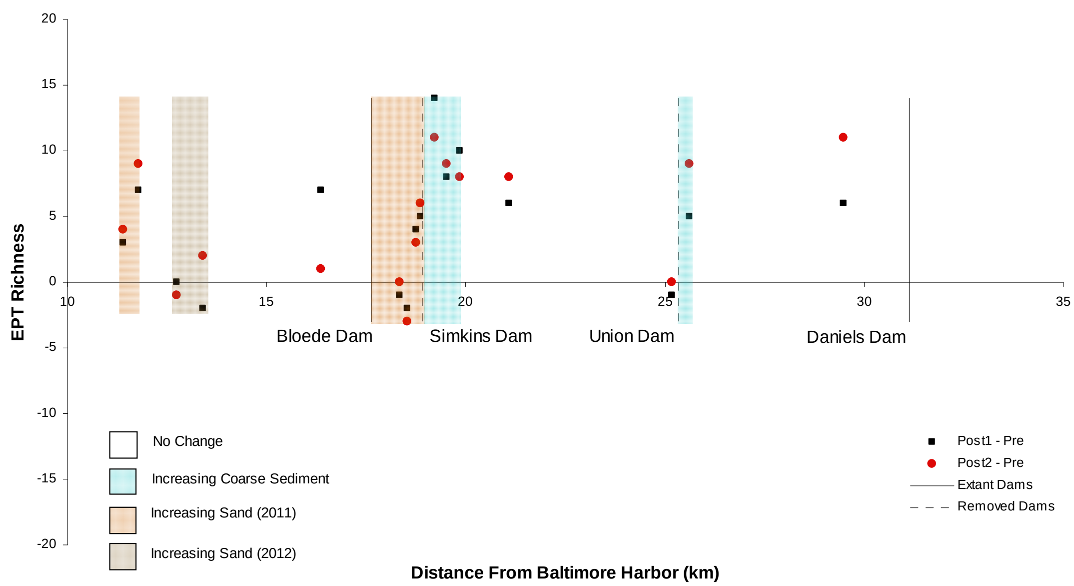

Figure 8.6: Change in EPT richness at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

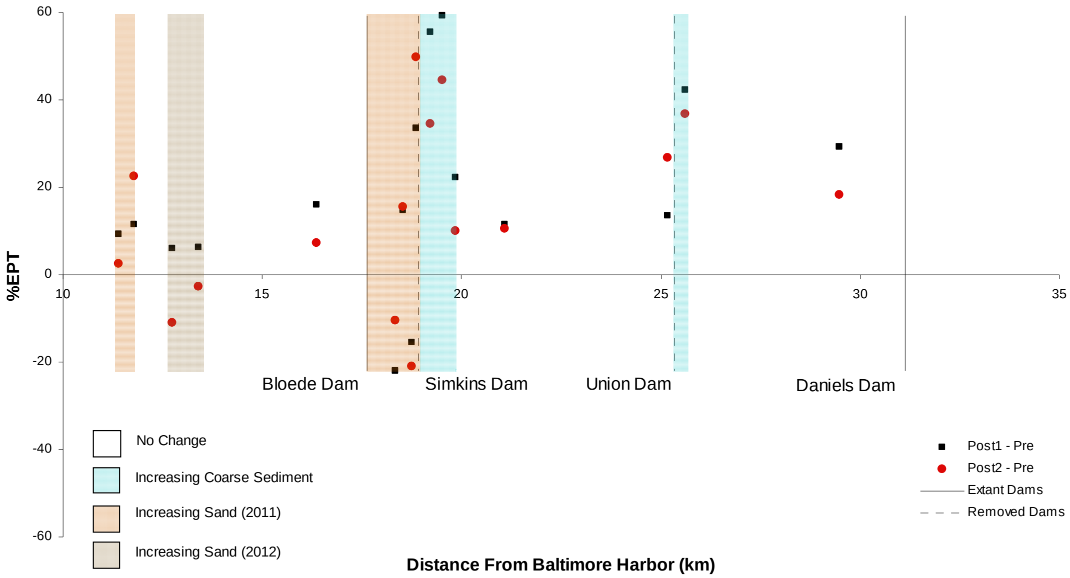

Figure 8.7: Change in %EPT at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

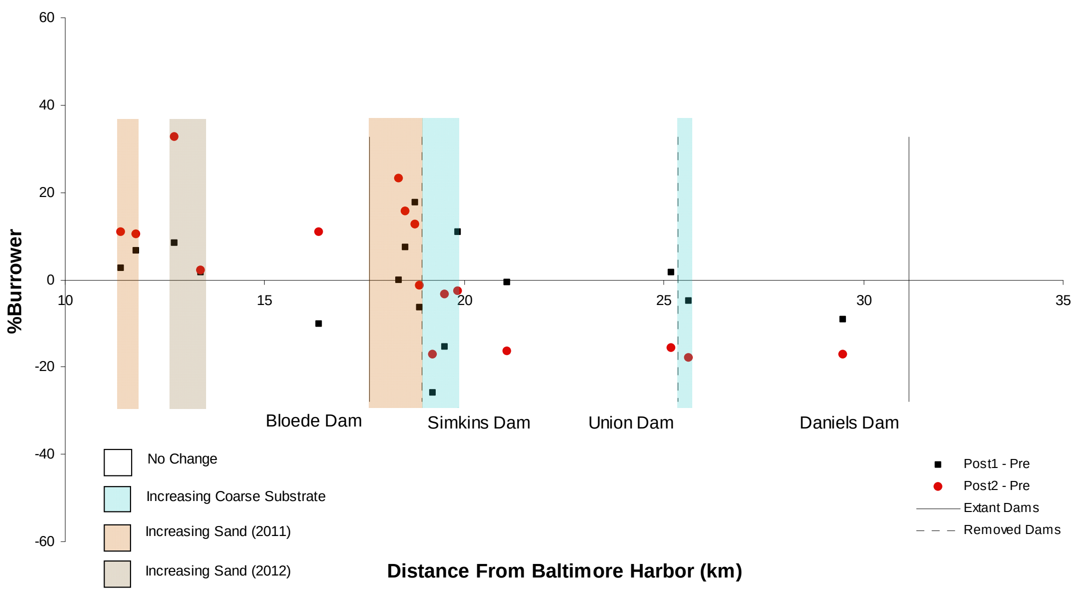

Figure 8.8: Change in %Burrower at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

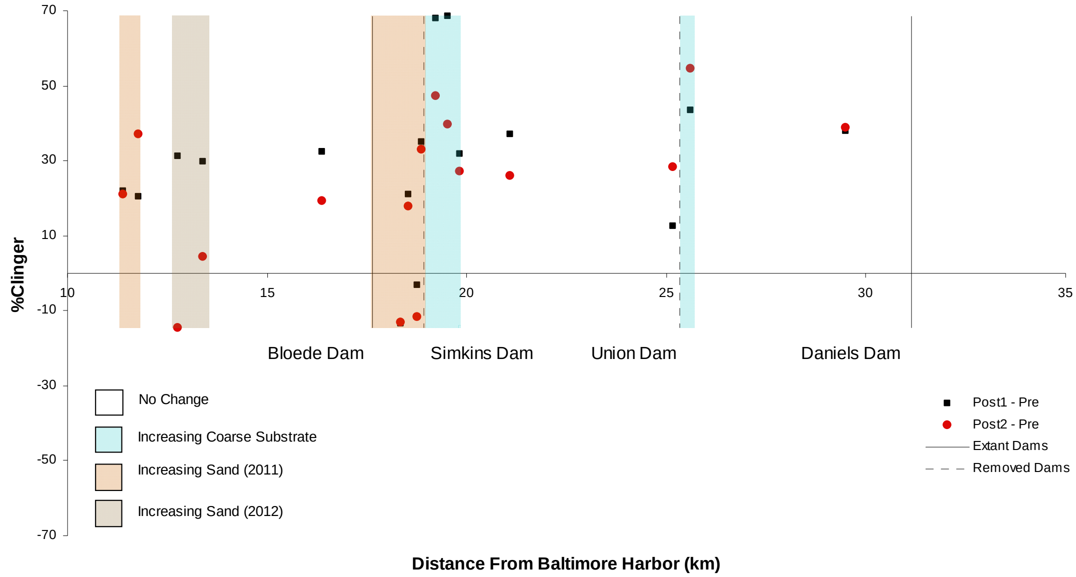

Figure 8.9: Change in %Clinger at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

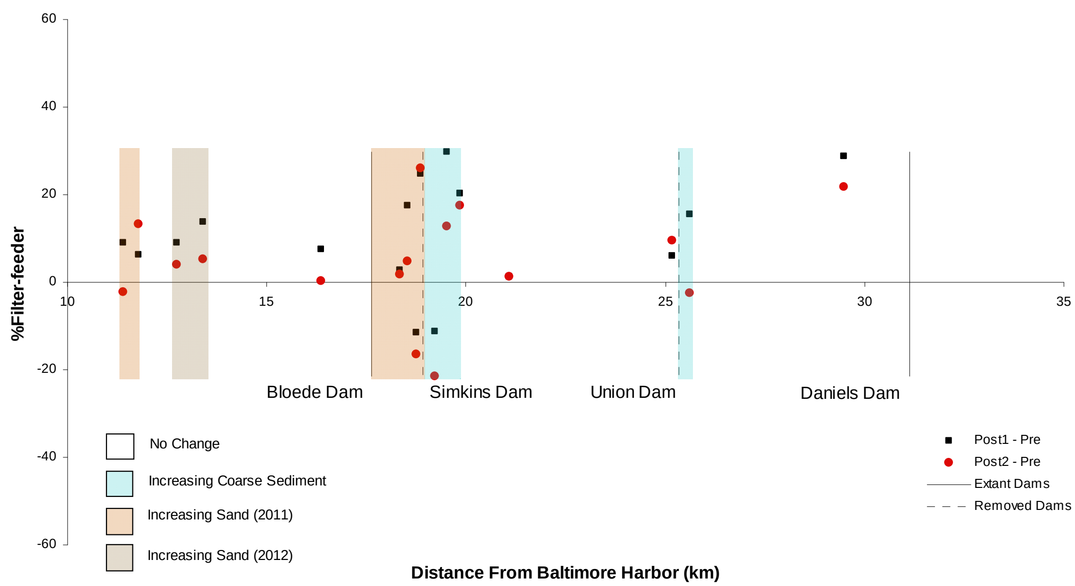

Figure 8.10: Change in %Filter-feeder at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

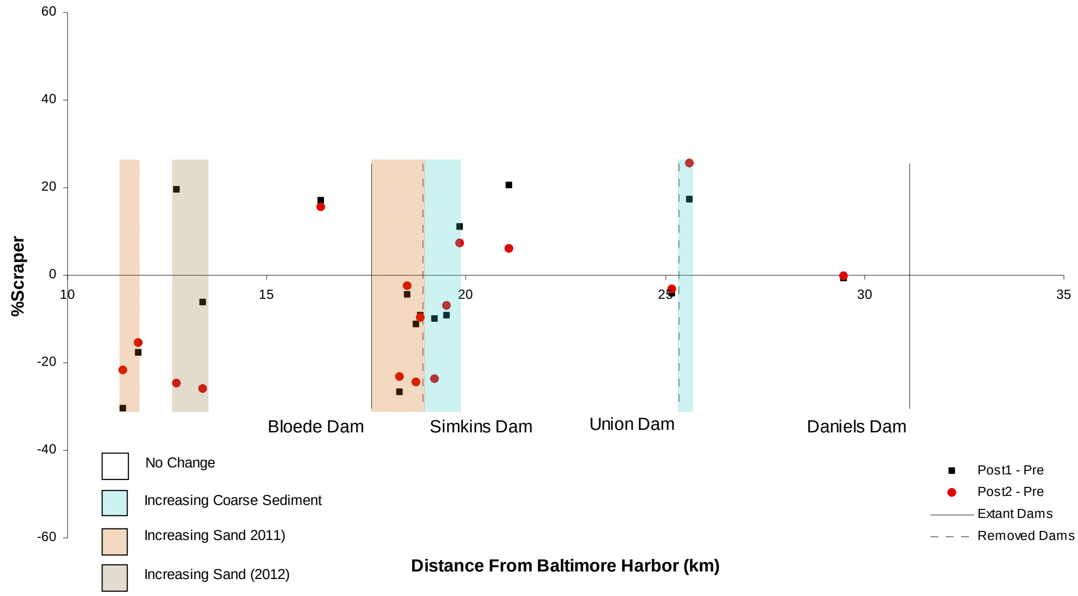

Figure 8.11: Change in %Scraper at Patapsco River sites during the time periods Post1 - Pre and Post2 - Pre.

Comparison of Macroinvertebrates Collected at a Free-flowing Site and an Impounded Site Prior to the Removal of Simkins Dam

EPT and clinger percentages, taxa richness, and number of individuals were all greater at the free-flowing site, 510, than the impounded site, B08 (Table 8.1). These results were similar to those reported in Stanley et al. (2002). *Hydrobaenus*, *Orthocladius*, and *Gammarus* were the dominant taxa at the free-flowing site, accounting for 72% of the individuals collected. The impounded site was dominated by pisidiid bivalves and *Hydrobaenus* which accounted for 75% of the individuals.

Table 8.1: Number of individuals, taxa richness, and percentage of EPT and clingers at an impounded site (B08) and a free-flowing site (510) in the Patapsco River.

	Impounded Site	Free-flowing Site
Metric		
EPT individuals	1	17
EPT richness	1	5
%EPT	0.5%	7%
Clinger individuals	12	33
Clinger richness	7	13
%Clinger	6%	14%

Change in Lentic and Lotic-erosional Taxa at Two Upstream Impounded Sites Before and After the Removal of Simkins Dam

In terms of taxa richness and number of individuals, lentic organisms were more abundant than lotic-erosional organisms at the impounded sites prior to the removal of Simkins Dam (Pre). During this same period, lotic-erosional taxa and individuals were more abundant than lentic at the control site, 510. These results are shown in Table 8.2. With the removal of Simkins Dam, lotic-erosional taxa and individuals became more abundant than lentic taxa during the Post1 and Post2 periods at the formerly impounded sites while lotic-erosional taxa remained more abundant at the control site.

Table 8.2: Number of lentic and lotic-erosional taxa and individuals collected during Pre, Post1, and Post2 periods at two sites upstream of Simkins, B08 and B09, and a control site, 510.

	Lentic taxa	Lotic-erosional taxa	Lentic individuals	Lotic-erosional individuals
Pre				
B08	14	11	201	115
B09	16	9	212	172
510	11	20	197	222
Post1				
B08	17	28	85	161
B09	13	22	47	122
510	16	29	105	205
Post2				
B08	22	33	139	192
B09	17	26	125	202
510	24	34	139	195

Shannon Wiener Diversity Index

Diversity index values calculated for each site during each period are shown in Table 8.3. With the exception of B02, the lowest diversity value prior to the Simkins Dam removal was found at sites upstream of Simkins and Union dams, with the lowest at site 504 (1.30). The highest diversity value during the Pre period was 2.80 at B05, the first site upstream of Bloede Dam. In the two years following the removal of Simkins Dam, B08, the first site upstream of Simkins Dam, had the highest diversity value in 2011 (Post1) and second highest in 2012 (Post2). B05 (highest Pre diversity) was the only site that exhibited a decrease in diversity from Pre (2.80) to both Post 1 (2.40) and Post 2 (2.68). Mean diversity values for the groupings in Table 8.3 (i.e., Downstream of Bloede Dam) all increased from Pre to Post1 and Post2. These increases were greatest at the sites upstream of Simkins Dam during Post1 and Union Dam during Post2. With the exception of B08, B09, and B10, the habitat at these sites remained dominated by cobble/gravel throughout the study. B08, B09, and B10 shifted from sand to cobble/gravel after the removal of the dams. The benthic macroinvertebrate results observed at B02 did not follow any pattern that was expected.

Table 8.3: Shannon-Wiener diversity index values for macroinvertebrate community composition at Pre, Post1, and Post2 Patapsco River sites.

	Downstream of Bloede Dam				
	B01	B02	B03	B04	501
Pre	2.09	1.76	2.46	2.13	2.28
Post1	2.08	2.38	2.80	2.70	2.97
Post2	2.47	3.10	2.29	2.42	2.78
	Between Bloede and Simkins Dam				
	B05	B06	502	B07	
Pre	2.80	2.22	2.54	2.20	
Post1	2.40	2.77	2.72	2.71	
Post2	2.68	2.65	2.46	2.81	
	Upstream of Simkins Dam				
	B08	B09	B10	504	507
Pre	1.74	1.96	1.66	1.30	2.44
Post1	3.01	2.69	2.93	2.40	2.75
Post2	3.07	2.69	2.91	1.86	2.65
	Upstream of Union Dam				
	508	510			
Pre	1.84	1.85			
Post1	2.57	2.83			
Post2	2.83	2.97			

Macroinvertebrate diversity values increased at all but four sites downstream of Simkins Dam during the Post1 – Pre and Post2 – Pre periods (Fig.8.12 and Table 8.3). During the Post1 – Pre period, diversity values increased at all sites with the exception of B05, a site in between Bloede and Simkins dams. Sites upstream of Simkins Dam recorded the highest diversity value increases. Three declines in diversity values occurred during the Post2 – Pre period. These sites were all downstream of Simkins Dam, two of which (502 and B05) were sites immediately below Simkins Dam. The majority of the greater diversity increases during this period were at sites upstream of Simkins Dam, a site upstream of Union Dam, and the control site, 510. The largest increase occurred at B02, the second furthest downstream study site. B02 and B01, the furthest downstream site, had the largest increases in diversity values in comparing Post2-Pre values to Post1-Pre values (Fig.8.12). The largest declines in comparing these periods were observed at 504 (upstream of Simkins Dam) and B03 (downstream of Bloede Dam).

Diversity values at the control site were among the highest, ranging from 1.85 (Pre) to 2.97 (Post2) (Table 8.3 and Fig.8.12). Four sites during either the Post1 – Pre or Post2 – Pre period had a diversity value higher than that of the control site during the same timeframe: B02 (Post2-Pre), B08 (both periods), B10 (both periods), and 504 (Post1 – Pre).

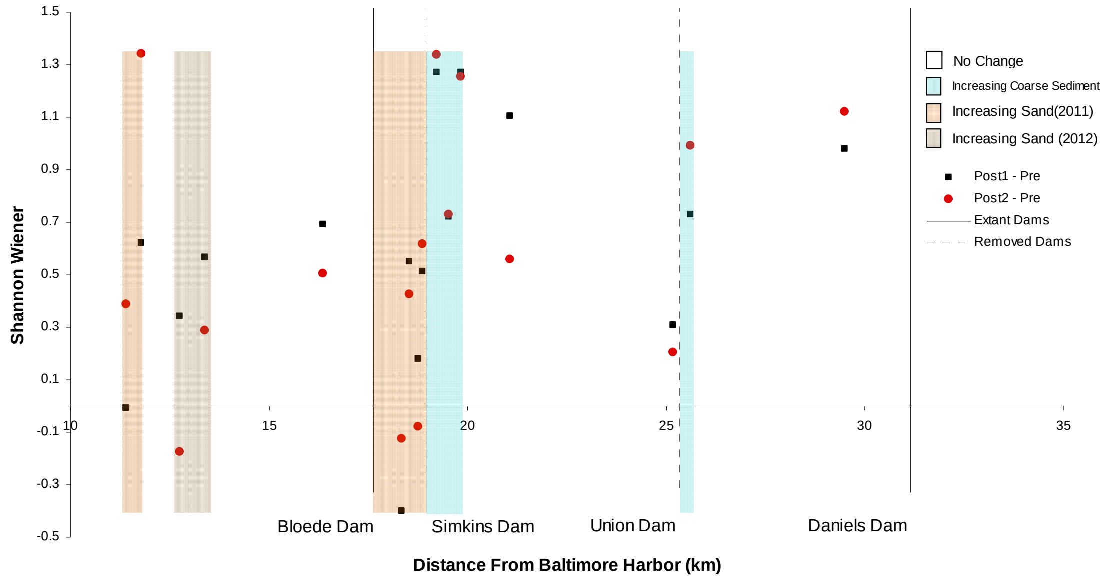

Figure 8.12: Changes in Shannon-Wiener diversity index values for macroinvertebrate community composition at Patapsco River sites during the Post1 - Pre and Post2 - Pre time periods.

Discussion

The observed changes in macroinvertebrate communities appear to be associated with shifts in substrate that occurred during the study period. The initial shift was likely a result of the removal of Union and Simkins dams in 2010. Flood flows created by two tropical storms in the fall of 2011 further contributed to the movement of sediment. The removal of the dams appeared to be the primary factor in the shift from sand to cobble/gravel habitat at sites upstream of Simkins and Union dams and a shift from cobble/gravel to sand at sites downstream of Simkins Dam. Changes in metrics that were either substantial increases or decreases from the change recorded at the control site were likely due to the dam removals when comparing the Post1 assemblage to the Pre assemblage. The substantial changes observed when comparing the Post2 assemblage to the Pre assemblage were likely due to both the removal of dams and flooding caused by two tropical storms. A substantial deviation in the value of a given metric observed at non-control sites from that observed at the control site when comparing Post2 – Pre to Post1 – Pre was likely a result of the 2011 tropical storms as these changes in the metric occurred from the time the site was sampled in 2011 to the 2012 sampling date. The changes in %Clinger that occurred when comparing Post2 – Pre to Post1 – Pre illustrate this point (Fig.8.9).

A number of macroinvertebrate community metrics were analyzed during this study. For the most part, the most notable changes occurred at the four sites upstream and downstream of Simkins Dam; sites at which dramatic shifts in dominant substrate occurred. Increases in EPT richness (Fig.8.6) and %EPT (Fig.8.7) at sites upstream of Simkins Dam appear to be associated with the removal of the dam and the subsequent change in dominant habitat from sand (pre-removal) to cobble/gravel (post-removal). This dominant habitat shift was likely the cause of the decreases in %Non-insect (Fig.8.3) and %Burrower (Fig.8.8) as well as the BIBI changes (Fig.8.2) at these same sites.

The large %EPT deviations from the values observed at the control site occurred at sites where shifts in habitat were recorded and appear to be a result of the removal of the dams. Furthermore, %EPT and other metrics primarily controlled by insects, such as %Clinger, appear to have been positively affected by the major non-insect losses that occurred at sites after the removal of the dams.

Changes in macroinvertebrate metrics due to shifts in dominant habitat were most likely due to the habitat becoming more suitable for taxa with morphological adaptations suited for the newer habitat. For example, the substantial increases in EPT at the sites immediately upstream of Simkins Dam following its removal are likely attributable to the increase in more heterogeneous cobble/gravel habitat- habitat more suitable to the majority of taxa in these groups. Likewise, the large decreases in burrowing taxa, primarily chironomids and fingernail clams, and non-insect taxa, primarily amphipods, were likely a result of the new cobble/gravel habitat being less suitable than the previous habitat. The burrowing taxa require loose sediments in which to burrow while the amphipods are more suited to slower-moving depositional areas where their primary food source, leaf litter, can accumulate. These organisms were likely either flushed downstream with the sediments or relocated to more suitable habitat after the dam was removed. A plausible explanation for the greater %Burrower decrease at sites upstream of Simkins Dam during Post2 – Pre is that the two tropical storms and subsequent flooding events that occurred in September 2011 pushed fine sediments further downstream, decreasing suitable burrowing habitat at sites upstream of the shaded blue areas and further increasing suitable habitat in the area downstream of Simkins Dam (tan shaded area in Fig.8.8).

A number of taxa were primarily responsible for the increase or decrease of a metric at sampling sites during the study. The majority of these "driver" taxa were among the most abundant macroinvertebrates collected during this study. Major decreases or increases in these taxa led to substantial decreases or increases in the metrics at a site. Baetid mayflies, primarily *Baetis*, were the most numerous mayfly taxa collected and as a result were the primary drivers in changes in %EPT (Fig.8.7), %Clinger (Fig.8.9), and %Scraper (Fig.8.11). Amphipods, primarily *Gammarus*, were the second most collected taxonomic group and drove changes in %Non-insect, %Shredder, and %Sprawler (Figs.8.2-8.4). *Orthocladius*, a chironomid, and fingernail clams in the family Pisidiidae drove changes in %Burrower (Fig.8.8). *Orthocladius* and other chironomids also resulted in changes in %Scraper and %Filter-feeder (Fig.8.10). *Chimarra*, *Cheumatopsyche*, *Ceratopsyche*, and *Hydropsyche* were the most abundant caddisfly taxa collected during this study. These taxa figured prominently in changes in %Filter-feeder, %Clinger, and %EPT. Even though beetles were not numerically abundant, substantial decreases or increases in the number of elmids (riffle beetles) drove %Scraper at a few sites.

As has been reported in other studies (Bushaw-Newton et al. 2001, Hart et al. 2001, and Stanley et al. 2002), sites impounded by Simkins Dam had a unique assemblage composed of more lentic taxa when compared to the free-flowing reference/control site at which more lotic-erosional taxa were present. The control site in this study, 510, had higher EPT and clinger percentages, richness, and number of individuals than the impounded site, B08 (Table 1). As expected, after removal of Simkins Dam lotic-erosional taxa became more abundant than lentic taxa at the formerly impounded sites, B08 and B09 (Table 2). During this period, the taxa and number of individuals of lentic and lotic-erosional groups remained relatively consistent at the control site. The relative consistency in the lentic and lotic-erosional data at the control site indicate that changes observed at the formerly impounded sites appear to be associated with the removal of Simkins Dam and the change in habitat from sand to cobble/gravel- a change that provided habitat more suitable to lotic-erosional taxa and less suitable to lentic taxa.

Prior to the removal of Simkins Dam, benthic macroinvertebrate diversity was highest at B05 (immediately upstream of Bloede and dominated by cobble/gravel), and with the exception of 507 (immediately downstream of Union Dam and dominated by cobble/gravel), diversity values at sites upstream of Simkins and Union dams were lower than all but one site downstream of Simkins (Table 3). After Simkins Dam was removed, the highest increases in diversity values during the Post1 and Post2 periods were recorded at sites upstream of Simkins Dam, including the two sites upstream of Union Dam, and one site below Bloede Dam, B02 (Fig.8.12). The habitat at these sites, with the exception of B02, either shifted from sand to cobble/gravel after the removal of the dams or was dominated by cobble/gravel throughout the study. The dominant habitat at B02 remained sand during the study and the results observed at this site did not follow any pattern that was expected. The highest increase from Pre to Post1 and second highest increase from Pre to Post2 occurred at B08, the first site upstream of Simkins Dam and the site at which diversity was the third lowest prior to the removal of Simkins Dam. During the study period, diversity increased from Pre to Post1 or Post2 at all sites except for B05. The higher-than-control diversity increases during the Post1 – Pre period appear to be associated with the removal of the dams while those occurring during Post2 – Pre could be attributed to the dam removals and flooding caused by the two storm events. The declines at B05 appear to be associated with the shift from cobble/gravel habitat (pre-removal) to sand (post-removal).

Investigating the impact the removal of Union and Simkins dams had on the macroinvertebrate community in the Patapsco River had Bloede Dam also been removed would have provided a more accurate assessment of the changes that occurred downstream of Union and Simkins. However, comparing the results of this study to the results of a similar study examining the macroinvertebrate community before and after the removal of Bloede Dam could be of great interest to the scientific community, especially those interested in the impacts of dam removal. We would expect the results we observed upstream of Simkins Dam during this study to be similar to those that would occur at the sites between Bloede and Simkins dams if Bloede Dam were to be removed.

Continual monitoring of the macroinvertebrate communities at these Patapsco River sites is strongly recommended. As previously mentioned, few studies have examined the long-term response of these communities to dam removal. The two years of post-dam removal data offer useful insights into how the river and biota have responded after disturbance from dam removals. A longer-term study (at least five more years) will significantly increase the knowledge of the macroinvertebrate community response to dam removals and allow for more accurate predictions by scientists examining future changes in the macroinvertebrate community in systems where dam removal is scheduled.

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Appendix 8.1. **Explanation of Metrics**

Benthic Index of Biotic Integrity

The Benthic Index of Biotic Integrity (BIBI) is a biotic index generated from macroinvertebrate data that takes into account taxonomic richness, taxonomic composition, tolerance values, and trophic/feeding groups in an effort to provide an estimate of the biological integrity of the site sampled (Stribling et al. 1998, Southerland et al. 2005). Using macroinvertebrates provides a more integrated estimate of the integrity of a site and is commonly viewed as preferable to a one-time water sample used for chemical analysis. BIBI scores range from 1-5. Higher scores indicate a healthier benthic macroinvertebrate community. A score of 1-1.9 indicates Very Poor stream biological integrity, 2-2.9 indicates Poor, 3-3.9 indicates Fair, and 4-5 indicates Good.

%Non-insect, %Sprawler, and %Shredder

Non-insect taxa commonly found in streams are crustacean amphipods, aquatic worms (e.g., tubificids, lumbriculids), snails, and bivalves (e.g., *Corbicula*, fingernail clams, and mussels). In the Patapsco, amphipods were the dominant non-insect taxa collected during this study. *Gammarus*, the dominant amphipod genus, and *Caecidotea* and *Crangonyx* were the amphipod genera collected. *Gammarus* is a sprawling taxon that is considered a shredder that primarily feeds on leaf litter. Because of this, these organisms are typically found in areas where leaf litter accumulates in debris dams and, more likely, in depositional areas.

EPT Taxa Richness and %EPT

EPT refers to the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies). These organisms are relatively intolerant of pollution and EPT taxa richness is generally higher in lotic areas (Merritt et al. 2008). EPT is a commonly used parameter in studies investigating water quality, disturbances, and general benthic macroinvertebrate community composition. Relative to the Ephemeroptera and Trichoptera, few plecopterans were collected in the Patapsco during this study. Because of this, changes in these two parameters occurred as a result of changes in the number of individuals and taxa of mayflies and caddisflies.

%Burrower

Burrowers are organisms typically found burrowed in fine sediments. Gomphid odonates, dipterans, especially chironomids, and bivalves, such as mussels and clams, are burrowing taxa common in many rivers and streams. For studies using a sampling technique that targets the "most productive habitat", such as that used by MBSS, burrowing taxa are typically only abundant in samples collected from areas with an abundance of fine sediments, primarily sand in the case of the Patapsco.

%Clinger

Organisms described as clingers have morphological adaptations designed to attach to hard substrate, typically in riffles and faster-flowing areas of streams. Common clinger taxa include mayflies, stoneflies, elmid beetles, and numerous net-spinning caddisfly taxa such as hydropsychids. Clingers are generally more abundant in areas with rocky substrate which provide an abundance of attachment surfaces.

%Filter-feeder

Filter-feeders are organisms that construct nets or use morphological adaptations to capture food particles in the water column. Historically, filter-feeders stood alone as a FFG group. However, recently this group has been placed along with collector-gatherers into a collector-filterer group. For the purpose of showing the possible impact dam removal has on filter-feeders, this group was separated from collector-gatherers for this study.

%Scraper

Scraper taxa have mouthparts adapted to scraping, or grazing, substrate, typically a hard surface such as rock, for periphyton and other organic material. Most scrapers also have morphological adaptations that allow them to maintain position in fast-flowing areas. Because of these adaptations and a need for hard surfaces, scrapers are generally most abundant in areas with exposed rocks.

Change in Lentic and Lotic-erosional Taxa at Two Upstream Impounded Sites Before and After the Removal of Simkins Dam

Lentic taxa are adapted to habitats dominated by fine sediments and slower-moving or standing water. These taxa are typically more abundant than lotic-erosional taxa (taxa typically found in faster-flowing areas of streams) in impounded areas, such as those created by a dam. With the removal of a dam, the formerly impounded area should become more suitable to lotic-erosional taxa and less suitable to lentic taxa, due to the exposure of gravel, cobble, and/or boulder substrate caused by the increased current velocity and resultant dispersal of fine sediments downstream.

Chapter 9: Biotic and Abiotic Conditions in the Patapsco River Following the Removal of Simkins and Union Dams: Is the Post-removal River Now Suitable for Eastern elliptio?

Introduction

Freshwater mussels are a diverse group of filter feeding bivalves that provide important services in aquatic ecosystems (Vaughn and Hakenkamp 2001, Vaughn et al. 2008). They are sensitive to a variety of habitat, water quality, and landscape alterations (Bogan 1993, Box and Mossa 1999, Watters 2000). Consequently, many mussel species are imperiled and even common species are in decline (Williams et al. 1993). A major reason for their decline is the ubiquitous damming of rivers and streams (Taylor and Vaughn 1999). Dams alter flow regimes, lotic habitat, and sediment cycles, with consequences that cascade throughout the ecosystem (Poff and Hart 2002). Furthermore, dams fragment the distribution and restrict movement of stream fishes that serve as intermediate hosts in the reproductive cycle of freshwater mussels (Watters 1996).

The eastern elliptio, *Elliptio complanata* (Lightfoot 1786), is a globally common mussel distributed throughout Atlantic and Great Lakes drainages in a variety of habitats (Matteson 1948). Where present, it often dominates the mussel community and can be found in dense aggregations (≈20 mussels/m²) that constitute considerable amounts of stream biomass (Kat 1982). Cumulatively, the aggregations have the ability to filter substantial volumes of water, cycle nutrients and sediment, stimulate primary and secondary production, and stabilize stream substrate (Vaughn and Hakenkamp 2001, Vaughn et al. 2008). In parts of the Chesapeake Bay basin, the distribution of *E. complanata* has been reduced and populations show little reproduction (Strayer and Fetterman 1999). This phenomenon may be due to a lack of its primary host fish, American eel, since conditions are suitable for other mussels (Galbraith et al. *In review*). The loss of migratory fishes and mussels caused by dams has likely had tremendous consequences on stream ecosystem function (Freeman et al. 2003).

The Patapsco River is the only Chesapeake Bay basin in Maryland that lacks current records of live *E. complanata* (Ashton 2010a). Its distribution in the river is represented by a single large, weathered valve (Harbold 2012). Other mussel surveys in the Patapsco River documented a small population of *Alasmidonta undulata* (Ashton 2010b), which is tolerant to conditions created by impoundments and is considered a fish-host generalist (Clarke 1981). These surveys failed to encounter *E. complanata*. They likely once existed in the Patapsco River given the prevalence of mussel shells, including *E. complanata*, at pre-historic sites near the river and throughout the Chesapeake Bay (Stearns 1949, Luckenbach 2011). The river has a history of agricultural and industrial pollution that has degraded water quality, habitat, and biological communities (MDE 2005, 2009a, 2009b). It has also been called one of the most dammed rivers in the United States (Travers 1990). Presumably, these blockages collectively impeded the distribution of American eels within the river (e.g., Machut et al. 2007), a situation that would have disrupted the reproductive cycle of *E. complanata* and contributed to its decline (e.g., Kelner and Sietman 2000, McNichols et al. 2011). Conditions measured annually in the Patapsco River have improved since the onset of Clean Water Act of 1972 (Friedman 2009). The prevailing belief is that dam removal in the Patapsco River can further assist in its ecological recovery by reconnecting lotic habitat, affording migratory fish better passage, and improving water quality (e.g., Gregory et al. 2002).

It is plausible that dam removal is a viable tool for restoring freshwater mussels given the numerous studies that highlighted impacts of dams (e.g., Watters 1996, Vaughn and Taylor 1999, Dean et al. 2002, Baldingo et al. 2004, Tiemann et al. 2007). For example, Smith (1985) observed expansion of *Anodonta implicata* in the Connecticut River following the implementation of passage for alosines, which are the fish-hosts of *A. implicata*. However, there is little information on the efficacy of dam removal as a restoration technique for mussels. Recent studies indicate that the removal of small, intact dams may have little to no benefit to freshwater mussels (Sethi et al. 2004, Gangloff et al. 2011). In particular, the immediate habitat disturbances created by some types of dam removal (i.e., stranding, suspended and deposited sediments, and head cutting) are especially harmful to mussels (Hartfield 1993, Box and Mossa 1999, Sethi et al. 2004). By no means should this preclude dam removal where mussels are present. Nevertheless, it highlights the need to 1) assess the potential for impacts at various spatial, temporal, and faunal scales, 2) conduct pre- and post dam removal monitoring, 3) consider various dam removal methods, and 4) set reasonable goals so trade-offs between ecological costs and benefits can be adequately addressed (Poff and Hart 2002, Stanley and Doyle 2003).

Given the predicted improvements to the stream ecosystem and migratory fish passage from dam removal in the Patapsco River, we compared a suite of natural and anthropogenic variables in nearby streams with and without *E. complanata* to the Patapsco River to determine if post-dam removal conditions in the river may now be suitable for this mussel species. We evaluated hypotheses regarding factors that might potentially limit the distribution of *E. complanata* in the Patapsco River and the potential for its reintroduction following dam removal by 1) comparing host-fish availability at monitoring sites in the Patapsco River to other sites where *E. complanata* was present, 2) characterizing patterns in abiotic conditions coincident with *E. complanata*, and 3) contrasting abiotic conditions among sites in the Patapsco River (pre- and post-dam removal) with sites where *E. complanata* was present and where it was apparently absent.

Methods

Study sites

Physiochemical parameters and American eel abundance were measured at sites using standard protocols employed by the Maryland Biological Stream Survey (Klauda et al. 1998, Stranko et al. 2007). MBSS sites sampled within the Piedmont physiographic province were then paired to locations of freshwater mussel surveys conducted by the MBSS, Maryland Natural Heritage Program, and the U.S. Fish and Wildlife that detected *E. complanata*. Because mussel surveying techniques and sampling effort differed among surveyors, these data were limited to presence-absence. This data set of co-located sites was then used to identify other sites in the region where *E. complanata* was absent, but presumably once existed. We screened these sites from all Piedmont sites by using the minimum or maximum observations of several natural determinants of mussel distribution measured at sites with *E. complanata*, including watershed size (ac), wetted stream width (m), gradient (% slope), and discharge (cfs) (Sepkowski and Rex 1974). In GIS, we then determined if any potential site was located upstream from a stream blockage that impedes American eel distribution (>12.2-m-high, Wiley et al. 2004) and excluded them from further analyses because mussel absence could be due to a lack of eels and not an abiotic factor.

Sites in the Patapsco River (N = 7) were monitored annually using the same methods (Stranko et al. 2007) as other Piedmont streams for two years prior to dam removal (2009, 2010) and two years following dam removal (2011, 2012). At the onset of the study, four dams (6.1 to 8.2 m in height) were present in the Patapsco River study area from river kilometer (Rkm) 17.7 to 31.1. At least 11 comparable low-head dams once existed within the study area (Travers 1990). The two most downstream dams (Bloede and Simkins) had fish passage structures and the furthest upstream dam (Daniels) has a fish ladder. The third dam (Union) was breached in 1972 by Hurricane Agnes. This presumably permitted the movement of some fishes, but was a safety hazard. Detailed descriptions of the dam removal processes and monitoring of the response to removal in stream communities and habitat can be found in other chapters of this report.

Statistical analysis

A combination of uni- and multivariate techniques were used to compare and contrast stream conditions to determine if the Patapsco River may now be suitable for *E. complanata*. First, we tested whether *E. complanata* distribution in the Patapsco River might be limited by a lack of host-fish, using analysis of variance (ANOVA) with Tukey's studentized range test to investigate differences among means of host-fish density. We also examined spatio-temporal patterns in host-fish density in relation to dam removal and compared it to the host-fish density at Piedmont sites where *E. complanata* was present. Secondly, we used principal components analysis (PCA) to characterize intercorrelations in abiotic conditions across Piedmont streams with *E. complanata*. Physiochemical data were rotated (orthogonal varimax) and normalized (Kaiser criterion) to maximize the variance explained among principal components. We retained components with eigenvalues >1.0 and loaded variables onto component axes with factor coefficients $\geq|0.60|$. We again used PCA to illustrate patterns in abiotic conditions among classes of stream monitoring sites (i.e., the Patapsco River before and after dam removal and Piedmont stream sites with and without *E. complanata*). We then used post-hoc ANOVAs with Tukey's post hoc analyses to examine differences in standardized physiochemical-based PC scores across stream classes.

Results

Host availability

American eels were collected at all Piedmont stream sites (N = 27) where *E. complanata* was also encountered, although eel density (mean ± 1 SD) was variable among sites (0.07 ± 0.06 eels/m²). Additionally, eels were collected at each Patapsco River dam removal monitoring site (N = 7) throughout the study. Prior to dam removal, the mean American eel density at these sits in the Patapsco River was similar to the mean of Piedmont sites with *E. complanata* (Fig. 9.1). In the two years following dam removal, eel density throughout the study area was not significantly different than Piedmont stream sites or pre-dam removal densities ($F = 0.27$, $p = 0.77$). Although we observed no significant difference in eel density after dam removal in the Patapsco River, we saw minor spatial changes over time at individual monitoring sites. For example, at sites in the lower reaches of the dam removal study area, eel density was comparable or higher than mean eel density of Piedmont streams with *E. complanata* (Fig. 9.2). Conversely, eel density was generally lower at sites in the middle and upper reaches of the study area. Density increased slightly in the middle reach following dam removal to the point where it equaled the mean eel density of Piedmont streams with *E. complanata*.

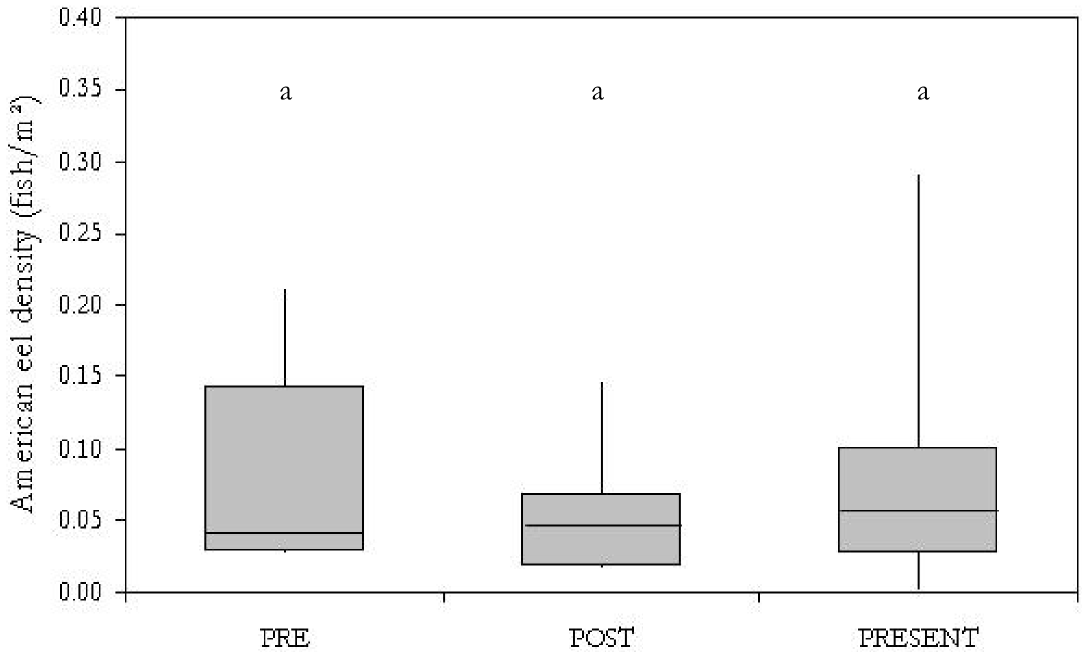

Fig. 9.1. Mean (± 95% CI) American eel density (eels/m²) observed in Piedmont streams where *E. complanata* was present compared to eel density in the Patapsco River, pre- and post-dam removal. Significant differences ($p < 0.05$, Tukey's studentized range test) among stream classes are indicated by different letters above the box-and-whisker plots.

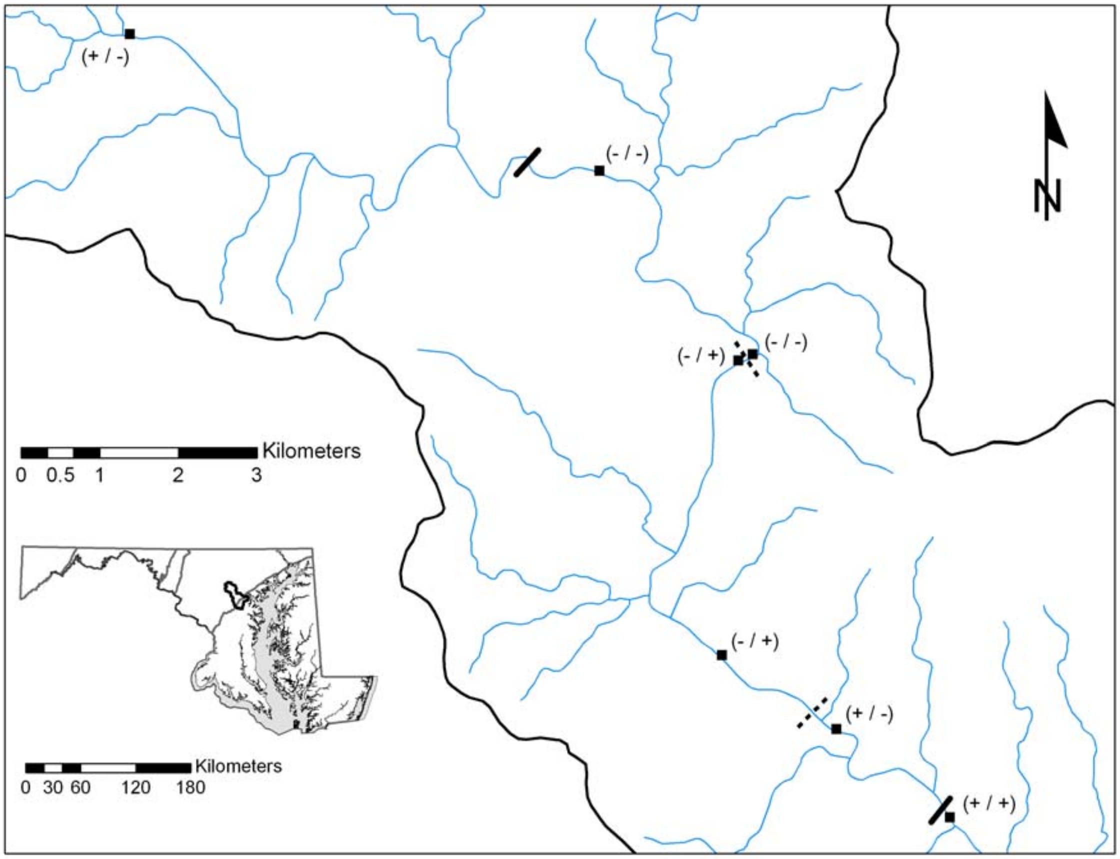

Fig. 9.2. Temporal comparison of the mean American eel density in Piedmont streams where *E. complanata* was present (0.07 ± 0.06 fish/m²) to mean eel density at dam removal sites in the Patapsco River. A plus (+) indicates mean eel density at a site for a given period (pre-removal/post-removal) in the Patapsco River was higher than the mean of Piedmont streams with *E. complanata* and a minus (-) indicates density was lower for a given period. The status of dams as intact or removed are indicated by solid (Bloede and Daniels) or dashed (Simkins and Union) lines, respectively.

Abiotic conditions coincident with E. complanata

Piedmont streams where *E. complanata* was present were generally wide, of slight gradient, moderate discharge, and located in large catchments (Table 9.1). Using minimum or maximum observations of physical characteristics that could limit mussel distribution, we identified comparable Piedmont stream monitoring sites where *E. complanata* were not encountered (N = 61), none of which were upstream of blockages to eel movement. We then compared American eel density and physiochemical variables measured at sites where *E. complanata* was present to stream sites where they were absent to look for factors that may explain their distribution and aid in interpretation of subsequent analyses. Mann-Whitney *U*-tests indicated that medians of specific conductance, acid neutralizing capacity, and SO_4 were significantly lower ($z \leq -2.64$, $p \leq 0.008$) at Piedmont stream sites with *E. complanata* than sites without. NO_3-N concentration was also significantly lower at these sites ($z = 2.76$, $p = 0.006$). Physical habitat metrics did not differ between Piedmont stream classes. Although eels were collected at most sites where *E. complanata* was absent, eel density was significantly lower in comparison to sites where *E. complanata* was present ($z = 3.15$, $p = 0.002$).

Table 9.1. Characteristics of Piedmont streams where *E. complanata* was present (N = 27).

Variable	Mean ± 1 SD	Minimum	Maximum
Average wetted width (m)	15.39 ± 6.54	6.30	---
Gradient (% slope)	0.36 ± 0.37	---	1.50
Discharge (cfs)	38.68 ± 34.09	3.80	---
Catchment size (ac)	35,096 ± 28,686	6,427	---

Suitability of environmental conditions

Principal components analysis of abiotic conditions at sites where *E. complanata* was present revealed three PCs with eigenvalues >1.0 (Table 9.2). Together, these PCs accounted for 72.7% of the variation in abiotic conditions across sites. The first component accounted for 43.5% of the variation among sites and was a surrogate for high quality physical habitat as it relates to fish and benthic macroinvertebrate communities. The second PC accounted for 15.5% of the variation among sites and was indicative of high specific conductance often associated with streams in urbanized landscapes. PC3 was a surrogate for streams having heterogeneous velocity and depth regimes with complex riffle-run habitat and accounted for 13.8% of among site variation.

Table 9.2. Factor loadings and the variance accounted for in retained principal components (PCs) from the physiochemical dataset of Piedmont streams where *E. complanata* was present. Factors that loaded onto component axes are in bold.

Physiochemical variable	PC1	PC2	PC3
pH	-0.059	0.018	-0.067
Conductivity	-0.188	**0.963**	-0.025
Acid neutralizing capacity	-0.390	**0.834**	-0.184
Dissolved organic carbon	-0.086	0.164	-0.124
Sulfate	-0.291	0.530	-0.176
Nitrate-Nitrogen	0.102	-0.172	-0.163
Instream habitat	**0.867**	-0.287	0.147
Epifaunal substrate	**0.878**	-0.304	0.241
Velocity-depth diversity	0.288	-0.117	**0.898**
Pool-glide quality	0.011	-0.024	0.062
Riffle-run quality	0.555	-0.108	**0.630**
Riffle embeddedness	**-0.695**	0.194	-0.390
% variance (% cumulative)	43.5	15.5 (59.0)	13.8 (72.7)

We used seven of the original 12 physiochemical variables (Table 9.2) to further examine patterns across classes of Piedmont streams (i.e., mussel present, mussel absent, pre-dam, and post-dam) as they relate to abiotic condition, *E. complanata* distribution, and dam removal. Principal components analysis of the reduced set of variables revealed two PCs with eigenvalues >1.0. These components explained a majority of the variance (66.4%) in abiotic conditions across sites. The first PC accounted for 44.7% of the variance among sites and loaded with instream habitat ($r = 0.953$) and epifaunal substrate ($r = 0.687$) scores (Fig. 9.3). Higher scores of both habitat metrics indicate a predominance of cobble-boulder substrate, woody debris, and root wads. The second component accounted for 21.7% of variation among sites and loaded with percent riffle embeddedness ($r = 0.922$). Water chemistry parameters were not important factors to explaining the variability in conditions across stream classes.

Composite habitat metrics of Piedmont streams where *E. complanata* was present overlapped with streams where it was apparently absent, including most sites in the Patapsco River located upstream of the dam removal study area (Fig. 9.3). However, sites with mussels were generally not located in the multivariate space representing low (<15%) riffle embeddedness, unlike some sites where *E. complanata* was not encountered. There also was substantial overlap in conditions between these groups of sites and sites in the Patapsco River prior to and following dam removal. Pre-dam removal sites were located in multivariate space representing slightly higher instream habitat and epifaunal substrate scores (PC1) compared to post-dam removal sites. Pre-dam removal sites were also slightly shifted towards lower riffle embeddedness scores (PC2) compared to post-dam removal sites. The abiotic conditions represented in component scores (PC1, PC2) did not significantly differ among pre-dam removal sites, post-dam removal sites, Piedmont stream sites with *E. complanata*, and sites without ($F = 0.70$, $p = 0.55$ and $F = 1.44$, $p = 0.23$).

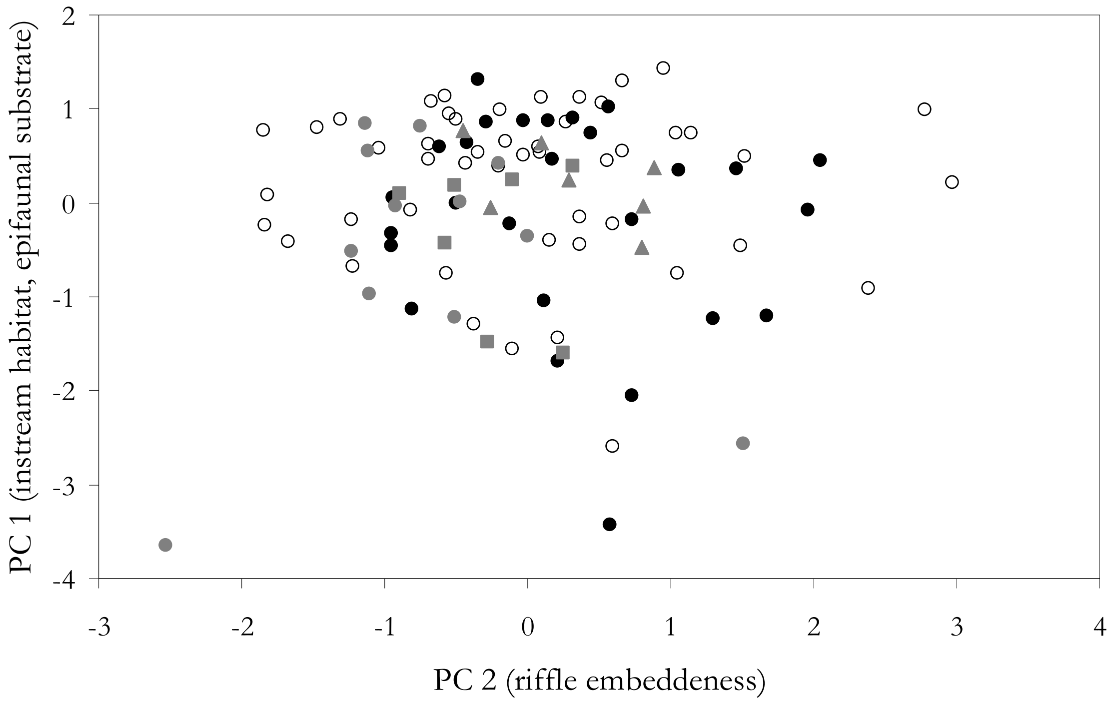

Fig. 9.3. Principal component plot of physiochemical variables measured at sites *with E. complanata* (closed circle), without *E. complanata* (open circle), pre-dam removal (grey triangle), post-dam removal (grey square), and sites in the Patapsco River upstream of the dam removal study area (grey circles).

Discussion

Biotic and abiotic influences

Compared to parts of southern Maryland and the eastern Coastal Plain, *E. complanata* is found in relatively few Piedmont streams (Ashton 2010a). This is of note considering they were historically found throughout streams of the Chesapeake Bay (Matteson 1948, Strayer and Fetterman 1999). The exact cause of this loss is not known, but can likely be attributed to a combination of factors including land alteration and resulting degradation in water quality, habitat, and lack of suitable host-fish (Bogan 1993, Watters 1996, Box and Mossa 1999, Watters 2000). For example, some Piedmont watersheds in Maryland have large dams that completely block eel migration (Wiley et al. 2004) and lack current records of *E. complanata* presence (Ashton 2010a). Others have found the current distribution of some mussels related to the disappearance of their primary host-fish (Watters 1996, Kelner and Stietman 2000, Baldingo et al. 2004, McNichols et al. 2011). The relationship found in this study between American eel density and *E. complanata* presence lends support to a lack of primary host as a potential cause for their decline. In addition, water quality (ion and sulfate concentration) was related to streams with extant populations of *E. complanata*, although their long-term viability is not known nor is how these abiotic factors relate to host-fish density. Interestingly, these streams also had concentrations of baseline nutrients (NO_3-N) that are generally considered to impair aquatic life (Morgan et al. 2007), although they may not influence mussel distribution (Strayer and Fetterman 1999). The relationships between *E.*

complanata distribution and water quality seem to correspond with differential land use patterns (agriculture versus urban) within the Piedmont province of Maryland and deserve further investigation. Our findings also suggest that physical habitat quality (instream habitat and epifaunal substrate) does not play a major role in the distribution of *E. complanata* in Maryland's Piedmont streams, which corresponds with a general consensus regarding the influence of proximal factors on mussels (Strayer 2008).

Regardless of abiotic condition, nearly all Piedmont streams where *E. complanata* was encountered also had dense populations of their primary host-fish, American eel. In fact, eel densities at these sites were often two or more times greater than the average density for the Upper Chesapeake Bay (Wiley et al. 2004). The association between eel density and mussel presence in this study support recent laboratory studies that found eels were the primary host of *E. complanata* in streams of the Chesapeake Bay basin (Galbraith et al. *In review*). Host quality has also been related to the status of other mussels (e.g., McNichols et al. 2011). Since American eel density in the Patapsco River prior to dam removal was equal to the density in Piedmont streams where *E. complanata* was present, it seems unlikely that freshwater mussel absence in the Patapsco River was at least recently due to a lack of their primary host. More likely, a combination of stressors over decades including disruption of the host-fish relationship and severe water quality, habitat, and landscape degradation that followed European settlement and continued through industrialization and suburbanization (Travers 1990, MDE 2005, 2009a, 2009b) led to an irreversible decline and ultimate extirpation of *E. complanata* from the Patapsco River (e.g., Tilman et al. 1994).

Effects of dam removal on mussels

The short-term effects of dam removal on streams and their aquatic communities have been well documented (Bednarek 2001, Doyle et al. 2005). In part, the apparently pronounced effect dam removal can have on mussels as opposed to benthic macroinvertebrates can be attributed to differences in organism response and spatio-temporal characteristics of mussel populations (Sethi et al. 2004). Unfortunately, we have no data to infer the potential effects from dam removal in the Patapsco River on its extant mussel population. It is possible that a small, undocumented population of *A. undulata* was adversely affected by the release of sediment that accumulated behind the impoundments, since live mussels were encountered ≈1 Rkm upstream of Union Dam and shells of dead mussels were found well downstream of both Union and Simkins dams prior to their removal (Ashton 2010b). Conversely, these shells could have deposited into the dam removal area from known upstream populations. We feel this finding highlights the need to conduct appropriate pre-removal monitoring when information gaps exist. Further evaluation on the effects of dams and their removal over multiple spatial and temporal scales will be critical to prioritize mussel conservation efforts, especially when blockages disrupt dispersal abilities of fish-hosts and population dynamics of mussels (e.g., Smith 1985, Watters 1996, Kelner and Seitman 2000). A better understanding will be particularly important because some small dams may enhance mussel populations and habitat in downstream reaches (Baldingo et al. 2004, Gangloff et al. 2011).

Although dam removal has been hypothesized as a potential tool for freshwater mussel restoration (e.g., Tiemann et al. 2007), we cannot conclude at this time if it is a viable option in the Patapsco River primarily because of the apparent extirpation of *E. complanata* from the watershed. However, there are benefits resulting from dam removals that could aid future mussel restoration efforts yet to be realized. First, few eels have been encountered upstream of Daniels dam and when present their densities were very low. Therefore, the removal of the most downstream dam (Bloede) could facilitate the upstream movement of eels (e.g., Hitt et al. 2012) and elevate their densities to a level potentially necessary for successful *E. complanata* recruitment. Such an increase in eel density was observed upstream of Simkins dam immediately following its removal. Improving eel passage at Daniels Dam or moving eels upstream of the dam could accelerate this goal. These actions could be important because in addition to 14 Rkm of habitat within the dam removal area, there was up to 32 Rkm of habitat upstream of Daniels Dam where abiotic conditions were similar to Piedmont streams sites with *E. complanata*, but generally lacked eels. Secondly, *A. undulata* populations may be able to disperse downstream now that their fish-hosts (Watters et al. 1998) can move within uninterrupted lotic habitat between Daniels and Bloede dams. Thus, any short-term impact to *A. undulata* populations caused by the dam removals (e.g., burial or stranding) may be outweighed by potential long-term gains for mussels in the Patapsco River overall.

Implications for mussel restoration

Freshwater mussels have irregular reproductive cycles, low mobility, and rely upon a host-fish for their reproduction and dispersal. For successful recruitment, 1) adequate populations of fish-hosts and mussels must co-occur, 2) attachment and transformation of glochidia must produce considerable numbers of juveniles, 3) these juveniles must settle into habitat favorable for survival, and 4) the number of recruits that mature must be enough to sustain a population (Strayer 2008). In spite of apparently suitable abiotic conditions in the Patapsco River, improving the population size and dispersal abilities of American eels alone could not restore *E. complanata* populations because of their extirpation from the river and thus interruption of the host-parasite relationship. The migratory pattern of eels and sensitivity of mussel glochidia to salt (Gillis 2011) also precludes recolonization from other Chesapeake Bay populations. The restoration of *E. complanata* in streams like the Patapsco River will depend on active management strategies (e.g., reintroduction) coupled with improvements in stream connectivity for mussels and their host-fish.

Freshwater mussel relocation has been used as a management tool for over 30 years, particularly with imperiled species and as mitigation for stream impacts (Cope and Waller 1995). These efforts have recently included common species as their role in the ecosystem has been recognized along with their contribution to biodiversity (Vaughn 2010). Overall, the success of mussel restoration efforts has been quite variable (Cope and Waller 1995). This is in large part due to a lack of quantitative information on habitat requirements, which affects mussel survival and the ultimate success of relocation (Cope et al. 2003). The suitability of habitat for mussel reintroduction can be readily assessed as part of enclosure studies that measure survival, along with other important factors to consider, such as condition, growth, and optimum stocking density or size (e.g., Bolden and Brown 2002). Such information could also be transferred from other streams in Maryland supporting *E. complanata* populations after characterization of their demographics and habitat.

While host availability and abiotic conditions in the Patapsco River appear suitable for the survival of *E. complanata* (Figs 9.1, 9.3), we cannot conclude based on the available data if and where substrate composition and complex hydraulic forces (e.g., shear stress) are suitable in the river. Given that *A. undulata* persists in the river upstream of Union Dam, substrate and hydrology are also likely suitable for *E. complanata* since they co-exist elsewhere in Maryland (Ashton 2010a). The dispersal capabilities of American eel could make recolonization of *E. complanata* into areas with suitable habitat relatively rapid (e.g., Smith 1985). However, in parts of the dam removal study area, habitat may not be stable for years as restoration of geomorphic processes takes place (Gregory et al. 2002, Sethi et al. 2004). Further study of complex hydraulic factors is essential and will play a critical role in the ultimate outcome of mussel restoration efforts in the Patapsco River since they influence mussel distribution, bed formation, and recruitment (Hardison and Layzer 2001, Box et al. 2002).

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Chapter 10: Conclusions

The conclusions based on our work in the Patapsco River come from only two years of pre- and two years of post-dam removal data and observations, and should be considered preliminary. Although additional post-dam removal monitoring is needed, we have already learned a great deal about the short-term ecological response of the Patapsco River to dam removal. Based on this knowledge, we offer the following conclusions.

1. For rigorous assessments of ecosystem changes from restoration, five to ten years of pre- and post- monitoring are typically recommended (Kondolf 1995). Given this, ecological monitoring should continue for at least four more years to document the long-term ecological response to Simkins and Union dams' removals. Major ecological changes in the Patapsco River are still in progress and will likely take several years to reach a new dynamic equilibrium. Documenting changes as they occur is the best way to demonstrate the benefits of dam removal. Lessons learned from monitoring in the future will inform decisions pertaining to future fish passage and prospective dam removal projects. All the data collected so far serve as useful indicators of stream condition, and should continue to be used in future years. More than four years of monitoring may be needed to make definitive conclusions about ecological changes resulting from dam removals.

2. Most anadromous fishes are still excluded from most of the non-tidal Patapsco River due to the presence of Bloede Dam. Additionally, Bloede Dam temporarily prevents some of the sand and sediment that enters the Patapsco River from moving downstream out of the non-tidal portion of the river. Therefore, improvements to the ecological conditions of the Patapsco River will be greatly enhanced by the removal of Bloede Dam. This dam is the downstream-most blockage on the Patapsco River and the fish ladder there appears to be largely ineffective at passing anadromous fish. Removing Bloede Dam would provide unimpeded passage for anadromous fish, improve habitat for resident fish and other riverine species, and allow sediment trapped behind it to move downstream and out of the non-tidal Patapsco River. The data described in this report will provide four years of baseline data for examining the ecological benefits of Bloede Dam's eventual removal.

3. If Bloede Dam is removed, Daniels Dam may still impede the passage of migratory fishes for many miles of the Patapsco River and will be the last remaining barrier to fish movement in the mainstem Patapsco River. In lieu of removing Daniels Dam, the efficacy of its fish ladder for passing migratory fishes

could be examined. However, removal of Daniels Dam is likely to provide the most ecological benefits to the river because passage for fish would likely become entirely unimpeded at that location. However, we recognize that the potential removal of Daniels Dam will require the examination of many different factors in addition to the potential ecological benefits.

4. Freshwater mussels were once much more abundant in the freshwater streams and rivers of Maryland. They provided important ecological services and played important roles in the trophic dynamics of lotic ecosystems. The removal of dams in the Patapsco River provides improved access for the preferred host fish (the American eel) for the most common freshwater mussel in Maryland (the eastern elliptio). With improved fish passage in Patapsco River, American eel abundance is likely to increase, thus providing an opportunity to also restore the eastern elliptio to the non-tidal river. DNR's Fisheries Service, Monitoring and Non-tidal Assessment, Natural Heritage Program, and the USFWS should jointly develop and implement a freshwater mussel restoration plan for the Patapsco River. Such a plan should include: conducting surveys of current freshwater mussel distribution, identifying and prioritizing stream reaches with suitable habitat, American eel (freshwater mussel host) stocking, and conducting habitat suitability and survival studies with the goal of eventually re-introducing freshwater mussels into the most suitable areas of the river.

5. The sand and gravel released from upstream of Union and Simkins dams (and from other sources) have continually moved downstream, making their way into the tidal portion of the river, potentially degrading habitat for resident and migratory species. The rate and pattern of movement of this material over time could be an important controlling factor for restoring abundant anadromous fish runs in the non-tidal portion of the Patapsco River. Measuring and tracking this movement of sand and sediment in the tidal Patapsco will provide useful information for determining ecological limitations it may impose on ecological restoration efforts.

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