Patapsco River Dam Removal Study: Assessing Changes in American Eel Distribution and Aquatic Communities, 2013-2014

Biennial Report

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Chapter 1 : Forward

This report is an addendum to one first published in May 2013 describing 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 dams on the Patapsco River from 2009 though 2012. This latest document presents the results of an additional two years of monitoring, from 2013 though 2014. Data collected continues to assess the impacts of the removal of Simkins Dam on the Patapsco River's biota, as well as provides a baseline for the assessment of biological impacts from Bloede Dam's future removal. There are five aspects of the Patapsco River's ecosystem that were examined: anadromous fish, American eels, American eel passage, resident fish, and benthic macroinvertebrates. 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, as well as changes to the monitoring design since the publication of the last report. 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 Simkins Dam are intended to guide continued Patapsco River restoration efforts and other attempts to restore river connectivity in Maryland and elsewhere.

Chapter 2 : 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, began performing biological monitoring in the Lower North Branch Patapsco River in 2009. Results of this monitoring through 2012, intended to gauge the impacts of the removal of Simkins Dam in Winter 2010 as well as provide baseline data for the potential removal of Bloede Dam in the future, were first published in May 2013. Monitoring has continued since that time, and results of the subsequent two years' effort, from 2013 through 2014, are presented here. The goals of this project have changed slightly, but are largely the same as those presented in the previous report. Primarily, they seek to determine the continued impacts of the removal of Simkins Dam on American eel (*Anguilla rostrata*) and anadromous fish distributions as well as on resident fish and benthic macroinvertebrate 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 resident fish and benthic macroinvertebrate communities 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 and benthic macroinvertebrates for two years beyond the original reporting period (2013-2014), extending the data set to four years beyond the removal of Simkins Dam (Simkins Dam was removed in December 2010). 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.

Despite the fact that Union Dam was also removed during the same time period (February 2010) this report, like the one before it, focuses on the removal of Simkins Dam, as it was found to have the most substantial influence on the ecology of the Patapsco.

This report focuses on six key topics: anadromous fishes, American eel distribution/abundance, American eel passage, resident fish, benthic macroinvertebrates, and general conclusions/recommendations. Three topics addressed in the previous report- physical changes, water quality, and freshwater mussels- have not been addressed in the current document.

Anadromous Fishes

Monitoring for anadromous fish in the Patapsco River in 2013 and 2014 was focused primarily on determining the status of river herring (including both alewife and blueback herring) and hickory shad in relation to presence of a migration barrier- Bloede Dam. These species were sampled using electrofishing equipment and a hoop net at three sites downstream from Bloede Dam and one site upstream during spring (March-May) 2013-2014, just as was done during spring 2011-2012. All three species were found throughout the river downstream of the dam during spring spawning migrations. Despite this, no individuals of any of the three species were observed upstream of the dam, even when present in relatively high abundance immediately downstream. These results suggest that Bloede Dam functions as a nearly complete migration barrier for these species. Bloede Dam is the first major obstacle encountered by these fish moving upstream from the Chesapeake Bay and likely blocks access to the entire river upstream, despite the presence of a fish ladder designed specifically for their passage. Certain elements of the ladder's design, including sharp turns and competing, turbulent flows, may be limiting is effectiveness. While redesigning the ladder to address these issues may improve passage, the simplest and most efficient method of improving fish passage is to remove the dam entirely.

American Eel Distribution and Abundance

Sampling for American eels (*Anguilla rostrata*) was conducted at 24 sites in the Patapsco River during summer (June-September) 2013-2014 as a continuation of the monitoring conducted during 2009-2012. Beginning in 2013, all eels collected were individually measured for total length in order to better observe changes in size distribution throughout the river over time. Overall, eel abundance decreased with increasing distance upstream, a trend that has held since monitoring began in 2009. Examination of length distributions collected during 2013-2014 revealed that there are larger proportions of smaller eels in populations downstream of Bloede Dam than in populations upstream, perhaps due to small upstream-migrating eels concentrating below the barrier. Eel density, used as a measure of abundance, decreased at two sites downstream of Simkins Dam following its removal in 2010, and has remained lower than pre-removal level during 2013 and 2014. These decreases in density are likely the result of habitat disturbance following dam removal as sediment formerly impounded by Simkins Dam dispersed downstream. Recovery of both habitat and eel density is potentially hampered by the continued presence of Bloede Dam immediately downstream of the impacted area. Removal of this barrier would likely speed recoveries and easier migration of smaller eels into available habitats upstream.

American Eel Passage

Efforts to improve the passage of American eels around Daniels Dam were begun in 2013. With the removal of Bloede Dam looming on the horizon, Daniels Dam will eventually be the first major barrier encountered by young eels as they migrate upstream from the Chesapeake Bay. In preparation for this, steps were taken to alleviate the dam's status as a migration barrier and improve both the health of the Patapsco River's eel population and the health of the river ecosystem as a whole. Movement of eels occurred in two phases- active transport in 2013 and passive transport in 2014. In 2013 85 eels were collected downstream of Daniels Dam and released upstream. In 2014 an eel ladder was constructed on the dam, and allowed the passage of 14 eels between 24 July and 3 September. Sites upstream and downstream of Daniels Dam which are already monitored to assess impacts of the Simkins Dam removal are also used to determine any changes in eel abundance or size structure following these passage improvements. No changes have been observed to date, likely resulting from a relatively small number of eels being able to disperse into a comparatively large amount of upstream habitat.

Resident Fish

The removal of Simkins Dam was both a restoration of connectivity within portions of the Patapsco River and a major disruption to habitat and stream flows. Resident fish communities at five sites on the main stem river have now been monitored for six years- two pre-removal (2009- 2010) and four post-removal (2011-2014). These sites include three in the "impact area" adjacent to Simkins Dam and two controls further upstream. Following removal, fish assemblages at sites adjacent to and separated by Simkins Dam became more similar in species composition, but there were also presumably short-term, less positive consequences stemming from habitat alteration. Sediment stored behind the dam has moved downstream following dam removal, disrupting habitat used by stream fishes, especially those that utilize clean, coarse substrate as refuge and for feeding. As a result, declines in certain groups- benthic riffle fishes, intolerant fishes, lithophilic spawners, and smallmouth bass- were observed in the two years immediately following dam removal. These impacts are apparently abating, as improvements have now been noted during the two additional years since the publication of the last report (2013-2014). Abundance of fishes in these groups, as well as overall fish density, has increased at the site immediately downstream of Simkins Dam during 2013-2014 to levels equal to those observed pre-removal (2009-2010). Despite this, recoveries are not universal. Though they are improving, metrics at the furthest downstream site, just downstream of Bloede Dam, have not fully recovered to pre-removal levels like those observed upstream. We suspect these delays are a result of confounding influence of Bloede Dam, and predict that fish assemblages will continue to recover in time as conditions stabilize and habitat quality continues to improve.

Benthic Macroinvertebrates

Changes in the macroinvertebrate community in the Patapsco River appear to continue to be associated with shifts in dominant habitat following dam removal. The most dramatic changes occurred at sites in the immediate vicinity (both upstream and downstream) of Simkins Dam, where there have been the most dramatic shifts in habitat type in the four years following the dam's removal. Changes upstream of the dam were driven primarily by shifts from lentic to lotic habitats following removal, while those downstream were driven by movement of released sediment over top of more preferable benthic habitats. An influx of EPT taxa upstream of the dam (especially stoneflies in 2013) coupled with a decrease in burrowing taxa were the primary metrics that illustrated the response to habitat changes. While the habitat shifts and associated changes in the benthic community upstream of Simkins Dam appears more pronounced and permanent (impounded lentic habitat has been replaced by lotic flowing habitat), changes downstream were less pronounced and are mostly temporary. This is due to impacted habitats recovering as sediment continues to move downstream and relatively mobile taxa dispersing to recolonize impacted areas.

Conclusions and Recommendations

The conclusions and recommendations based on our work in the Patapsco River stem from two years of pre- and four years of post-dam removal data and observations, and should be considered preliminary. Although additional post-dam removal monitoring is needed, we've 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 recommendations:

1. Ecological monitoring should continue for as long as possible to document the long-term ecological response to dam removals, both past and future. The removal of Bloede Dam is likely within the next one to two years, and impacts from this process will begin to occur before ecological changes in the Patapsco River attributable to the removal of Simkins Dam have had a chance to stabilize. It will likely not be possible to tease impacts of one removal from the other, and continuing to document biological changes as they occur will be the best way to demonstrate the benefits of dam removals on the Patapsco River. Lessons learned from monitoring in the future will inform decisions pertaining to future fish passage and prospective dam removal projects. All data collected so far serve as useful indicators of stream condition and will provide the basis for restoration progress. These indicators should continue to be used in future years.

2. Bloede Dam should be removed. 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 six years of baseline data for examining the ecological benefits of Bloede Dam's eventual removal.

3. If Bloede Dam is removed, Daniels Dam should also be removed. When 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 on it for passing migratory fishes should be examined.

Chapter 3 : 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 (Figure 3.1). 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. 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. Biological monitoring of the impacts of these dam removals has been occurring since 2009, covering two years pre-removal (2009-2010) and four years post-removal (2011-2014).

Simkins Dam was removed 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. Assessing the impacts of this sediment was a major focus of the last monitoring report, published in 2013 and covering first four years of monitoring effort (two years pre-removal and two post, 2009-2010 and 2011-2012, respectively). That report addressed several key areas, including anadromous fish, American eels, resident fish, and benthic macroinvertebrates. Responses to improved ecosystem connectivity were minimal (owing largely to continued presence of additional dams in the immediate area), but responses to altered habitat were evident. Given the transient nature of the released sediment, it was assumed that observed impacts would be temporary, and conditions would revert to levels equal to or improved from pre-removal levels as sediment moved out and habitats stabilized.

Monitoring has now continued in the Patapsco River for an additional two years beyond the completion of the last report (2013-2014). The same key areas (anadromous fish, American eels, resident fish, and benthic macroinvertebrates) have been examined, with additional attention to improving American eel passage at another barrier, Daniels Dam, further upstream in the watershed. This latest effort builds from the previous report, anticipating continued improvements, or at the very least returns to pre-removal baselines, as habitat impacts abate. Additionally, these ongoing monitoring efforts can begin to serve as baselines for the next dam removal project on the Patapsco River.

Two additional dams remain on the river– Bloede and Daniels Dams. Bloede Dam is currently the downstream-most dam on the Patapsco. It is approximately 10 meter high and stores approximately 312,000 cubic yards of sediment. Daniels Dam is 14.3km upstream of Bloede Dam and is 8.2 meters high. Both dams include Denil fish ladders, but they are of limited functionality. Plans are in place for the removal of Bloede Dam in 2016.

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 has opened the habitat between them and provided an opportunity to examine the potential ecological advantages and disadvantages of dam removal when the time comes for either structure's demolition. The monitoring conducted thus far, the results of which are presented in both this and the preceding report, serves both to assess the impacts and recoveries following the two previous dam removals and to provide a baseline from which to determine the ecosystem response to any future removals.

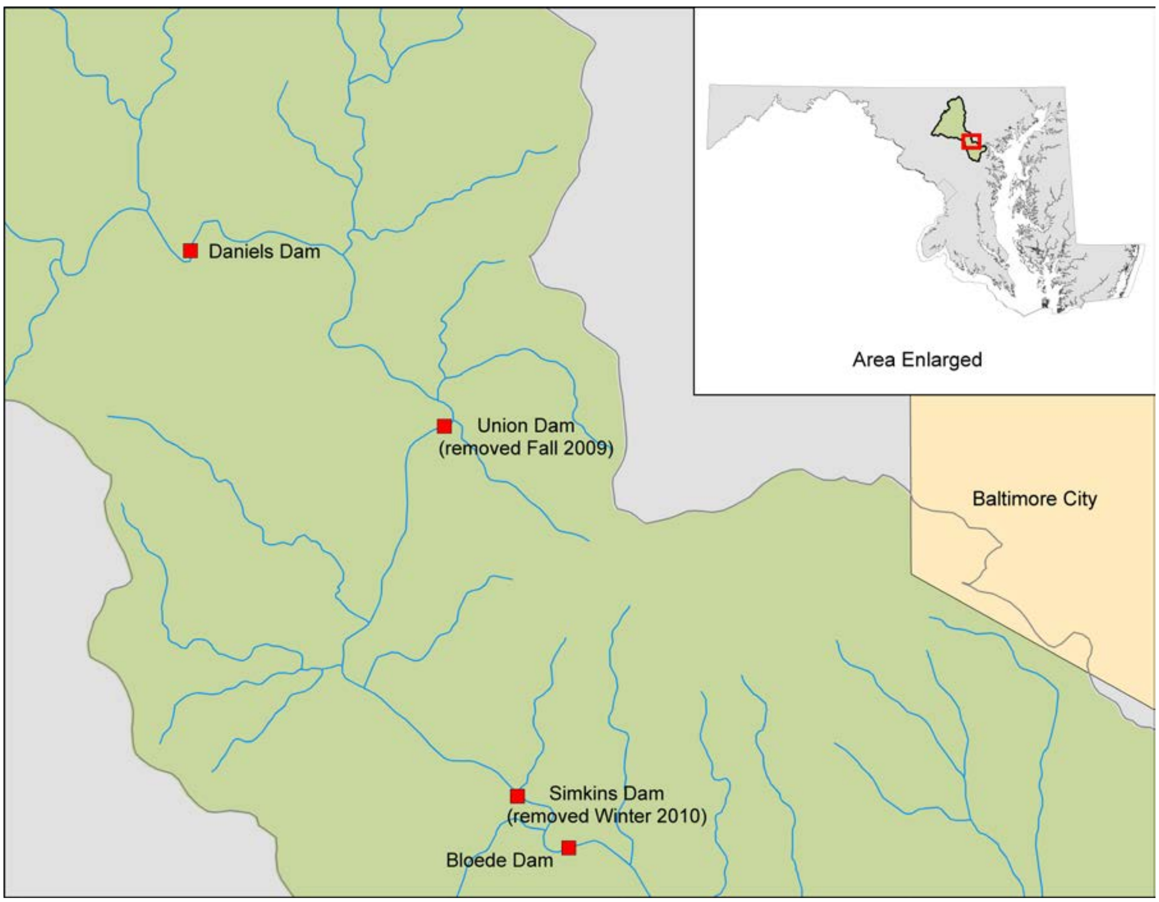

Figure 3.1: Locations of current and former dams on the Patapsco River

Chapter 4 : Anadromous Fish in the Patapsco River

Introduction

Spring spawning migrations of anadromous fish in the Patapsco River have been monitored extensively since 2011. The primary motivation has been to establish a baseline data set prior to the anticipated removal of Bloede Dam. Any subsequent sampling after the removal of this barrier will serve to assess the potential removal's impacts on anadromous species' range and abundance in the river after the opening of access to additional spawning habitat.

Currently, the only habitat available to anadromous fish in the Patapsco River is that which is downstream of Bloede Dam. Data from fish surveys conducted in the mid-1970s through the late- 1990s showed that, despite the presence of a fish ladder (constructed in 1992), Bloede Dam was a significant barrier to anadromous fish. Of the eight anadromous species observed during that time downstream of the dam, only three migrated past it via the fish ladder. Even those that did pass did so in incredibly low numbers- one American shad, 4 blueback herring, and eleven sea lampreys in four years of monitoring (O'Dell et al. 1975; MDNR Fisheries Service, unpublished data).

Monitoring efforts by the Maryland Biological Stream Survey (MBSS) in 2011 and 2012 found much the same results as those seen in previous surveys. Electrofishing at four sites downstream of Bloede Dam detected seven species of anadromous fish, but of those seven none were observed exiting the fish ladder to be collected in a hoop net attached to the ladder exit. The conclusion of the report, submitted in spring 2013, was that Bloede Dam remains a significant, if not complete, barrier to anadromous fish migration (Harbold et al 2013).

Monitoring of anadromous fish migrations has continued during spring 2013 and 2014, sampling the same sites and targeting the same species- hickory shad, blueback herring, and alewife- encountered in the Patapsco River in 2011-2012. These three species, combined with American shad, were once incredibly important commercial species that had been known to spawn in virtually all Chesapeake Bay tributaries. Over the last century, stocks in the Chesapeake Bay (including the Patapsco River) and elsewhere along the East Coast suffered major declines. These have been attributed to overfishing, dams and other migration barriers, degraded water quality, and degraded physical habitat for spawning. Faced with increasingly lower numbers, the fisheries for both American shad and hickory shad were closed by 1981, and by 1991 only a remnant population of these fish was suspected to remain in the Patapsco River (Klauda et al 1991). The fishery for river herring (both blueback herring and alewife) followed a similar course, but was not closed until 2012. There is now an effort to restore their numbers in the Patapsco River.

The Maryland Department of Natural Resources Fisheries Service (MDNR Fisheries) has recently partnered with the United States Fish and Wildlife Service (USFWS) to restock American shad, hickory shad and river herring in the Patapsco River, using mitigation funding from the Maryland Port Administration. This effort includes three years of stocking and 5 years of monitoring. Stocking and monitoring both began in 2013 and have continued through 2014. This monitoring, conducted by the USFWS, uses similar methods to those employed by the MBSS in 2011 and 2012. Sampling locations include the downstream most site sampled by the MBSS in 2011 and 2012, as well as two additional sites further downstream. MBSS monitoring in the Patapsco River during the same time period (2013 and 2014), in addition to continuing to provide baseline data for the impending Bloede Dam removal, served to supplement this effort.

Monitoring of anadromous fish during the last two years at four of the five sites sampled since 2011 (monitoring at one site was taken over by USFWS) builds on the data collected previously, as well as addresses the following goals:

1. Continuing to assess the status of Bloede Dam as a migration barrier to anadromous fish- particularly migratory alosids.
2. Documenting any new records of anadromous species that may have been missed during monitoring in 2011 and 2012.
3. Sharing data with USFWS to supplement monitoring of American shad, hickory shad and river herring restocking efforts.

Monitoring during 2011 and 2012 provided preliminary information on the presence and extent of anadromous species in the Patapsco River. The addition of two more years of data from 2013 and 2014 adds to this, painting an even clearer picture of Bloede Dam's status as a migration barrier. We hope to continue monitoring anadromous fish until the removal of this barrier, and beyond, documenting any expansion of these species into habitat made available once the dam is gone.

Methods

Sampling has occurred at five locations on the Patapsco River– three sites downstream of Bloede Dam in the tidal portion of the river, one site directly downstream from Bloede Dam in the dam's tailrace, and one site directly upstream from the dam (Figure 4.1). The downstream most site in the tidal portion of the river (T90) was only sampled once by MBSS in 2013 before monitoring at this site was taken over by USFWS. The remaining four sites have been sampled throughout the spring (March-May) from 2011-2014.

We sampled sites downstream of Bloede Dam using an electrofishing boat and monitored any passage of migratory fish above the dam with a hoop net fitted to the fish ladder exit. Electrofishing was performed while moving downstream, with total shocking time, fish species present, and abundance of migratory species recorded for each site. The hoop net was set for periods ranging up to 400 hours, and was checked roughly every twenty-four hours during each set, with species, abundance, and time of deployment recorded.

Using the recorded time (either spent electrofishing or with the net deployed) and the numbers of migratory alosids caught we calculated abundance (fish per hour) for each species. We did not record abundance of resident (non-migratory) fish species while electrofishing, keeping only a list of the species encountered. We did record abundance of resident fish collected using the hoop net at the Bloede Dam fish ladder.

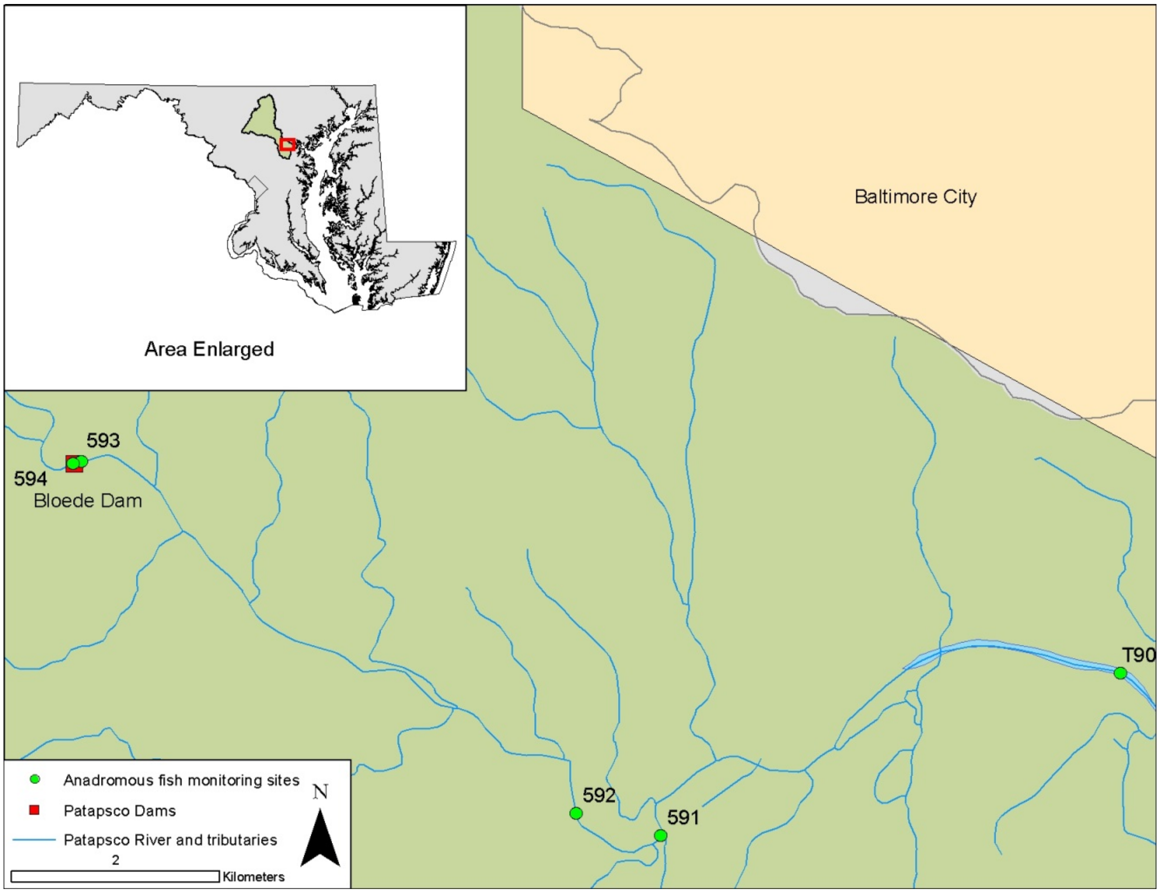

Figure 4.1: Anadromous fish sampling locations on the Patapsco River visited March-May from 2011 to 2014.

There was some uncertainty regarding the identification of blueback herring collected during 2011 and 2012. As a result, records of both species from 2011 and 2012 have been combined as "river herring"- a term often used to refer to both species jointly. In 2013 and 2014 blueback herring and alewife were identified and recorded separately.

In addition to abundance data, we collected additional data in 2013 and 2014 on migratory alosids to assist USFWS with their monitoring efforts. Every alosid collected was measured (total length and fork length), sexed, and had a scale sample removed for aging. Additionally, ten individuals per species per site were retained to have otoliths removed the check for markers that would identify them as hatchery-reared fish. All fish samples (individuals and scales) and measurements were given to USFWS at the end of the sampling season to include in their data set.

Results

We collected migratory alosids downstream of Bloede Dam during each year of monitoring in the Patapsco River. River herring were collected each year from 2011-2014. Hickory shad were collected in 2011, 2013, and 2014, but not in 2012. River herring were more common than hickory shad all four years, but varied in abundance year to year. They were most abundant in 2011 and 2014 and least abundant in 2012 (Figure 4.2). Also in 2013 a single adult American shad was collected by USFWS at one of their electrofishing sites on the Patapsco River downstream from those sampled by the MBSS. There were no American shad observed in either 2013 or 2014 at MBSS monitoring sites.

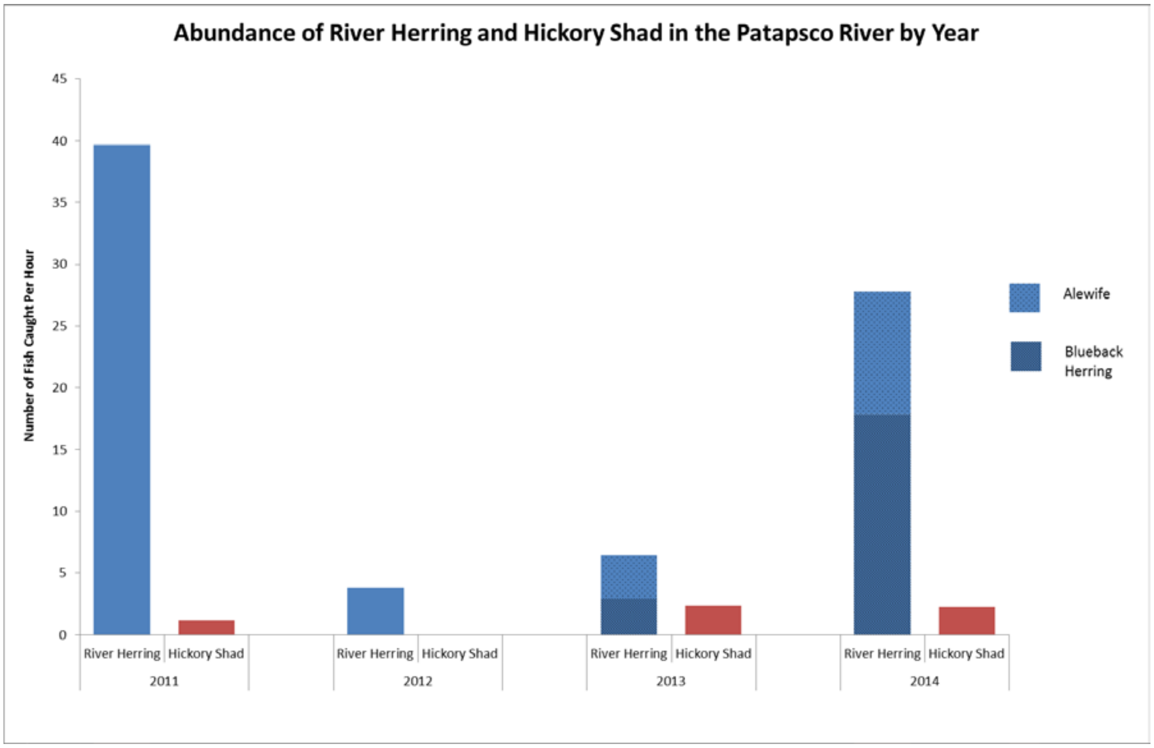

Figure 4.2: Number of migratory alosids caught per hour during four years of spring electrofishing on the Patapsco River, 2012-2014.

Plotting the observed abundance (fish caught per hour) of each species over time showed that, at least in 2011 and 2013-2014, numbers of migratory alosids in the Patapsco River peaked around late April to early May. Abundance of alosids in 2012 was too low to see any discernable pattern over the sampling period. See Figure 4.3 through Figure 4.6 below:

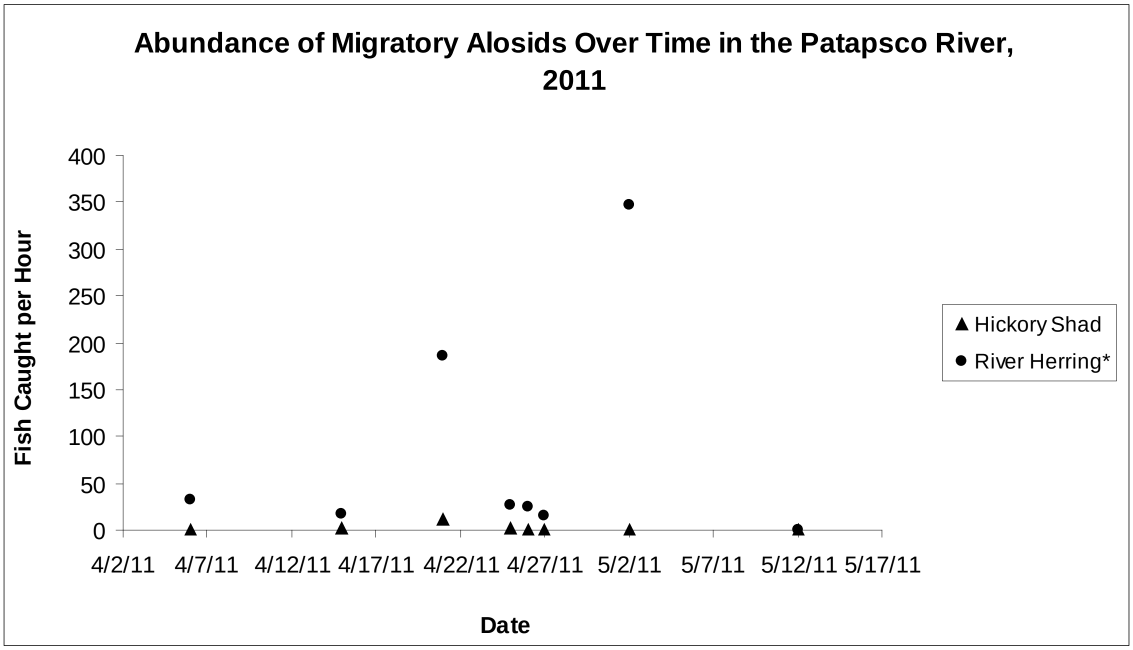

Figure 4.3: Numbers of migratory alosids caught per hour on various days of electrofishing sampling during Spring 2011. *Includes both alewife and blueback herring.

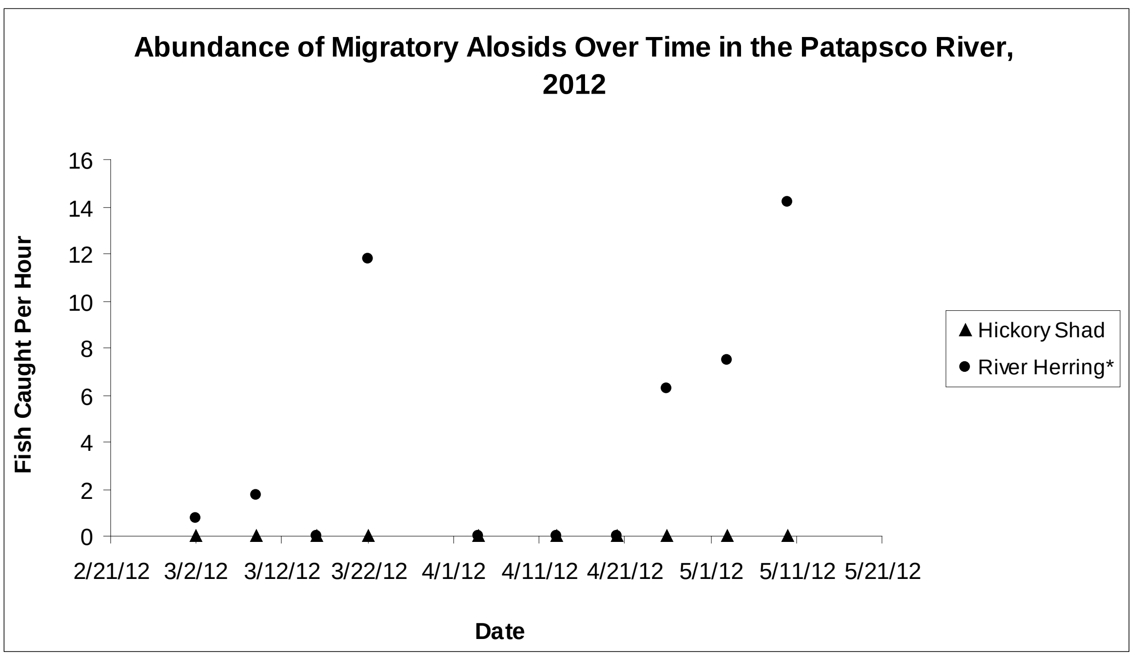

Figure 4.4: Numbers of migratory alosids caught per hour on various days of electrofishing sampling during Spring 2012. *Includes both alewife and blueback herring.

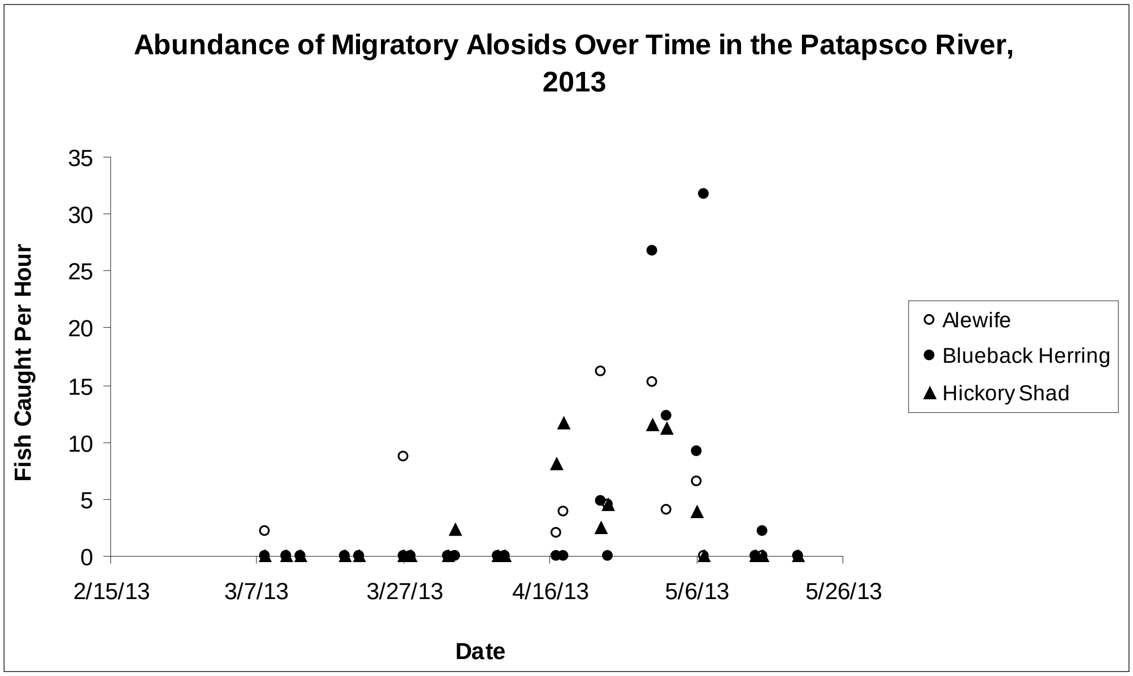

Figure 4.5: Numbers of migratory alosids caught per hour on various days of electrofishing sampling during Spring 2013.

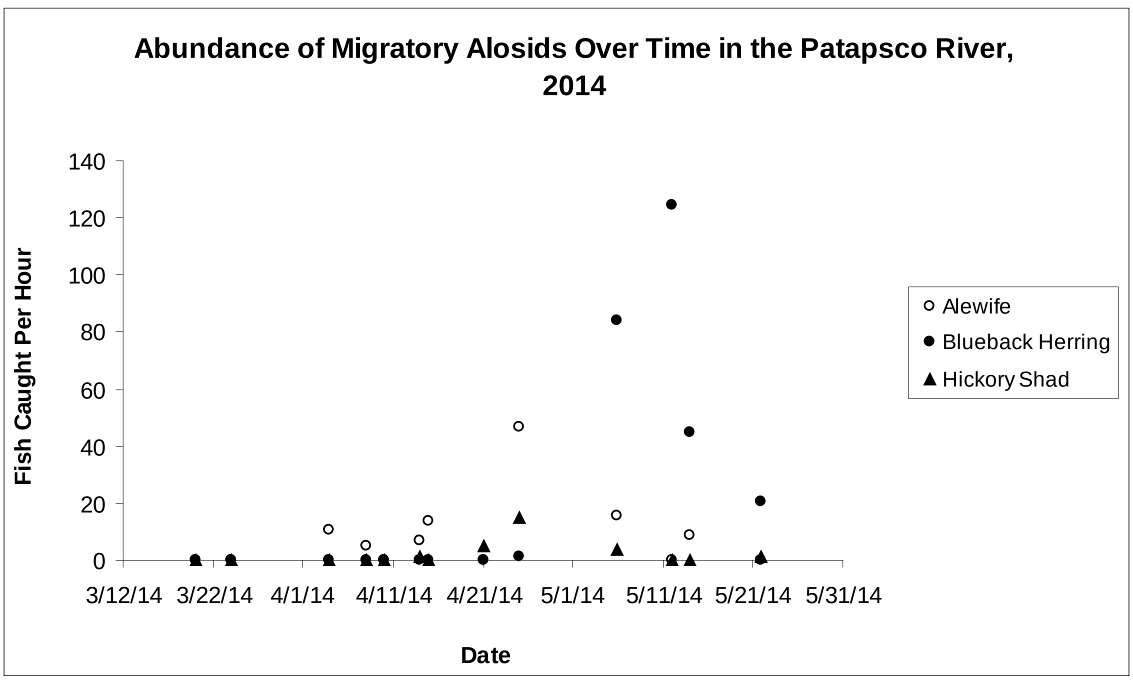

Figure 4.6: Numbers of migratory alosids caught per hour on various days of electrofishing sampling during Spring 2014.

Table 4-1: Non- Migratory Alosid Fish Species collected by boat electrofishing in the tidal Patapsco River from March through May

Species	**2011**	**2012**	**2013**	**2014**
American Eel	X	X	X	X
Banded Killifish	X	X	X	X
Black Crappie	X	X	X	X
Bluegill	X	X	X	X
Bluntnose Minnow		X		
Brown Bullhead		X	X	X
Brown Trout	X	X	X	
Chain Pickerel		X	X	
Channel Catfish	X		X	X
Common Carp	X	X	X	X
Common Shiner	X	X	X	X
Cyprinella sp.			X	
Fallfish	X	X	X	X
Gizzard Shad	X	X	X	X
Golden Shiner			X	
Green Sunfish	X	X	X	
Inland Silverside	X	X	X	
Lamprey sp.	X			
Largemouth Bass	X	X	X	X
Margined Madtom		X	X	
Northern Hogsucker	X	X	X	X
Pumpkinseed	X	X	X	X
Quillback	X	X	X	X
Rainbow Trout	X	X	X	X
Redbreast Sunfish	X	X	X	X
River Chub	X	X	X	X
Rock Bass		X	X	
Satinfin Shiner	X	X	X	X
Sea Lamprey	X	X		
Smallmouth Bass	X	X	X	X
Spotfin Shiner	X	X	X	X
Spottail Shiner	X	X	X	X
Striped Bass	X		X	X
Swallowtail Shiner		X	X	X
Tessellated Darter	X	X	X	X
White Perch	X	X	X	X
White Sucker	X	X	X	X
Yellow Bullhead		X	X	
Yellow Perch	X	X	X	X

In addition to the three species of migratory alosid present in the Patapsco River, we collected 39 other species of fish between 2011 and 2014 (Table 4-1) during electrofishing surveys.

During four years of monitoring we collected 15 species of fish in the hoop net placed over the exit of the Bloede Dam fish ladder (Table 4-2). While we did encounter one adult sea lamprey in the net in 2013, there have been no other anadromous fish seen using the ladder to migrate past Bloede Dam. Despite their presence immediately downstream of the dam, no migratory alosids were observed using the ladder, collected in the hoop net, or observed elsewhere upstream of Bloede Dam during the entire four year monitoring period.

Table 4-2: Fish Species Collected Exiting the Bloede Dam Fish Ladder, 2011-2014

Species	2011	2012	2013	2014
Bluegill		X	X	
Brown Trout		X	X	
Channel Catfish		X		
Common Carp		X	X	X
Common Shiner			X	
Fallfish		X	X	
Gizzard Shad		X	X	X
Northern Hogsucker		X	X	X
Rainbow Trout		X	X	X
Redbreast Sunfish		X	X	
Rock Bass		X	X	
Sea Lamprey			X	
Smallmouth Bass		X	X	
White Catfish		X		
White Sucker		X	X	

Summaries of the migratory alosid length and sex data collected and given to USFWS are available in an appendix at the end of this section (Appendix A).

Discussion

After an additional two years of monitoring in 2013 and 2014 we have reached the same conclusion as our report from 2012- Bloede Dam remains a barrier to anadromous fish, particularly migratory alosids. In the two years since the last report we have continued to document anadromous fish reaching the base of the dam, but still have seen virtually no movement past the dam via the fish ladder. The ladder does, however, continue to pass resident fish species.

While there was a new record of alewife in the Patapsco River in 2013, it is most likely that these fish have been present all along. Photos of fish taken in 2012 were examined after the last report was written and it was discovered that one individual recorded as a blueback herring displayed characteristics more indicative of an alewife. This brought into question the identity of all the fish recorded as "blueback herring" during 2011 and 2012, and the data set was thus amended to refer to all of these individuals as "river herring"- a group that includes both species. More careful field identification during the subsequent 2013 and 2014 sampling seasons showed that there were in fact both blueback herring and alewife present in the Patapsco River. Confirming that there are two species of "river herring" making spawning migrations in the Patapsco River is useful for the purposes of fisheries management, but it does not change the fact that neither species is able to access the river upstream of Bloede Dam.

Research on creating effective fish passage structures at dams has identified several aspects of fish ladder design that might inhibit the attraction and passage of migratory alosids. The "attraction flow" of water coming out of the ladder entrance is a very important cue for fish locating the passage structure. When there is turbulent, competing flow from the dam interfering with this attraction flow, it may be difficult for fish to locate the ladder's entrance, and the efficiency of the structure is diminished (Franklin et al 2012). Once fish have entered the ladder, their progress can be impeded by obstacles like sharp bends, and shad in particular may become trapped in corners (Larinier and Travade 2002).

Both of these detrimental features- competing flows at the ladder entrance and sharp corners - are present in the Bloede Dam fish ladder. The ladder entrance is situated very close to the face of the dam. Especially during high flow events, this arrangement may cause the attraction flow from the ladder to be overpowered by the larger flow from the main river washing over the dam. Also, there is also a sharp corner in the fish ladder- a switchback roughly half way along the ladder's length. While this turn allows to the ladder to maintain a lower gradient and more manageable velocity for migrating fish, it may appear as a dead end and cause migratory alosids that reach it to turn back.

These aspects of the Bloede Dam fish ladder may never allow it to pass large numbers of migratory alosids, even with regular maintenance and repairs. The repairs made to the ladder during winter 2011 did improve passage of resident species (Harbold et al 2013), and fifteen have continued to use it through 2014. Regardless, there still have been no migratory alosids observed in the three years since. Redesigning the ladder would surely be a large and costly undertaking, so we hope that progress toward the removal of Bloede Dam continues, eliminating the barrier and any need for a ladder entirely.

Literature Cited

Franklin, A., A. Haro, T. Castro-Santos, J. Noreika. 2012. Evaluation of nature-like and technical fishways for the passage of alewives at two coastal streams in New England, Transactions of the American Fisheries Society.141:624-637.

Harbold, William. 2012. Second Quarterly Status Report: Assessing the Effects of Dam Removal on the Ecology of the Patapsco River. Quarterly report submitted to American Rivers by the Maryland Department of Natural Resources.

Harbold, W., S. Stranko, J. Kilian, M. Ashton, and P. Graves. 2013. Patapsco River Dam Removal Study: Assessing changes in American eel distribution and aquatic communities. Maryland Department of Natural Resources, Monitoring and Non-Tidal Assessment Division, Annapolis, Maryland. 115 pp.

Klauda, R. J., S. A. Fischer, L. W. Hall, Jr., and J. A. Sullivan. 1991. American Shad and Hickory Shad. Pages 9-1 to 9-27 in S. L. Funderburk, J. A. Mihursky, S. J. Jordan, and D. Riley, editors. Habitat requirements for Chesapeake Bay Living Resources. Chesapeake Bay Program, Annapolis, Maryland.

Klauda, R.J., S.A. Fischer, L.W. Hall, Jr., and J.A. Sullivan, 1991. Alewife and Blueback Herring. In: Habitat requirements for Chesapeake Bay living resources (S.L. Funderburk, S.J. Jordan, J.A. Mihursky, and D. Riley, eds.), Chesapeake Research Consortium, Inc. Annapolis, Md., pp. 10-29

Larinier, M. and F. Travade. 2002. The Design of Fishways for Shad. Bulletin Francais De La Peche Et De La Pisciculture. 364:135-146.

O'Dell, J., J. Gabor, R. Dintaman. 1975. Survey of anadromous fish spawning areas. Completion Report, Project AFC-8 July 1970 – January 1975 for Potomac River Drainage and Upper Chesapeake Bay Drainage. Maryland Department of Natural Resources, Fisheries Administration, Anadromous Fish Survey Program. Annapolis, Maryland

Appendix A : Summary Figures of Migratory Alosid Length and Sex Distributions in the Patapsco river.

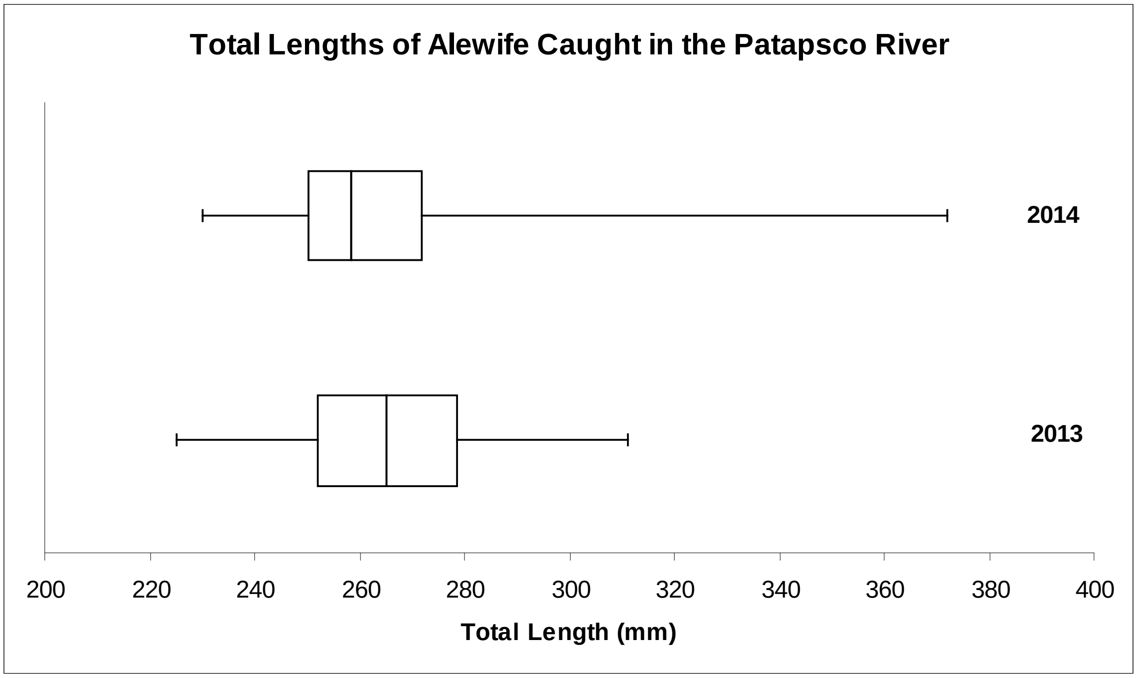

Figure 4.7: Distribution of alewife total lengths from Patapsco River electrofishing surveys during Spring 2013 and 2014.

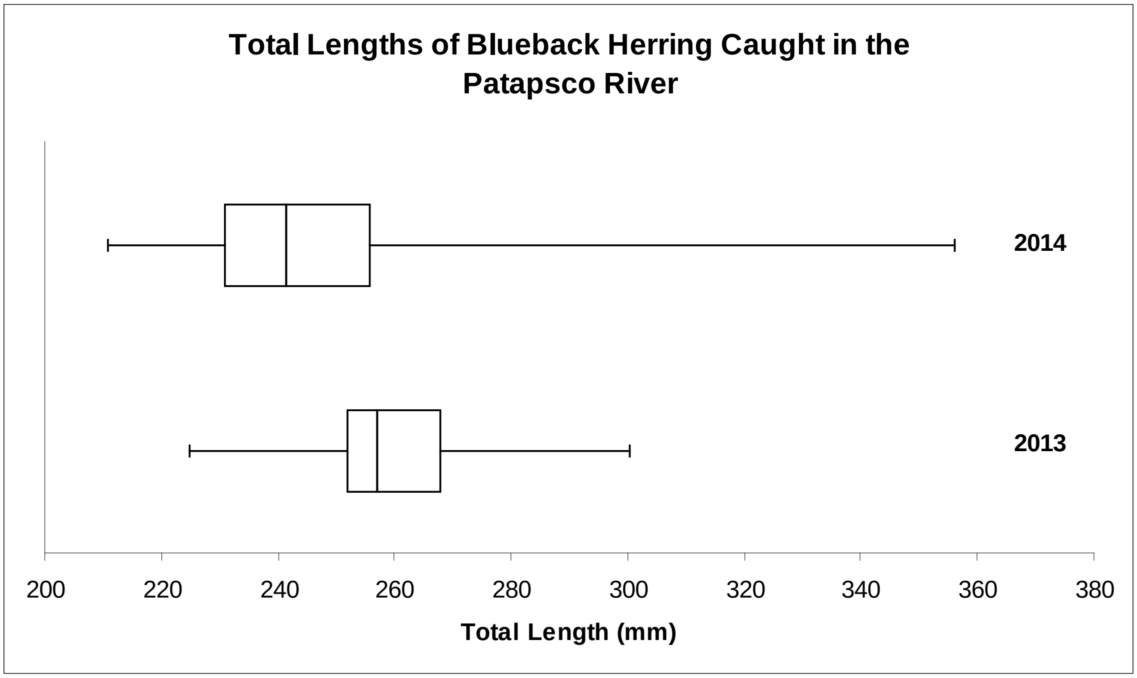

Figure 4.8: Distribution of blueback herring total lengths from Patapsco River electrofishing surveys during Spring 2013 and 2014.

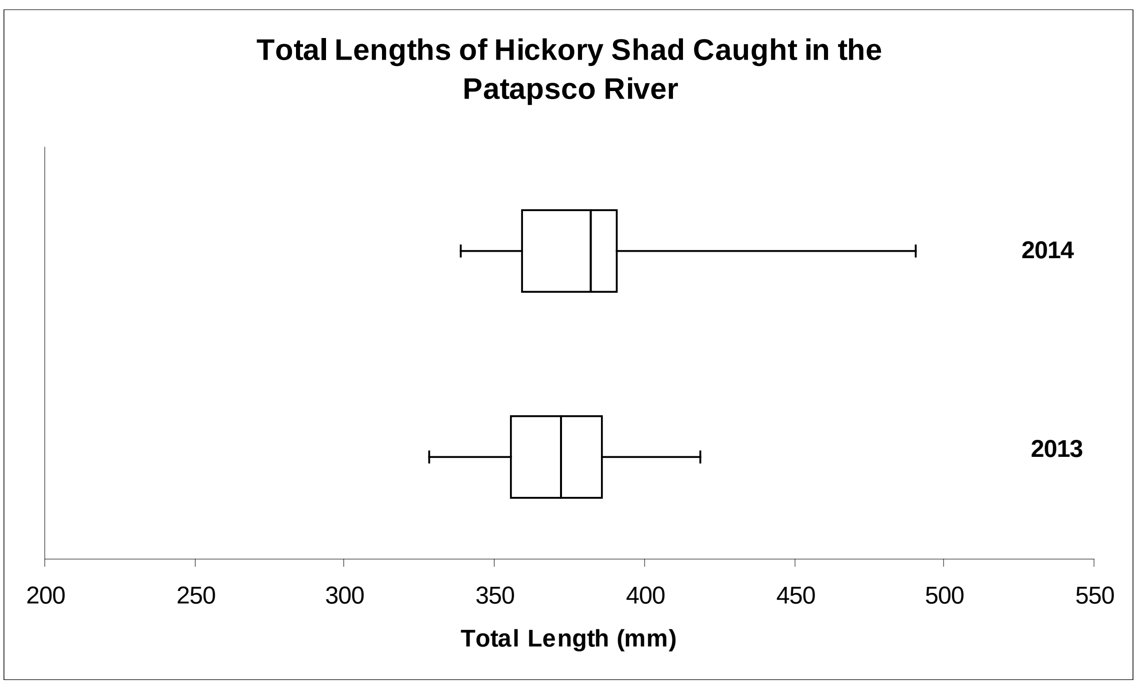

Figure 4.9: Distribution of hickory shad total lengths from Patapsco River electrofishing surveys during Spring 2013 and 2014.

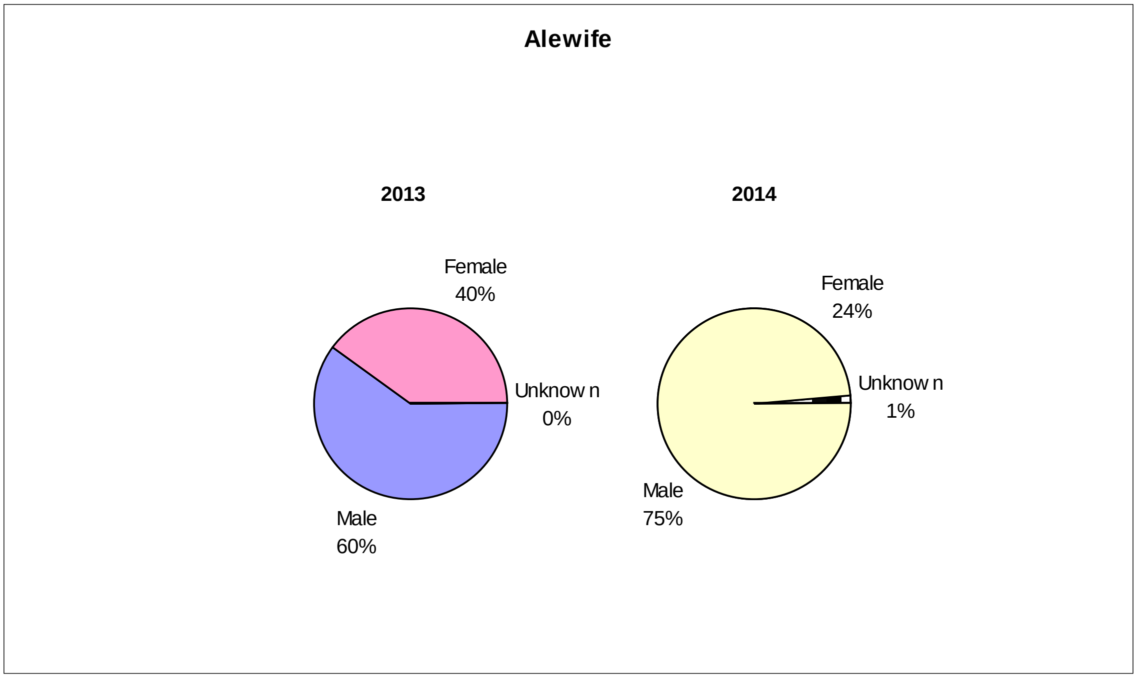

Figure 4.10: Sex distribution among alewife from Patapsco River electrofishing surveys during Spring 2013 and 2014.

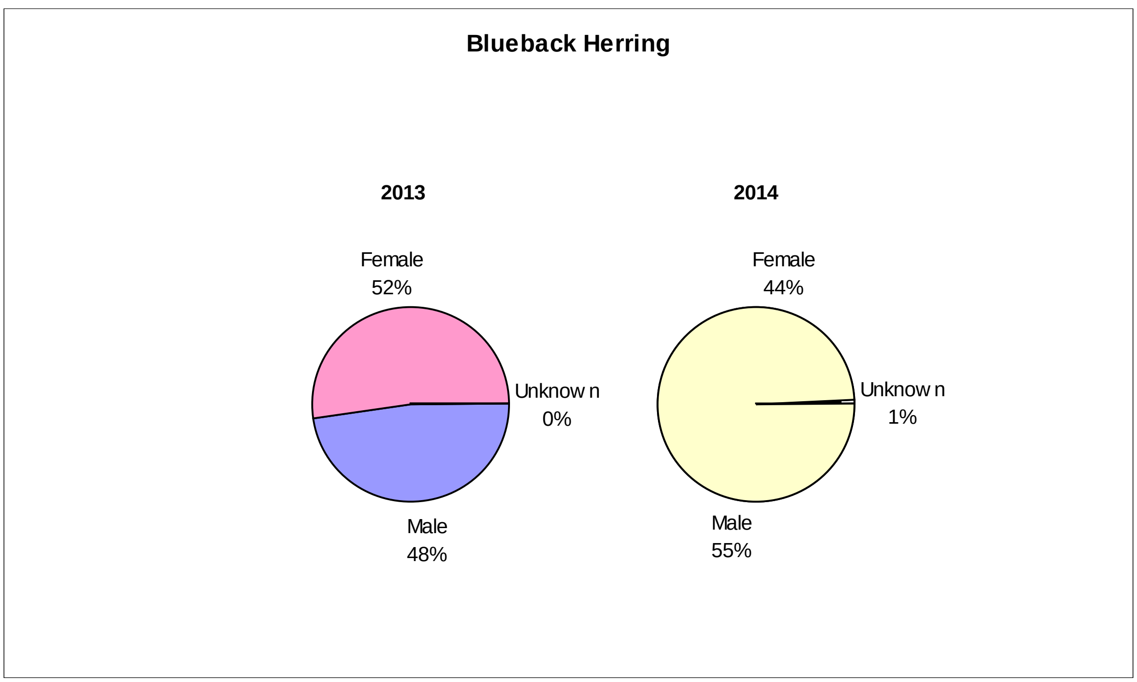

Figure 4.11: Sex distribution among blueback herring from Patapsco River electrofishing surveys during Spring 2013 and 2014.

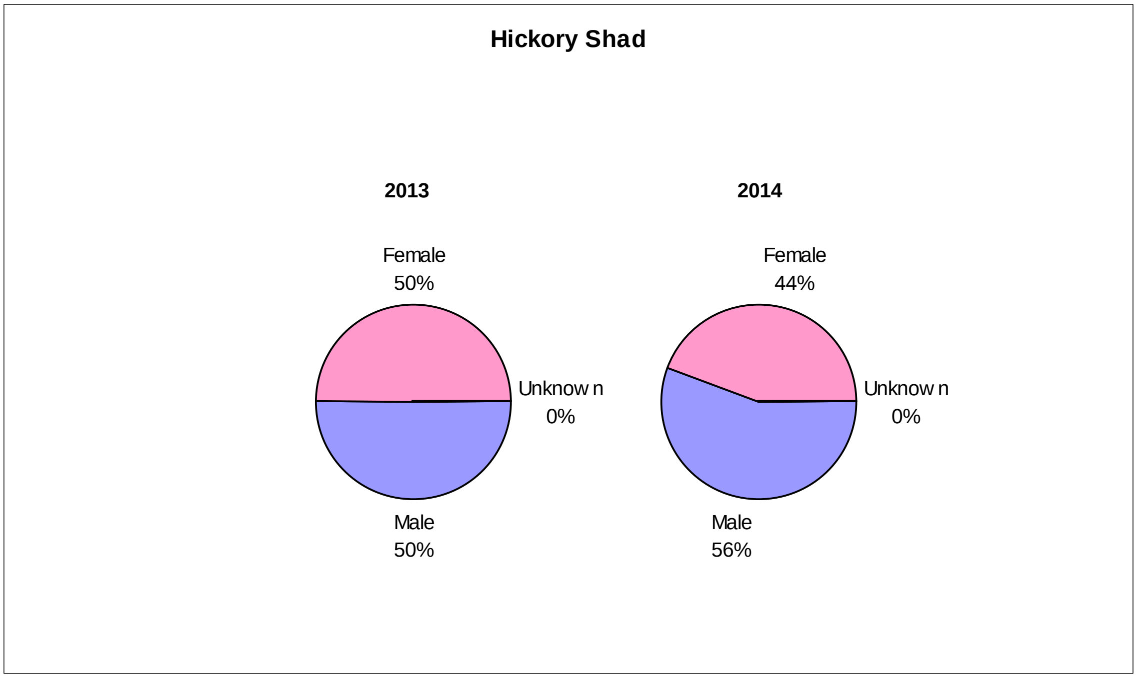

Figure 4.12: Sex distribution among hickory shad from Patapsco River electrofishing surveys during Spring 2013 and 2014.

Chapter 5 : American eel distribution, abundance, and total length in the Patapsco River four years after the removal of Simkins Dam

Introduction

American eels (hereafter eels) were a major focus of research during the first phase of monitoring for potential biological impacts of dam removal in the Patapsco River during 2009-2012. They have remained so during 2013-2014. Eels are both an ecologically and commercially important species that are in decline throughout their range in the face of a wide array of anthropogenic impacts (Haro et. al 2000). Of the many stressors affecting the eel population in the Patapsco River, we presume that migration barriers are the most important, and that the removal of those barriers would have a large influence on eel distribution and abundance.

Eel abundance typically decreases with increasing distance upstream from tidal waters (Smogor et al. 1995, Oliveira 1997, Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007) while size increases (Smogor et al. 1995, Krueger and Oliviera 1999, Goodwin and Angermeier 2003, Cairns et al. 2004, Machut et al. 2007). Barriers, such as dams, may exacerbate this pattern. Eel abundance is usually lower upstream of barriers, and may be higher immediately downstream as eels concentrate below the barrier (Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007). When dams on other rivers have been removed, eels tended to respond by increasing in abundance and decreasing in size above the former barrier as more, smaller eels gained unrestricted access to upstream habitats (Hitt et al. 2012). We anticipated a similar result in the Patapsco River.

The demolition of Simkins Dam in the winter of 2010 afforded us an opportunity to examine whether eels would respond to the removal of a migration barrier similarly to examples in the literature. We recorded abundance and estimated size of eels in the Patapsco River from 2009 to 2012 (two years pre-removal and two post-removal) and examined the data to determine trends over time. As expected, abundance decreased with increasing distance upstream while size increased. Following the removal of Simkins Dam, eel abundance in the Patapsco River decreased at sites immediately downstream of the barrier, but did not increase upstream. Any changes in size were similarly inconclusive. We speculate that this was due more to dam-related habitat change than to the removal of a barrier, as well as the persistence of Bloede Dam immediately downstream (Harbold et al 2013). More than anything else, we concluded that additional monitoring would help produce a clearer picture of the long term impacts of dam removal on the Patapsco's eel population.

We now have spent an additional two years monitoring eels in the Patapsco River since making these conclusions regarding the impact of the Simkins Dam removal. During this period we have added extensively to our sampling design in an effort to improve our ability to generate conclusive results regarding changes in eel abundance and size in the Patapsco River. Considering both the additions to our sampling design and building from our previous conclusions we expect that:

- Eel numbers will continue to show a decreasing trend with increasing distance upstream
- Improved methods of measurement will confirm that eel size increases with increasing distance upstream
- Impacts to eel abundance attributed to habitat degradation following dam removal will abate as affected areas recover from the dam removal disturbance

Our monitoring now serves two purposes- continuing to evaluate the impacts of the removal of Simkins Dam, as well as gathering baseline data to better determine the impact of the anticipated removal of Bloede Dam.

Methods

In 2013 and 2014 we collected eels in the Patapsco River during the summer (June-September) following the same electrofishing protocols (Ciccotto et al. 2009) used in 2009-2012. We sampled 24 sites for eels during this time period, including 19 sites that have been sampled since 2009 and five new sites added in 2013 (Figure 5.1). We recorded eel abundance and time spent electrofishing at all sites and, beginning in 2013, measured each eel collected (total length, millimeters). Additionally, we recorded select physical habitat parameters (stream length, transect widths, and instream habitat score) at twelve sites.

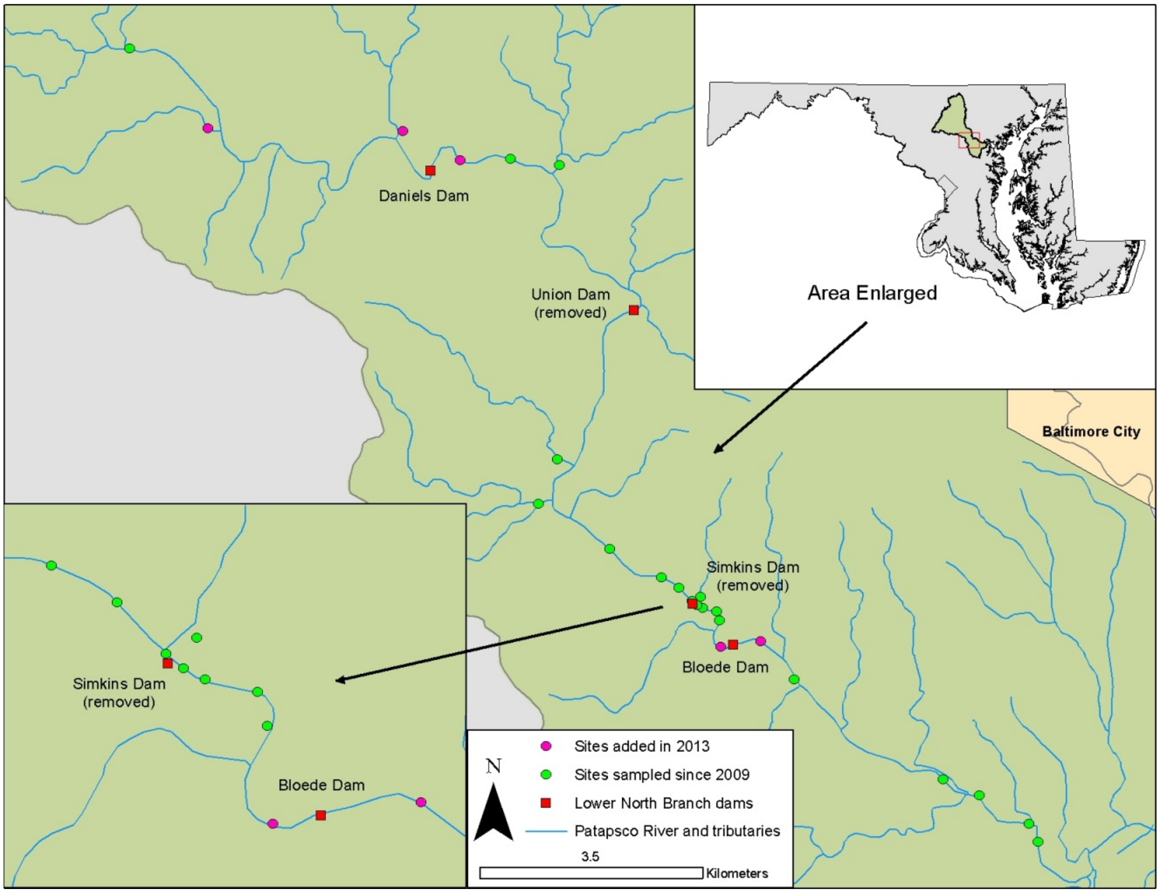

Figure 5.1: Locations of American eel monitoring sites in the Patapsco River.

We recorded the total number of eels collected at all 24 sites and calculated the number of eels collected per hour (hereafter abundance) using the time spent electrofishing. At sites where both eel abundance and physical habitat parameters were recorded from 2009-2014 we estimated eel density (eels per square meter, hereafter density) using the total eel abundance, total length of the site and the measured width at four transects.

Additionally, we scored instream habitat, a qualitative measure used to rate the quality of available fish habitat relative to other streams in the State of Maryland (Stranko et al 2015), concurrent with eel collection at the same sites since 2009.

We appended abundance data collected in 2013-2014 to the 2009-2012 data set and plotted the mean eel abundance for sites sampled throughout the period with respect to the site's distance in river kilometers (Rkm) from the Patapsco River mouth. We used the resulting graph to determine any trends in eel abundance with increasing distance upstream. We repeated this process with 2009-2014 density estimates, again looking for trends moving upstream from the river mouth.

To investigate possible changes in the Patapsco River eel population following the Simkins Dam removal, we examined density values from five sites situated above and below dams over time, from 2009 to 2014, looking for correlations with distance, time, and physical habitat parameters (Figure 5.2). We selected these five sites to represent the conditions upstream and downstream of three dams on the Patapasco River. Sites 501 and 502 capture the downstream and upstream (respectively) conditions around Bloede Dam.

Site 502 also reflects the conditions downstream of Simkins Dam, and is complimented by site 504 representing the conditions upstream of that blockage. Sites 510 and 511, acted as controls, far from both Bloede Dam and Simkins Dam.

We used density rather than abundance to look at changes over time to avoid errors resulting from possible changes in the efficiency of eel capture. Density estimates are available only for the subset of sites where physical habitat parameters are measured. Abundance estimates were still used for monitoring the eel population at sites where physical habitat data were not collected.

By plotting the density of eels over time we were able to observe changes in density that may have occurred following dam removal. We performed a repeated measures two factor analysis of variance (ANOVA) using post hoc pair-wise tests with a Bonferonni correction ($\alpha = 0.006$) to determine the significance of changes in eel density over space and time.

We grouped individually measured eels collected in 2013 and 2014 by location and examined the distribution of total lengths in each group to determine trends in size with respect to the locations of dams still present at the time of sampling. The three groups were:

- Downstream of Bloede Dam (A)
- Between Bloede Dam and Daniels Dam (B)
- Upstream of Daniels Dam (C)

Total lengths from 2013 and 2014 were pooled within in each group. We then compared the means of each group to one another (A to B, A to C, B to C) using an Analysis of Variance (ANOVA) and post hoc Tukey's test.

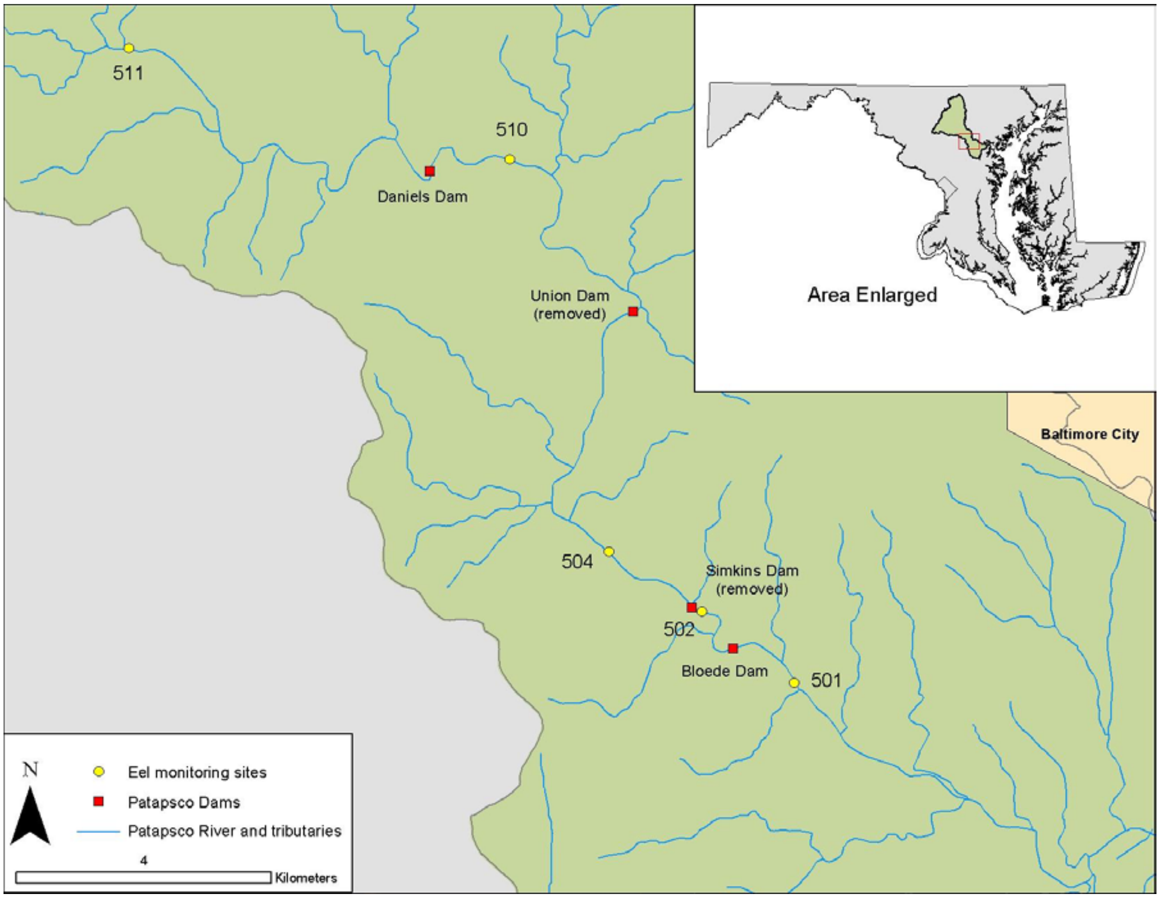

Figure 5.2: Locations of sites used to analyze changes in American eel density following the removal of Simkins Dam.

Results

We collected eels at all 24 sites sampled during 2013 and 2014. This included all the sites where monitoring has continued since 2009-2012 as well as the five new sites added in 2013.

Just as we observed in 2009-2012 (Harbold et al 2013), average abundance of eels over the entire monitoring period (2009-2014) declined with increasing distance upstream (Figure 5.3). Density displayed a similar trend, showing a gradual decrease in the number of eels per square meter of stream bottom with increasing distance from Baltimore Harbor (Figure 5.4).

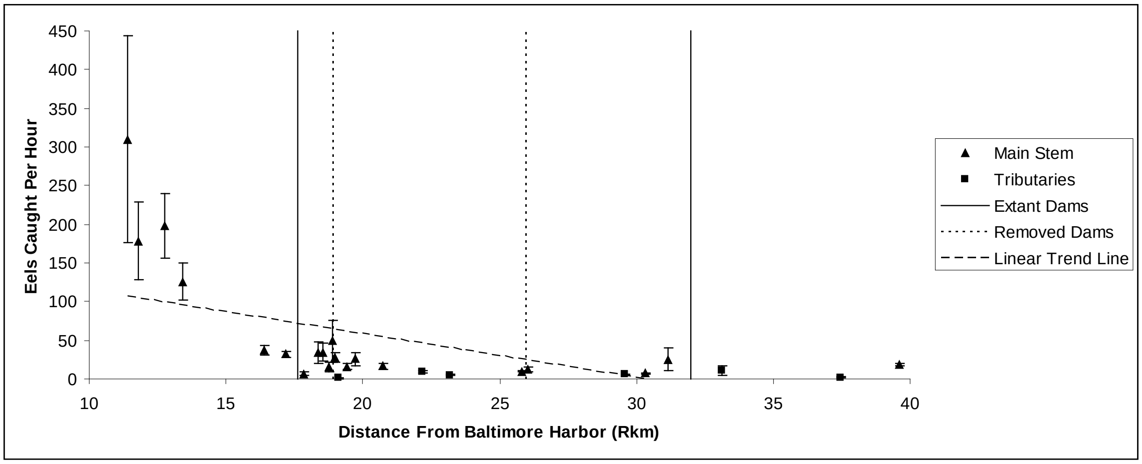

Figure 5.3: Average abundance (+/- 1 SE) of American eels at Patapsco River monitoring sites sampled between 2009 and 2014.

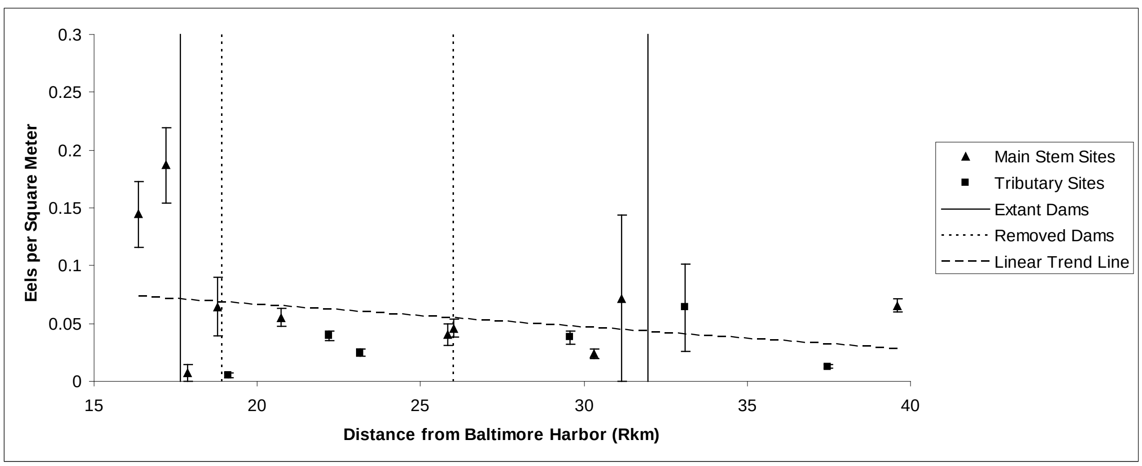

Figure 5.4: Average density (+/- 1 SE) of American eels at Patapsco River monitoring sites sampled between 2009 and 2014.

We observed changes in eel density at 502, the site just downstream of Simkins Dam, almost immediately following removal. In the two years preceding dam removal (2009 and 2010), eel densities at 502 were the second highest of the five sites used in this analysis. Density at 502 declined dramatically in 2011 after Simkins Dam was removed, and has remained lower than pre-removal levels ever since (Figure 5.5). A repeated measures two factor analysis of variance (ANOVA) using post hoc pair-wise tests a Bonferonni correction ($\alpha = 0.006$) verified the significance of the decrease in density between pre- and post-removal time periods ($p=0.0160$) at this site, while indicating that eel density just upstream of Simkins Dam at site 504, as well as those at the control site, 510, did not change.

Eel density downstream of Bloede Dam, at site 501, was the highest of all those used in this comparison from 2009 to 2011 ($\mu = 0.206$ eels/m^2). In 2012, two years after Simkins Dam was removed, density at this site dropped by over half, and has remained low through 2014 ($\mu = 0.082$ eels/m^2). From 2012-2014, mean density at 501 is actually closer to mean density upstream of Daniels Dam, at site 511 ($\mu = 0.069$ eels/m^2) than to mean density at 501 from 2009-2011 (Figure 5.5).

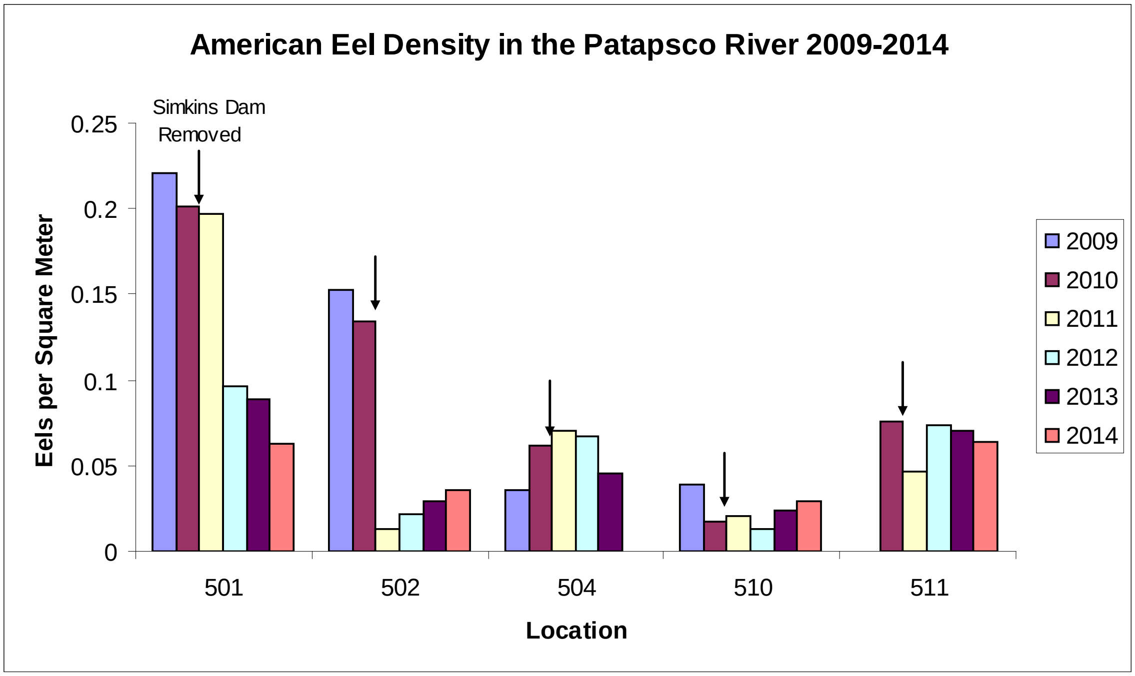

Figure 5.5: American eel density at Patapsco River monitoring locations 2009-2014.

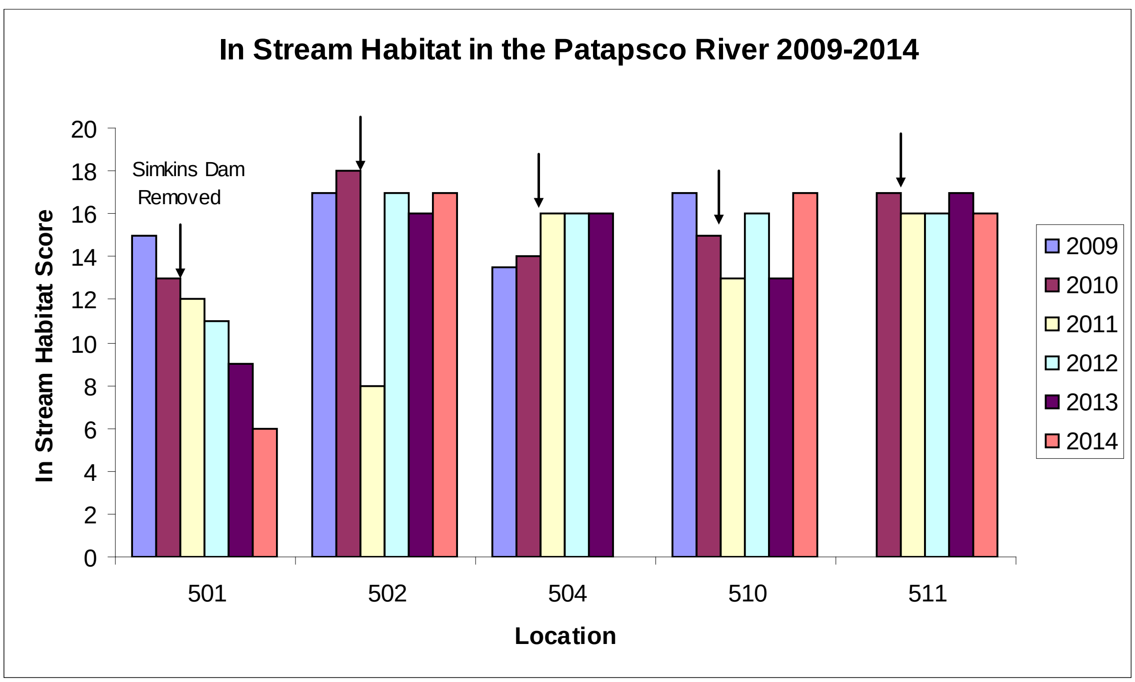

Figure 5.6: In stream habitat scores at Patapsco River monitoring locations 2009-2014.

Instream habitat was scored at the same five sites where eel density was compared from 2009-2014. During this time period, instream habitat declined steadily at site 501. It was relatively stable elsewhere, with the exception of a dramatic drop at site 502 in 2011, immediately following dam removal. In stream habitat scores for this site returned to and remained at pre-removal levels from 2012-2014 (Figure 5.6).

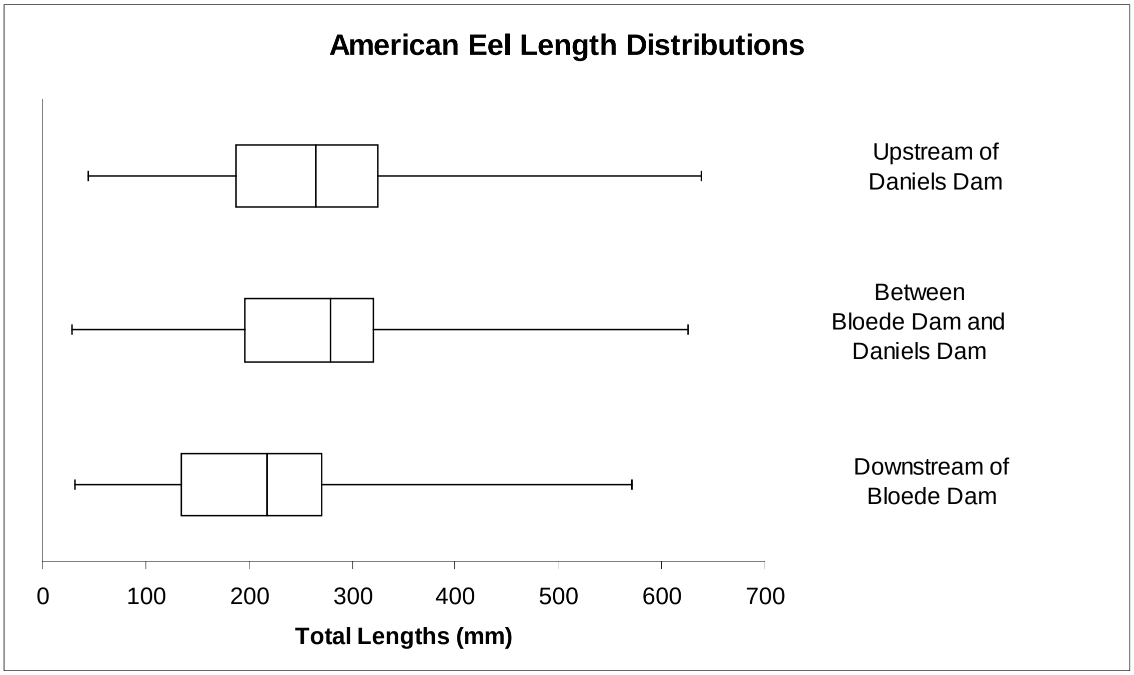

Figure 5.7: Distribution of American eel total lengths in three segments of the Patapsco River 2013-2014.

Eel size, represented by the distribution of individual total lengths in each of three river segments, tended to be smaller downstream of Bloede Dam ($\mu = 207.35$mm) than either between Bloede and Daniels Dams ($\mu = 271.60$mm) or upstream of Daniels Dam ($\mu = 270.51$mm)(Figure 5.7). An ANOVA with a post hoc Tukey's test showed that proportion of smaller (total length) eels at sites downstream of Bloede Dam was significantly higher ($p<0.0001$) than that at sites in either group upstream.

Conclusions

The results of our monitoring efforts since 2012 have shown that eel abundance continues to decrease with increasing distance upstream of the river mouth, while individually measuring eels has confirmed that smaller eels tend to be most abundant in downstream reaches. Eel numbers did not increase in areas impacted by post-dam removal habitat change, possibly due to the confounding influence of Bloede Dam.

We have been tracking eel abundance (number of eels collected per hour) in the Patapsco River since 2009. We have recorded abundance at twenty-four sites during this time period, spaced out over a span of roughly thirty kilometers. Throughout the entire period, from 2009-2014, abundance decreased with increasing distance upstream. These findings are consistent with those from other studies, which found eels to be more numerous downstream, especially downstream of migration barriers (Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007). Indeed, the highest abundances of eels during the entire monitoring period have been at the downstream-most sites, below Bloede and Simkins Dams (Figure 5.3 and Figure 5.4).

Previous research has shown that eel size increases with increasing distance upstream. These studies measured eels individually, finding the largest individuals to be most common further upstream (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 in 2009-2012, and instead estimated eel size by averaging individual biomass from the aggregate eel mass collected at each site (Harbold et al 2013). While this technique did allow us to conclude that average size of eels in the Patapsco River increased with increasing distance upstream, it complicated our ability to observe potential changes resulting from dam removal. The average size of eels may change, but it would be unknown whether the observed changes were due to the addition of numerous small individuals or the loss a few large individuals in a given year. Our new methods employed in 2013-2014, measuring each eel individually, are an improvement on the previous technique and will eliminate this problem.

For two years beginning in 2013 we measured the total length of each eel collected in the Patapsco River. Comparing the distribution of eel total lengths between three areas bounded by the extant Patapsco dams revealed that there are significantly higher numbers of smaller eels in the population downstream of Bloede Dam than there are in the populations between Bloede and Daniels Dams and upstream of Daniels Dam. In other rivers, when a dam has been removed, there has been a "release" of small eels from below the removed dam into the watershed upstream (Hitt et al 2012). We expect a similar result on the Patapsco River when and if Bloede Dam is removed. Knowing the distribution of eel total lengths in each portion of the river now, with the dam in place, will allow us to note changes in those distributions if measurements are continued post-removal. We suspect that, were this scenario to occur, the distributions above and below Bloede Dam would become more similar as additional smaller eels are able to pass upstream.

The continued depression of eel numbers from pre-removal levels downstream of Simkins Dam, as well as the continued decline of eel numbers downstream of Bloede Dam, are likely due to the combined influences of habitat change and migration barriers.

Eel abundance in the vicinity of Simkins Dam has certainly been impacted by habitat change since the removal of Simkins Dam. Eels prefer habitat with ample interstitial spaces (Machut et al. 2007) and dam removals have been shown to temporarily disturb these habitats when impounded sediment is released downstream (Bushaw-Newton et al. 2002, Maloney et al. 2008). When Simkins Dam was removed, large amounts of coarse grained sand contained in the former impoundment were free to move downstream. While sampling, we observed a change from a cobble and gravel substrate (preferred habitat) downstream of Simkins Dam pre-removal (2009-2010) to an all sand habitat post-removal (2011-2012) as this material mobilized. This coincided with a decrease in the number of eels immediately downstream of the dam, at site 502, just after removal in 2011. It took time for the released sand to move through the Bloede Dam impoundment and reach the river downstream, at site 501. Response of eels here was subsequently delayed, as numbers did not show a major decline until 2012 (Figure 5.5).

Our analysis of instream habitat suggests that impacts from the mobilized Simkins Dam sediment may have abated downstream of Simkins Dam at site 502, but not downstream of Bloede Dam at site 501. Habitat scores at site 502 have remained stable at levels similar to those observed pre-removal since recovering from a post-removal low in 2011 (Figure 5.6). Habitat scores at 501, on the other hand, have continually decreased during each year of monitoring. Sediment from the Simkins impoundment was delayed in reaching this area around site 501 to begin with, and some of it is likely still moving through the area. Additionally, the Bloede Dam impoundment itself is filled to its crest with stored sediment (personal observation). This material is free to mobilize during high flow events, washing over the face of the dam and moving downstream.

In addition to storing sediment and potentially prolonging downstream habitat impacts, Bloede Dam is likely slowing the recolonization of eels into areas upstream. As we have seen, the habitat downstream of Simkins Dam at site 502 has recovered (Figure 5.6), while the numbers of eels in this part of the river remain lower than pre-removal levels (Figure 5.5). We suspect that the recovery of eels in this area is being slowed by the presence of Bloede Dam. This barrier is more than twice the height of Simkins Dam, and is the first obstacle encountered by migrating eels. While eels do pass beyond the dam (evidenced by their presence at sites upstream), passage is certainly less efficient than it would be with no dam present. Migrating eels are known to concentrate at the base of barriers in other rivers, reducing their numbers upstream (Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007) .On the Patapsco River, Bloede Dam acts as a hurdle for migrating eels, and could be slowing their dispersal upstream. This will delay their recovery in areas impacted by the Simkins Dam removal, as well as the realization of any benefits to eel passage that the removal might have allowed.

After these latest two years of monitoring, our conclusion on the status of eels in the Patapsco River remains largely the same as the one presented in 2012- Bloede Dam has a major impact that overshadows the removal of Simkins Dam just upstream. Any significant changes in the abundance or size distribution of eels in the Patapsco River may not occur until this major barrier is removed, regardless of the work that's been accomplished above it.

Literature Cited

Bushaw-Newton, K.L., D.D. Hart, J.E. Pizzuto, J.R. Thomson, J. Egan, J.T. Ashley, T.E. Johnson, R.J. Horwitz, M. Keeley, J. Lawrence, D. Charles, C. Gatenby, D.A. Kreeger, T. Nightengale, R.L. Thomas, and D. J. Velinsky. 2002. An integrative approach towards understanding ecological responses to dam removal: the Manatawny Creek study. Journal of the American Water Resources Association 38:1581-1599.

Cairns, D.K., J.C. Shiao, Y. iizuka, W. –N. Tzeng, C.D. MacPherson. 2004. Movement patterns of American eels in an impounded watercourse, as indicated by otolith microchemistry. North American Journal of Fisheries Management 24:452-458.

Ciccotto, Patrick, Scott Stranko, Jim Thompson and William Harbold. 2009. Maryland Biological Stream Survey, Patapsco River Dam Removal Sampling Manual. Report submitted to American Rivers by the Maryland Department of Natural Resources.

Goodwin, K. R., and P. L. Angermeier. 2003. Demographic Characteristics of American Eel in the Potomac River Drainage, Virginia. Transactions of the American Fisheries Society 132:524-535.

Harbold, W., S. Stranko, J. Kilian, M. Ashton, P. Graves. 2013. Patapsco River Dam Removal Study: Assessing Changes in American Eel Distribution and Aquatic Communities. Report submitted to American Rivers by the Maryland Department of Natural Resources.

Haro, A., W. Richkus, K. Whalen, A. Hoar, and W.D. Busch. 2000. Population Decline of the American Eel- Implications for Research and Management. Fisheries Management 25:7-16.

Hitt, N.P., S. Eyler, and J.E.B. Wooford. 2012. Dam removal increases American eel abundance in distant headwater streams. Transactions of the American Fisheries Society 141:1171-1179

Krueger, W. H., and K. Oliviera. 1999. Evidence for environmental sex determination in the American eel, Anguilla rostrata. Environment Biology of Fishes 55: 381-389.

Machut, L.S., K.E. Limburg, R.E. Schmidt, and D. Dittman. 2007. Anthropogenic Impacts on American Eel Demographics in Hudson River Tributaries, New York. Transactions of the American Fisheries Society 136:1699-1713.

Maloney, Kelly O., Hope R. Dodd, Steven E. Butler, and David H. Wahl. 2008. Changes in macroinvertebrate and fish assemblages in a medium-sized river following a breach of a low-head dam. Freshwater Biology 53:1055-1068.

Oliveira, K. 1997. Movements and Growth Rates of Yellow-Phase American Eels in the Annaquatucket River, Rhode Island. Transactions of the American Fisheries Society 126:638-646.

Smogor, R.A., P.L. Angermeier, and C.K. Gaylord. 1995. Distribution and Abundance of American Eels in Virginia Streams: Tests of Null Models across Spatial Scales. Transaction of the American Fisheries Society 124:789-803.

Stranko, S., D. Boward, J. Kilian, A. Becker, M. Ashton, M. Southerland, B. Franks, W. Harbold, J. Cessna. 2015. Maryland Biological Stream Survey: Round Four Field Sampling Manual. http://www.dnr.state.md.us/streams/pdfs/R4Manual.pdf. Maryland Department of Natural Resources, Monitoring and Non-Tidal Assessment Division, Annapolis, Maryland.

Wiley, D.J., R.P. Morgan, R.H. Hilderbrand, R.L. Raesly, and D.L. Shumway. 2004. Relations between physical habitat and American Eel abundance in five river basins in Maryland. Transaction of the American Fisheries Society 124:515-526.

Chapter 6 : Improving Eel Passage around Daniels Dam on the Patapsco River

Introduction

American eels (hereafter eels) are both an important and imperiled species. They are catadromous migratory fish, living and growing to maturity predominantly in freshwater before migrating to the ocean to spawn. They are often a top predator in small headwater streams. Their presence in these habitats can alter the densities of benthic fish and macroinvertebrates (Stranko et al. 2014). Eels are a host for the larvae (glochidia) of freshwater mussels like the eastern elliptio (Lellis et al. 2013, Wiley et al. 2004), serving as vectors of dispersal for this largely sedentary species. They are also a valueable commercially exploited species, although declining populations have diminished their importance from historic highs (Macgregor el a.l 2009).

In recent times, eel abundance has declined throughout the species' entire range. A variety of factors are to blame, including habitat loss, pollution, overfishing, and migration barriers (Haro et al. 2000). Of these, migration barriers might be the most detrimental. There may be as many as 15,115 dams restricting eel passage on the East Coast of the United States (Macgregor et al 2009). As the majority of these dams contain no provisions for fish passage (Haro el al. 2000), this could amount to an exclusion of eels from up to 84% of the habitat once available in rivers and streams of the eastern seaboard (Macgregor et al. 2009).

Aside from merely reducing the total amount of habitat available to eels, dams can impact entire population structures. Eels naturally decrease in density (Smogor et al. 1995, Oliveira 1997, Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007) and increase in size (Machut et al. 2007) with increasing distance upstream, but density tends to decrease most dramatically upstream of dams (Machut et al. 2007). Additionally, densities may be inflated directly downstream of barriers as migrating eels are backed up and concentrated downstream of the blockage (Goodwin and Angermeier 2003, Wiley et al. 2004, Machut et al. 2007). Smaller individuals and males tend to predominate in dense populations downstream of dams, while larger individuals and females predominate in less dense populations upstream (Krueger and Oliviera 1999).

Fecundity in eels is correlated with female size (Barbin and McCleave 1997), being highest in the largest individuals. By skewing sex ratios and reducing access to habitats preferred by the largest, most fecund females, dams could potentially alter the reproductive potential of the entire population within a watershed.

Dams and their impacts have influenced the status of eels in the Patapsco River. The river has been referred to as one of the most dammed in the United States, its valley forming what was once the cradle of the Industrial Revolution in Maryland (Travers 1990). The dams that powered industry in the Patapsco River valley have remained long after the factories and mills they powered moved on. Recently, there has been a trend toward removing these abandoned relics. Two of them, Simkins Dam and Union Dam, were removed from the Patapsco River in 2009 and 2010, respectively. A third, Bloede Dam, is scheduled to be removed in 2016. This will leave Daniels Dam as the first barrier to eels migrating up the Patapsco River.

In an effort to improve the ability of the Patapsco River to support a healthy eel population, a goal has been set to improve eel passage around Daniels Dam. Improvements to eel passage can have dramatic and wide ranging impacts on eel populations throughout the watershed. When Embery Dam was removed from the Rappahannock River in 2004, increases in abundance of eels were noticed in tributaries as far as 150 km upstream of the former dam location (Hitt et al. 2012). While not as effective at improving passage as removing a dam entirely, eel ladders have the capacity to provide significant improvements. An eel ladder constructed on the Arzal Dam in France in 1996 saw an increase in eel abundance upstream of the dam up to six times higher than pre-construction levels within two years of completion (Briand et al. 2005). Since removing Daniels Dam is not currently an option, we have constructed an eel ladder to allow easier passage for eels around this barrier.

By constructing an eel ladder on Daniels Dam, we hope to reduce the dam's status as a migration barrier as well as its associated negative impacts on the Patapsco River's eel population. Should the ladder prove successful, we expect that over time there will be noticeable changes in both the abundance and size structure of eels throughout the river and tributaries upstream.

Methods

Our facilitation of eel passage over Daniels Dam occurred in two phases- active transport by capture and release in 2013 and passive transport using the eel ladder we constructed in 2014. Catching and releasing eels in 2013 was conducted as "emergency" measure after it became evident that the eel ladder would not be completed on schedule. This ensured that at least some improvement in eel passage occurred during 2013. We did not continue active transport of eels over Daniels Dam in 2014, as the ladder was operational at this time and passive transport was possible.

We conducted active transport of eels on 22 October 2013. We used two backpack electrofishers to capture eels in the Patapsco River within 200m of the base of Daniels Dam. All eels were individually measured (total length in millimeters) before being released upstream of the dam.

To facilitate passive transport of eels over Daniels Dam we built an eel ladder during the spring and summer of 2014. We started construction on 13 March 2013 and completed the ladder on 10 June 2014. The ladder consisted of a 12" wide covered steel channel lined with Enkamat® erosion control mat as a climbing substrate. The channel was mounted on the downstream side of Daniels Dam, ascending the entire height of the blockage before descending into a holding tank at the top of the dam (Figure 6.1).

Figure 6.1: The eel ladder constructed on Daniels Dam during spring 2014.

Water from the Daniels Dam impoundment was pumped to the top of the ladder and distributed by a spray bar down both sides of the ladder- the ascending side to facilitate eels during the climb and the descending side to wash them into the holding tank. Eels that were washed into the holding tank were collected in a mesh bag to be counted and released daily. The ladder ran continuously from 10 June to 1 October 2014. On 1 October 2014 we placed the pumps, hoses, and holding tank in winter storage. The ladder will be operated again during the spring and summer of 2015.

We are currently monitoring eel abundance and size throughout the river to assess the impacts of dam removals downstream (Chapter 5). Three sites used in this monitoring project since at least 2013 are upstream of Daniels Dam and can also be used to assess impacts from improved eel passage (Figure 6.2). Data collected at these sites both before and after the completion of the eel ladder will allow us to see any changes in eel abundance and size structure upstream of Daniels Dam that may occur as passage is improved.

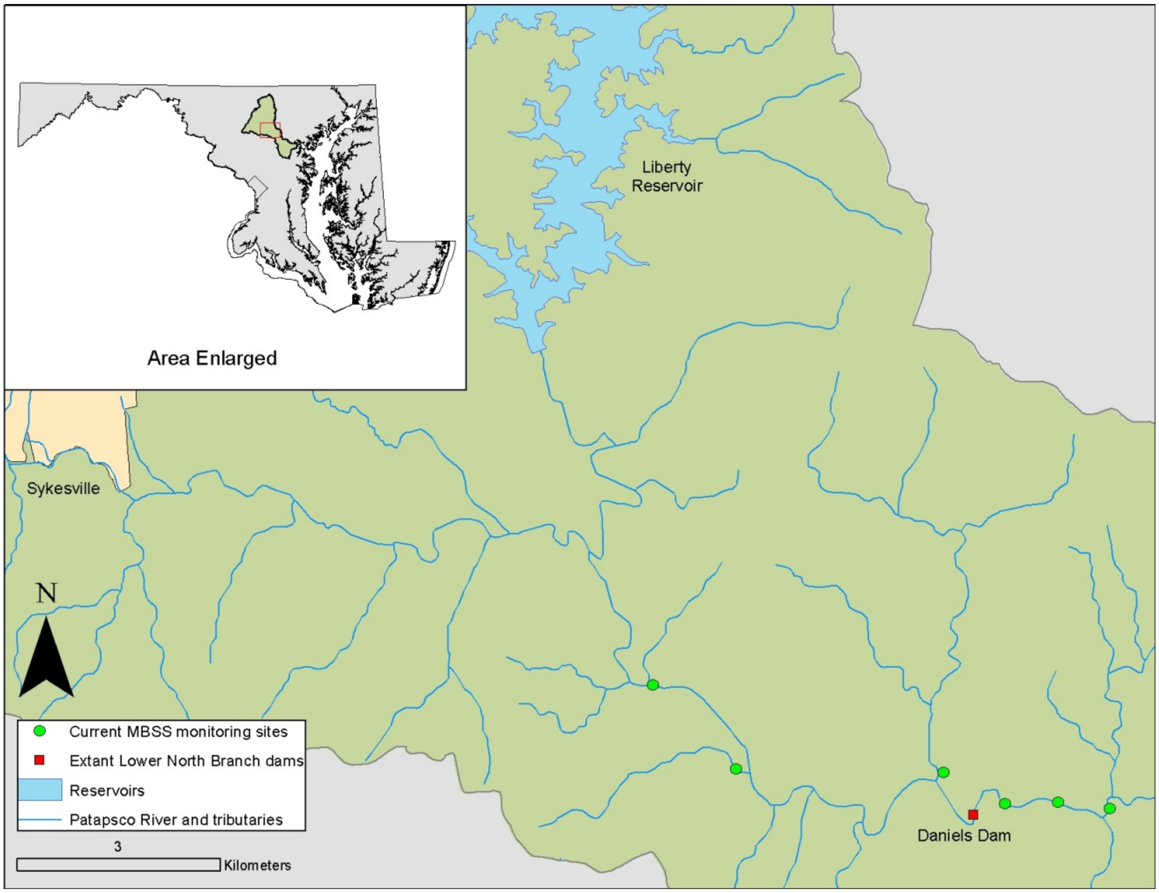

Figure 6.2: Existing monitoring sites in the vicinity of Daniels Dam used to assess impacts of improved eel passage after constructing an eel ladder.

Results

On 22 October 2013 we collected eels downstream of Daniels Dam and moved them above the blockage. We electrofished for 2.5hrs, collecting 85 eels from the wadeable habitat within 200m of the base of the dam. We measured each eel (total length, millimeters) and released them into the impoundment upstream of the dam. The eels measured between 99mm and 535mm total length with a mean of 219mm and a median of 152mm (Figure 6.3).

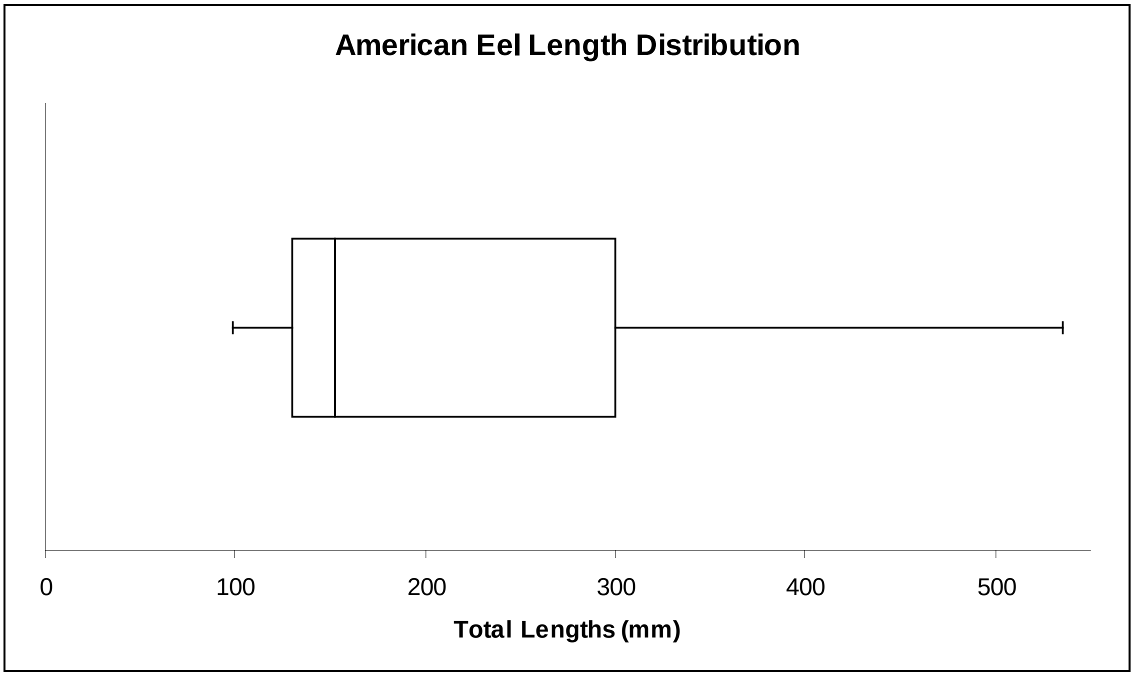

Figure 6.3: Distribution of total lengths of American eels collected downstream of Daniels Dam and relocated upstream of the blockage on 22 October 2013.

Eel ladder construction was completed on 10 June 2014. The ladder was turned on at 8:00AM on 10 June 2014 and remained in continuous operation until 3PM on 1 October 2014. During this time 14 eels passed over Daniels Dam by way of the eel ladder (Table 6-1). All the eels were small, between 90mm and 110mm total length (Figure 6.4).

Table 6-1: American Eels Passed at the Daniels Dam Eel Ladder

Date	**Number of Eels**
24 July	1
28 July	2
6 August	1
11 August	4
19 August	1
27 August	2
3 September	3
Total	**14**

Figure 6.4: Two American eels that used the ladder to bypass Daniels Dam.

Between 22 October 2013 and 1 October 2014 we moved (or facilitated the movement of) 99 eels over Daniels Dam. Despite this, there was no increase in eel numbers corresponding to these movements evident at monitoring sites sampled upstream during the summer of 2014.

Conclusions

While our eel ladder has proven to be functional in passing eels, it is too early and too few eels have been passed to determine how successful it may be in alleviating the impacts of Daniels Dam as a migration barrier. We suspect that there are two possible explanations for why so few eels were observed using the ladder- delays in completing the ladder may have caused us to miss the bulk of the spring migration for 2014 and eels that are reaching the dam are finding an alternative way around it.

Despite our best efforts, the eel ladder was not completed and operational until 10 June 2014. It is possible that by this time large numbers of eels had already migrated, missing the opportunity to utilize our ladder. Upstream eel migrations on the Shenandoah River in West Virginia are associated with increasing temperatures and high flows during the spring (Hammond 2003). By June 2014 river temperatures had been warming for some time, and there had been a large flow event on April 29-30 (Harbold et al. 2014). If large numbers of eels attempted to migrate past the dam at this time they would have been missed.

Even eels that attempted migration past Daniels Dam after the ladder was in place might have found another way around, either through the dam's fish ladder or some other route. Eels are present at every site monitored to assess the impacts of Patapsco River dam removals (Chapter 5), including three that are upstream of Daniels Dam. Additionally, eels have been observed at sixteen other sites sampled by the Maryland Biological Stream Survey between 1995 and 2014, showing that they are present, at least to some degree, throughout the watershed from Daniels Dam up to two impassable barriers at reservoirs further upstream (Figure 6.5). Since eel passage at Daniels Dam prior to our installation of the eel ladder was clearly possible, all we can definitively say so far after this first year's effort is that we have augmented their upstream numbers with the addition of 99 individuals.

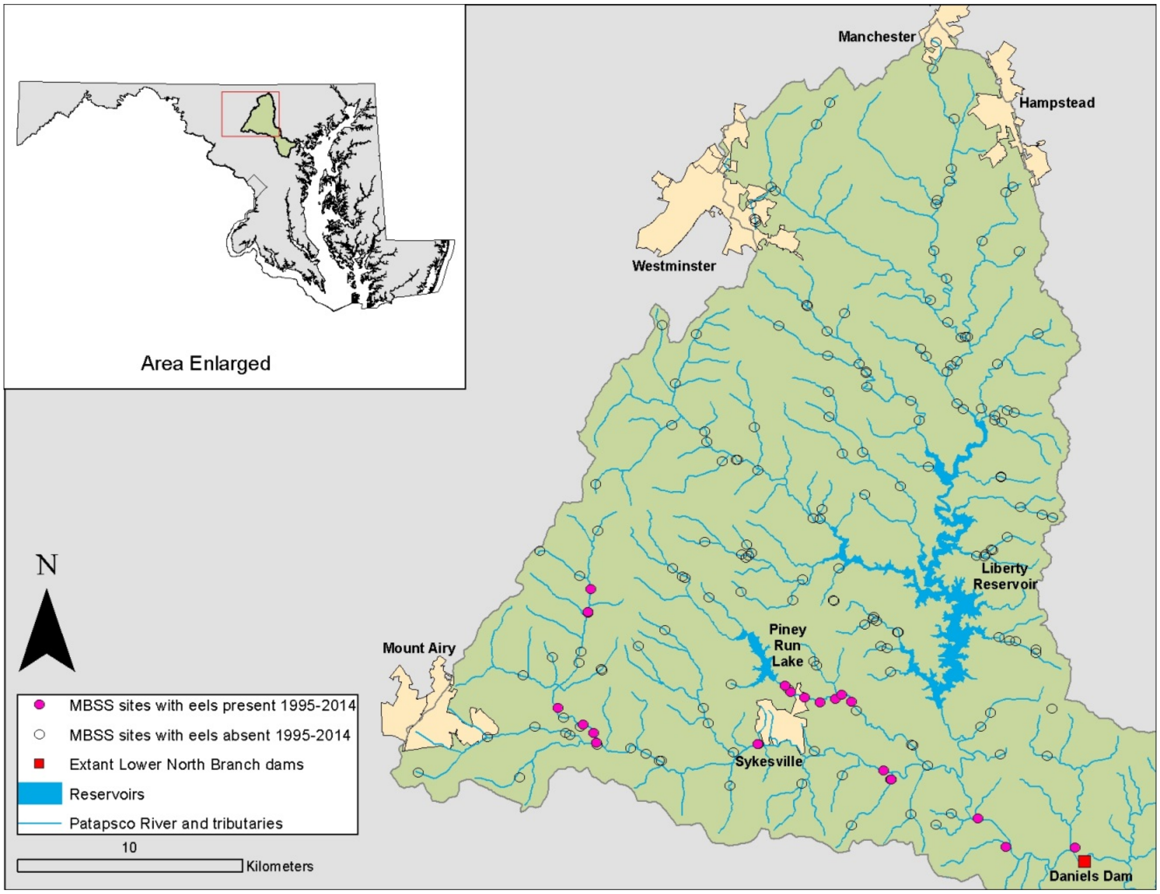

Figure 6.5: Locations of American eel records in the MBSS data set from the Patapsco River watershed upstream of Daniels Dam.

The fourteen eels that used the ladder in 2014 and the 85 that we actively moved in 2013 are too few to cause a noticeable impact in abundance or size structure at our monitoring sites upstream. There are approximately 225km of habitat (river and tributaries) available to eels upstream of Daniels Dam (Figure 6.6). We have no way of knowing where these eels went when released above the dam (if they even remained upstream), but it is safe to conclude that an influx of 99 eels would be too diluted over such a large area to create any sort of meaningful signal at our monitoring sites.

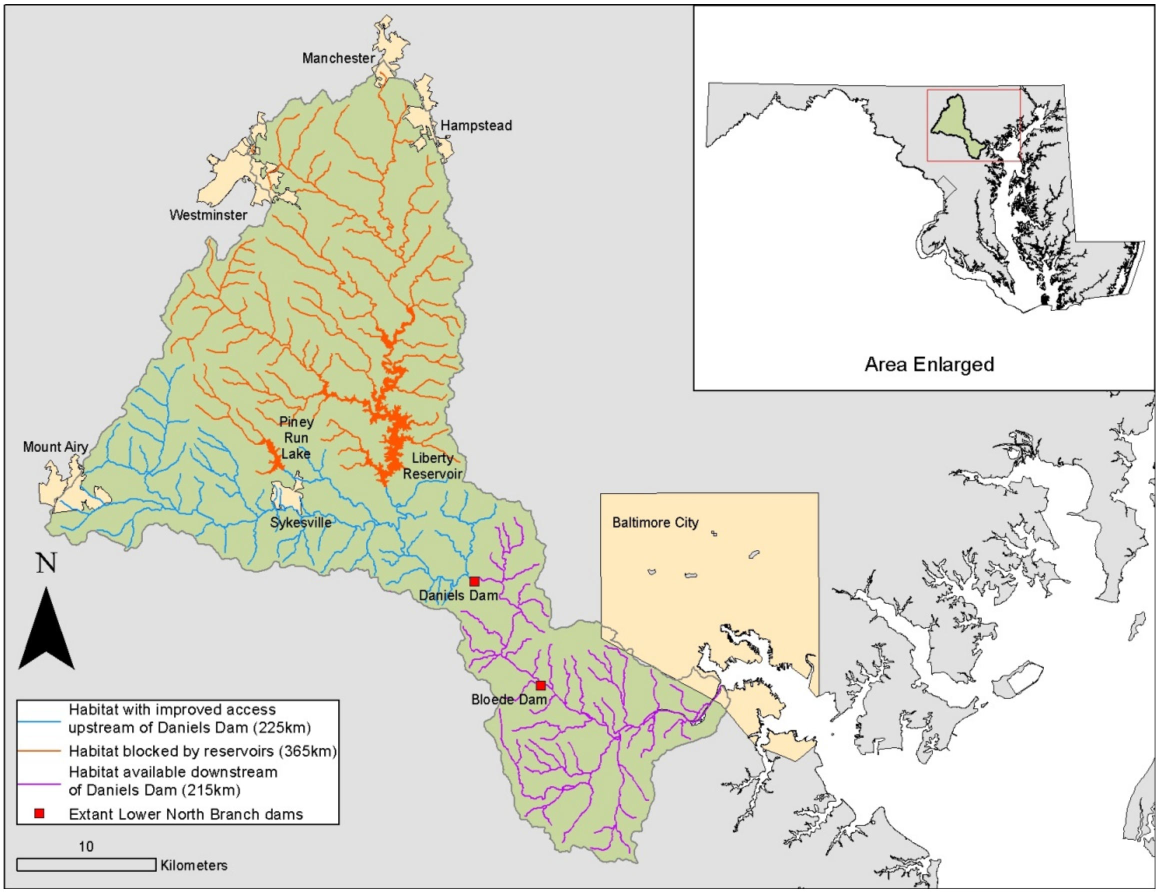

Figure 6.6: Stream habitat within the Patapsco River watershed and its availability to American eels relative to extant dams and reservoirs.

We plan to continue operating the eel ladder at Daniels Dam and monitoring eel abundance and size at sites upstream for at least the next two years. Once Bloede Dam is removed, Daniels Dam will be the first man-made obstacle encountered by eels migrating upstream from the Chesapeake Bay, with over half of the habitat available to eels in entire watershed located beyond it (Figure 6.6). By improving passage for eels over this barrier, we will allow easier access to large portions of otherwise restricted habitat, a great benefit to this important species in the Patapsco River watershed.

Literature Cited

Barbin, G.P. and J. D. McCleave. 1997. Fecundity of the American eel Anguilla rostrata at 45°N in Maine, USA. Journal of Fish Biology 51:840-847.

Briand, C., D. Fatin, G. Fontenelle, E. Feunteun. Effect of Re-Opening of a Migratory Pathway for Eel (Anguilla anguilla, L.) at a Watershed Scale. Bulletin Francais De La Peche Et De La Pisciculture. 2005. 378-379:67-86.

Goodwin, K. R., and P. L. Angermeier. 2003. Demographic Characteristics of American Eel in the Potomac River Drainage, Virginia. Transactions of the American Fisheries Society 132:524-535.

Hammond, S.D. Seasonal movements of yellow-phase American eels (Anguilla rostrata) in the Shenandoah River, West Virginia. Thesis submitted to the Davis College of Agriculture, Forestry, and Consumer Sciences at West Virginia University. 2003.

Harbold, W., J. Kilian, S. Stranko. 2014. Report on ecological sampling in the Patapsco River during April 9 – May 7, 2014. Report submitted to the Maryland Environmental Service by the Maryland Department of Natural Resources.

Haro, A., W. Richkus, K. Whalen, A. Hoar, and W.D. Busch. 2000. Population Decline of the American Eel- Implications for Research and Management. Fisheries Management 25:7-16.

Hitt, N.P., S. Eyler, and J.E.B. Wooford. 2012. Dam removal increases American eel abundance in distant headwater streams. Transactions of the American Fisheries Society 141:1171-1179

Krueger, W. H., and K. Oliviera. 1999. Evidence for environmental sex determination in the American eel, Anguilla rostrata. Environment Biology of Fishes 55: 381-389.

Lellis, W.A., B.S.J. White, J.C. Cole, C.S. Johnson, J.L. Devers, E.V.S. Gray, and H.S. Galbraith.

2013. Newly documented host fishes for the Eastern Elliptio Mussel (Elliptio complanata). Journal of Fish and Wildlife Management 4(1):75–85.

MacGregor, R., J.M. Casselman, W.A. Allen, T. Haxton, J.M. Dettmers, A. Mathers, S. LaPan, T.C. Pratt, P. Thompson, M. Stanfield, L. Marcogliese, J.-D. Dutil. Natural Heritage, Anthropogenic Impacts, and Biopolitical Issues Related to the Status and Sustainable Management of American Eel: A Retrospective Analysis and Management Perspective at the Population Level. 2009. American Fisheries Society Symposium. 69:713-740.

Machut, L.S., K.E. Limburg, R.E. Schmidt, and D. Dittman. 2007. Anthropogenic Impacts on American Eel Demographics in Hudson River Tributaries, New York. Transactions of the American Fisheries Society 136:1699-1713.

Oliveira, K. 1997. Movements and Growth Rates of Yellow-Phase American Eels in the Annaquatucket River, Rhode Island. Transactions of the American Fisheries Society 126:638-646.

Smogor, R.A., P.L. Angermeier, and C.K. Gaylord. 1995. Distribution and Abundance of American Eels in Virginia Streams: Tests of Null Models across Spatial Scales. Transaction of the American Fisheries Society 124:789-803.

Stranko, S., D. Boward, J. Kilian, A. Becker, M. Ashton, M. Southerland, B. Franks, W. Harbold, J. Cessna. 2015. Maryland Biological Stream Survey: Round Four Field Sampling Manual. http://www.dnr.state.md.us/streams/pdfs/R4Manual.pdf. Maryland Department of Natural Resources, Monitoring and Non-Tidal Assessment Division, Annapolis, Maryland.

Stranko, S. M. Ashton, R. Hilderbrand, S. Weglein, D. Kazyak, J. Kilian. 2014. Fish and Benthic Macroinvertebrate Densities in Small Streams with and without American Eels. Transactions of the American Fisheries Society. 143: 700-708.

Travers, P.J. 1990. The Patapsco: Baltimore's River of History. Tidewater Publishers, Centreville, MD. 220 pp.

Wiley, D.J., R.P. Morgan, R.H. Hilderbrand, R.L. Raesly, and D.L. Shumway. 2004. Relations between physical habitat and American Eel abundance in five river basins in Maryland. Transaction of the American Fisheries Society 124:515-526.

Chapter 7 : Response and recovery of resident fish assemblages of the Patapsco River following the removal of Simkins Dam

Introduction

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 and become mobilized into 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 convert 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 re-colonize from adjacent reaches (Catalano et al. 2007; Gardner et al. 2011). Conversely, dam removal can have short term negative effects on downstream fish assemblages. Aggradation of sediment in downstream reaches can damage fish spawning habitats and reduce cover and prey availability (Bednarek 2001). This can affect species in a variety of ecological guilds, reduce overall species richness, density, and biomass, and can impact important recreational game fisheries (Bushaw-Newton et al. 2002; Catalano et al. 2007; Doeg and Koehn 1994; Kanehl et al. 1997; Maloney et al. 2008).

To document changes in stream fish assemblages associated with the removal of Simkins Dam, we quantitatively surveyed stream fishes at five sites in the Patapsco River for two years (2009-2010) prior to and four years (2011-2014) following its removal. Harbold et al. (2013) provided a previous summary of the results of these surveys through 2012 and described notable changes in fish assemblages attributable to dam removal, especially at sites downstream of Simkins Dam. Specifically, these surveys documented changes in species richness, fish density, and biomass at sites downstream of the dam where the quality and quantity of fish habitat changed substantially as the river bottom shifted from predominately coarse (i.e., cobble, boulder) to finer substrate (mostly sand). We also detected shifts in specific ecological guilds including declines in the number of lithophilic spawning species and benthic riffle species - those most sensitive to sedimentation and changes in substrate composition. Lower numbers of pollution- intolerant species and smallmouth bass were also observed downstream following dam removal.

Previous studies investigating the ecological impacts of dam removal have noted similar responses in fish assemblages to the immediate changes in geomorphology, hydrology, and substrate composition that occur in affected areas (Doyle et al. 2005; Catalano et al. 2007; Gardner et al. 2011). Many of these studies have also documented recovery of fish assemblages following dam removal as river conditions adjust and stabilize. In this chapter, we further examine the response of fish assemblages in the Patapsco River to dam removal using data collected annually over four years (2011-2014). We also use these data to evaluate for recovery (i.e., return to pre-removal baseline levels) in the attributes of these assemblages most affected by the removal of Simkins Dam.

Methods

Fish Sampling

We conducted quantitative surveys of fish assemblages annually at five sites in the Patapsco River mainstem from 2009 to 2014 (Figure 7.1). Fish surveys were conducted during the summer (June – September) of each year following protocols described in Harbold et al. (2013). With the exception of two sites (described below), all sites were surveyed twice prior to, and four times following, dam removal. Site 511 was surveyed only once (in 2010) prior to the removal of Simkins Dam. Site 504 was not quantitatively surveyed in 2014.

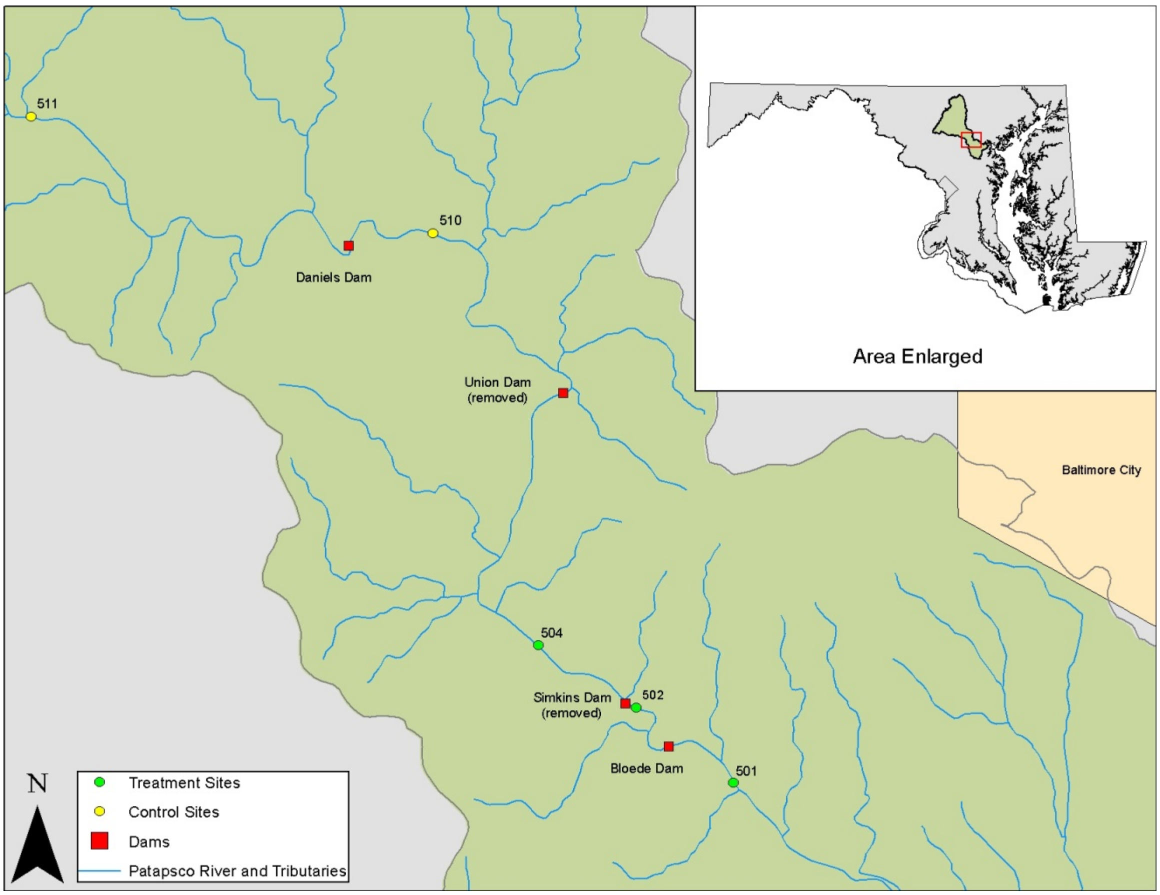

Figure 7.1: Locations of quantitative surveys of fish assemblages in the Patapsco River mainstem from 2009 to 2014.

Study Design and Data Analyses

We used a before-after-control-impact (BACI) design to assess changes in stream fish assemblages associated with the removal of Simkins Dam. This approach allowed us to differentiate changes in fish assemblages associated with the removal of Simkins Dam from natural variability. We compared pre- (2009-2010) to post-dam (2011-2014) removal assemblages at three treatment sites adjacent 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 (Figure 7.1). Site 502, located 0.2 river kilometers (Rkm) downstream of Simkins Dam experienced a dramatic shift from predominately coarse substrate to fine substrate within the first two years (2011-2012) following dam removal. Site 501, located 2.5 Rkm downstream of Simkins Dam and 1.3 Rkm below Bloede Dam also experienced a shift in substrate composition during the study period; however this shift was delayed in comparison to site 502 due mostly to the presence of Bloede Dam which affected the downstream movement of sediment. Sedimentation of substrate that occurred at these two downstream treatment sites in the first few years following dam removal has largely abated. As of 2014, habitat quality at both sites returned to pre-removal conditions as much of the sediment has moved further downstream out of the study area. The third treatment site, 504, located 2.2 Rkm upstream of Simkins Dam (Figure 7.1), was upstream of the Simkins Dam impoundment and above areas that underwent significant erosion following dam removal.

We used data collected prior to and following dam removal at two control sites (510 and 511) to assess natural variability in assemblages occurring during the study period (Figure 7.1). 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 (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 (2009-2014) was assumed to be natural and unrelated to dam removal.

For our analyses, we compared pre- and post- dam removal fish assemblages at the two downstream treatment sites (i.e., 501, 502) and examined any observed changes in fish assemblage similarity, fish density, biomass, and other assemblage metrics in relation to natural changes observed at the downstream control site (510). Similarly, we examined pre- and post-dam removal fish assemblages at the upstream treatment site (504) in relation to natural changes observed at the upstream control site (511). As mentioned previously, the upstream control site (511) was sampled only once prior to dam removal. To assess change at this site, post-removal conditions were compared to conditions observed in 2010. Changes in fish assemblages observed at treatment sites that exceeded in magnitude or were opposite the natural changes that were observed at control sites during the same period were attributed to dam removal.

Fish Assemblage Similarity:

We used Sorensen's Similarity Index (following Krebs 1989, Gardner et al. 2011) to evaluate changes in fish assemblage composition at each of the three treatment sites and two control sites through time. We compared fish species presence/absence data collected annually following the removal of Simkins Dam (2011 – 2014) to data collected from the same site prior to its removal (2009-2010). Sorensen's index scores range from 0.0 for assemblages that are completely dissimilar to 1.0 for assemblages that are identical.

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 five 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 treatment sites in relation to changes observed at the two control sites.

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 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 calculated the average number of species observed during the pre-removal baseline period (2009-2010). We then calculated the net change from baseline condition (post – pre) at each site for each of the four post-removal years. For density of non-native species, we summed abundance of all non-native species (with the exception of Smallmouth Bass) 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-removal periods, and calculated net change (post- pre) for each site by year.

Smallmouth Bass Populations:

We compiled data on Smallmouth Bass collected from the five mainstem sites from 2009 to 2014. Throughout the study period, Smallmouth Bass abundance increased with distance upstream in the Patapsco River (see Harbold et al. 2013). To account for this effect, we normalized abundance data by calculating annual percent change from baseline conditions in bass abundance per size class at each site. The control sites were used to assess natural variability in bass populations occurring during the study period. 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 Maryland Biological Stream Survey from 2000 to 2011. Stock and Harvestable classes were defined following Gabelhouse (1984).

Results

Fish Assemblage Similarity:

Fish assemblage species composition at all five sites varied over the course of the study period as measured by the Sorenson's Similarity Index. Natural annual variability measured at the upstream and downstream control sites ranged from 5% (0.95) to 14% (0.86). The greatest pre- to post-removal changes in assemblage similarity were observed at the two downstream sites - 501 and 502, with the largest change of 26% (0.74) observed at Site 502, located closest to the dam (Figure 7.2). Species composition at Site 502 following dam removal was most dissimilar from pre-removal composition in 2012; however Sorenson's Similarity Index scores at this site increased in 2013 and 2014. The largest change in species composition at Site 501 occurred in 2013 (0.21). As was observed at Site 502, the Sorenson's Similarity Index scores increased at Site 501 in 2014 (0.84). The fish assemblage at Site 504 varied little in species composition throughout the study period.

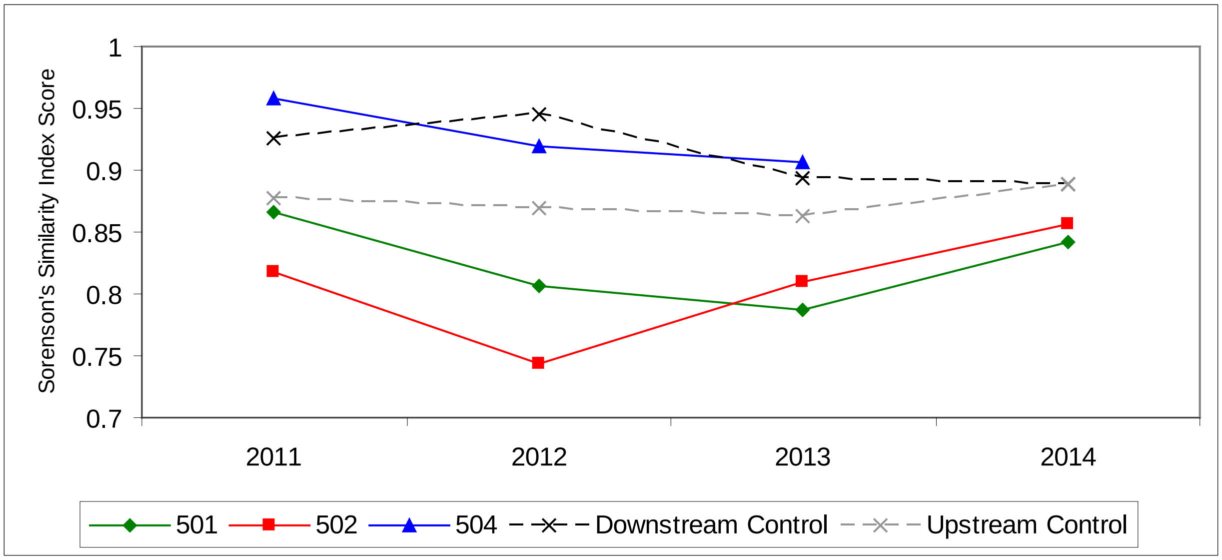

Figure 7.2: Post-dam removal change in fish assemblage species composition (Sorenson's Similarity Index scores) at the three treatment and two control sites.

Fish Density and Biomass:

Total fish density at all five Patapsco River sites sampled during this survey ranged from 0.18 to 0.90 individuals/m^2. Fish density at the downstream control site increased each year from 0.14 - 0.50 individuals/m^2 over densities measured at this site prior to dam removal. Both treatment sites located below Simkins Dam exhibited a decrease in the initial years following its removal; however the timing of the decline varied between the two sites (Figure 7.3a). Fish densities at Site 502, located closest to Simkins Dam, declined by 0.06 and 0.17 individuals/m^2 in 2011 and 2012, respectively. In 2013 and 2014, fish densities at this site were higher than measured prior to dam removal. Fish density at Site 501 was similar to that observed at the downstream control site in the first year following dam removal (2011). However, fish densities at this site were equal to (2013) or below pre-removal densities (2012 and 2014) – a pattern opposite to that observed at the downstream control site. Unlike Site 502, fish densities at Site 501 have not returned to pre-removal levels. Fish densities measured at Site 504, located upstream of Simkins Dam, were higher each year (2011-2013) following dam removal than densities measured prior to dam removal (Figure 7.3b). This change was greater in magnitude than changes observed at the upstream control site (Site 511).

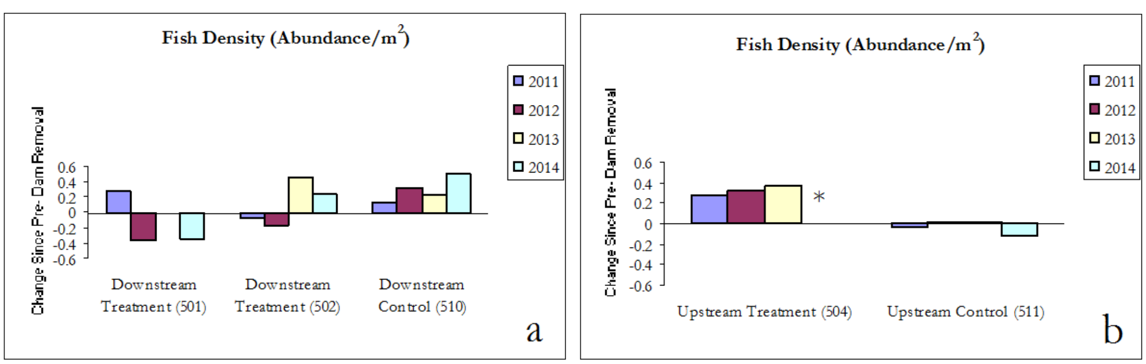

Figure 7.3: Post-dam removal change in fish density at sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site 504 was not sampled in 2014.

Total fish biomass per site at the five Patapsco River sites sampled during this survey ranged from 3.44 to 12.47 g/m^2. At both downstream treatment sites (501 and 502), we documented lower fish biomass each year following dam removal than was measured at each of these sites prior to the removal of Simkins Dam (Figure 7.4a). This decline in biomass was greater than that observed at the downstream control site during the same period. The decline in biomass was greatest for both downstream treatment sites in 2012. Although fish density increased at the upstream treatment site (504) following dam removal, we detected an overall decline in fish biomass during the same period (Figure 7.4b). This change exceeded that measured at the upstream control site from 2011 – 2013.

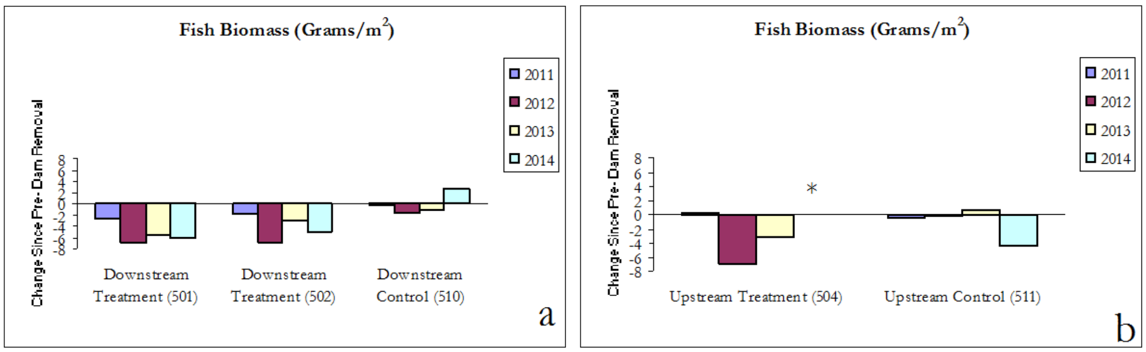

Figure 7.4: Post-dam removal change in fish biomass sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site504 was not sampled in 2014.

Ecological Composition:

Benthic Riffle Species: Benthic riffle species (following Bushaw-Newton et al. 2002) are those species that require clean, coarse substrate as refuge and for feeding. In the Patapsco River, these species include the Shield Darter (*Percina peltata*), Central Stoneroller (*Campostoma anomalum*), and Blue Ridge Sculpin (*Cottus caeruleomentum*). We documented a decline in these species at downstream sites following dam removal (Figure 7.5a). This change was opposite to that observed at the downstream control site during the study period. The greatest change in the number of benthic riffle species measured at the downstream sites (501 and 502) occurred in 2012, with the loss of Shield Darter and Blue Ridge Sculpin from Site 502 and the loss of Shield Darter, Central Stoneroller, and Blue Ridge Sculpin from Site 501. Since 2012, the numbers of benthic riffle species have increased at both downstream sites, but have not yet returned to pre-removal levels. The number of benthic riffle species following dam removal was highly variable at the upstream treatment site (504). This variability was greater than that observed at the upstream control site where no changes were observed from 2012 – 2014 (Figure 7.5b).

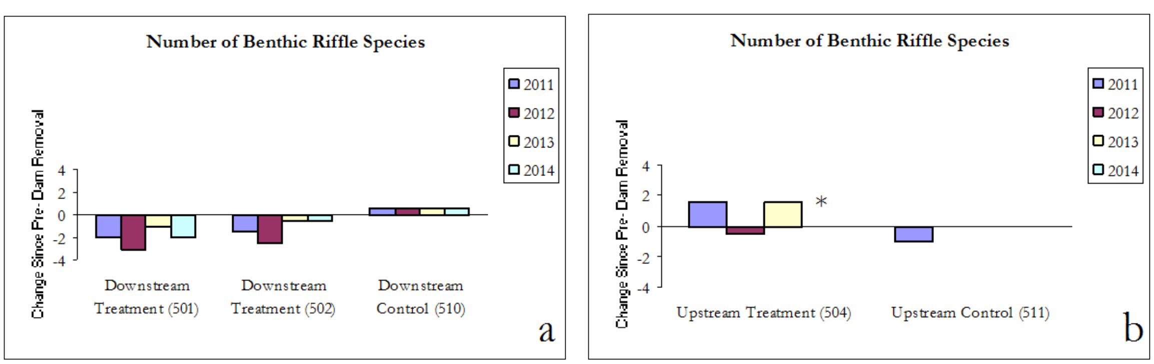

Figure 7.5: Post-removal change in the number of benthic/ riffle species at sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site504 was not sampled in 2014.

Lithophilic Spawners: There were a total of 17 lithophilic spawning species documented at the five Patapsco River sites sampled during the study period. These are species that require clean, coarse substrate for successful spawning. These species do not build nests or provide parental care and tend to be sensitive to sedimentation. Following dam removal, we documented an overall decline in lithophilic spawning species at site 501, with the greatest loss of these species occurring in 2012 at this site (Figure 7.6a). At Site 502, we documented a slight increase in lithophilic spawners in 2011, but no change in subsequent years beyond that which occurred naturally as measured at the downstream control site. Lithophilic spawning species increased at the upstream treatment site (504) over the study period (Figure 7.6b).

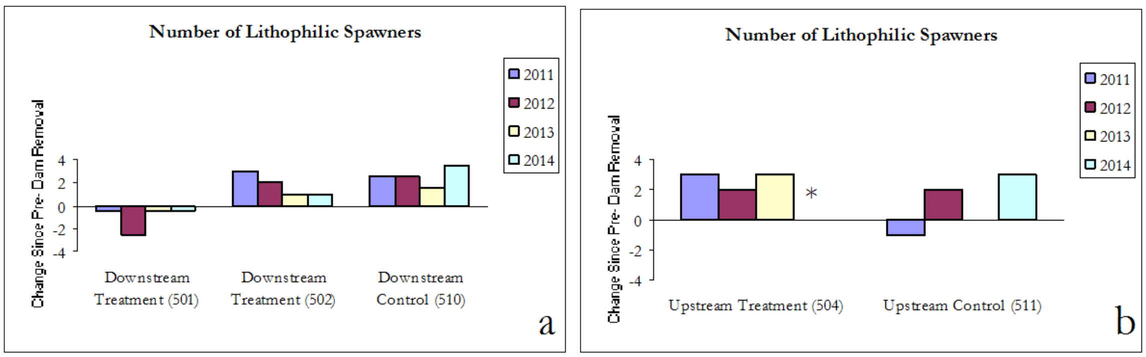

Figure 7.6: Post-removal change in the number of lithophilic spawning species at sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site504 was not sampled in 2014.

Intolerant Species: There were a total of 13 intolerant species documented at the five Patapsco River sites sampled during the study period. Intolerant fish species are those that are sensitive to environmental disturbance and tend to decline in abundance or disappear from degraded stream habitats. Following dam removal, we documented an overall decline in intolerant species at site 501 – a trend opposite to that observed the downstream control site (Figure 7.7a). The greatest decline in intolerant species at this site occurred in 2012. We observed no change in the number of intolerant species at Site 502 in comparison to natural variability observed at the downstream control site. As with lithophilic spawning species, we documented an overall increase in intolerant species at the upstream treatment site (504) over the study period (Figure 7.7b). This change was greater in magnitude or opposite than what was observed at the upstream control site during the same period.

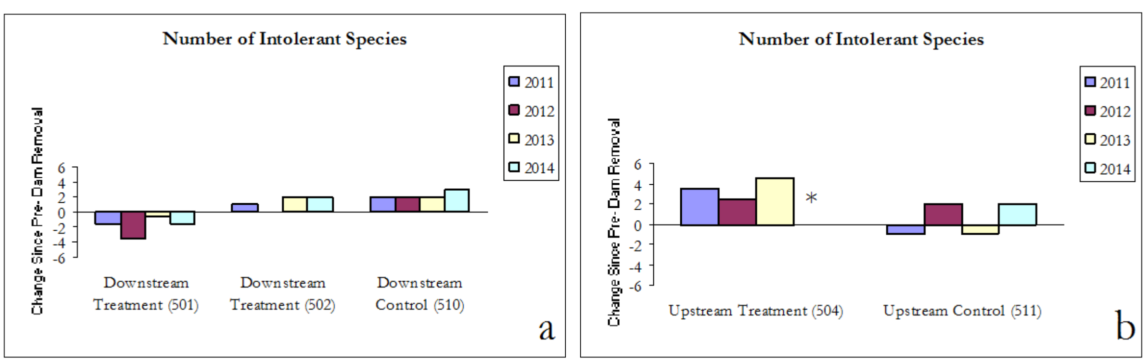

Figure 7.7: Post-removal change in the number of intolerant species at sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site504 was not sampled in 2014.

Tolerant Species: There were a total of 10 tolerant species documented at the five Patapsco River sites sampled during the study period. Tolerant fish species are those that can adapt to a variety of environmental conditions and are usually common to abundant in areas that are heavily disturbed by anthropogenic stressors. Following dam removal, we documented an increase in tolerant species at Site 501 in 2011, followed by a decline in 2012 (Figure 7.8a). In subsequent years, the number of tolerant species at this site did not vary from pre-removal baseline conditions or from what was measured at the downstream control site. At Site 502, we documented an increase in the number of tolerant species in 2011 and 2012, with numbers returning to baseline levels in 2013 and 2014. We noted no change in the number of tolerant species at the upstream site (504) attributable to dam removal (Figure 7.8b).

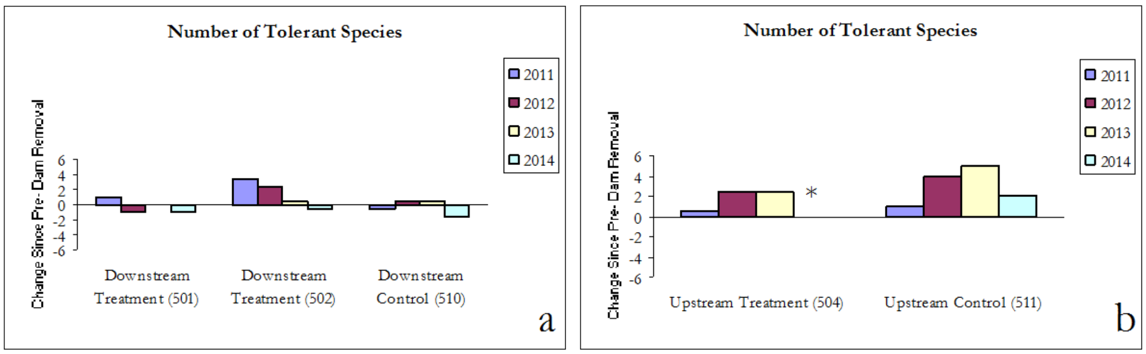

Figure 7.8: Post-removal change in the number of tolerant species at sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site504 was not sampled in 2014.

Non-native Species Density: With the exception of Smallmouth Bass (discussed below), there were a total of 11 non-native species documented during the course of the study at the five Patapsco River sites including five sunfish (Centrarchidae), two trout (Salmonidae), two minnows (Cyprinidae), Channel Catfish (*Ictalurus punctatus*, Ictaluridae), and Oriental Weatherfish (*Misgurnus anguillicaudatus*) (Cobitidae). Although non-native fish densities increased slightly in 2013 at Site 501, we documented no changes in the density of non-native fishes attributable to dam removal at downstream sites (Figure 7.9a). We documented declines in non-native fish density (specifically Green Sunfish (*Lepomis cyanellus*) and Rock Bass (*Ambloplites rupestris*) at the upstream treatment site (504) following dam removal in 2011 and 2012(Figure 7.9b). These declines were greater in magnitude than that observed at the upstream control site.

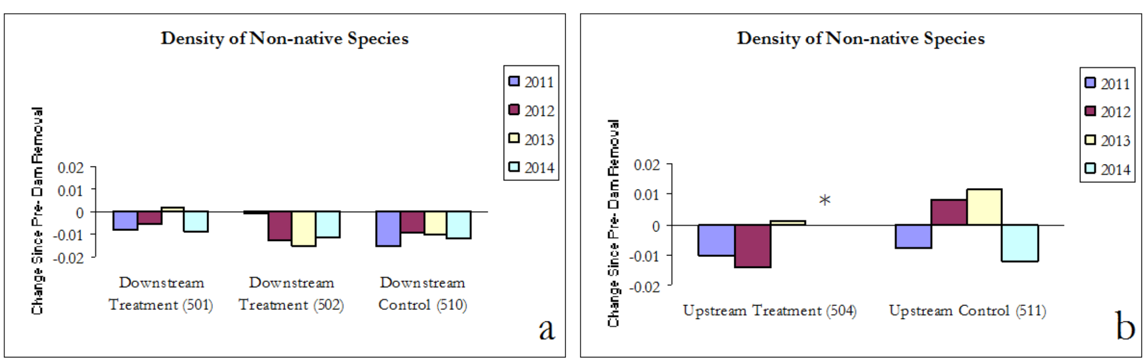

Figure 7.9: Post-removal change in non-native species density at sites downstream of Simkins Dam (501, 502), upstream of Simkins Dam (504), and control sites. * Site504 was not sampled in 2014.

Smallmouth Bass Populations:

Smallmouth Bass populations at each of the five sites were comprised predominately of YOY individuals followed by individuals within the Stock size class. We collected no individuals within the harvestable size class during the study period. We documented a decline in Smallmouth Bass at both downstream treatment sites associated with the removal of Simkins Dam; however the timing of the decline varied between the two sites. Bass populations at Site 502, located closest to Simkins Dam, appeared to be most affected initially following dam removal (Figure 7.10a) – with changes in YOY abundance of Smallmouth Bass observed within one year (Figure 7.11a). Total abundance at this site declined to zero in 2012 and remained below pre-removal levels through 2013 (Figure 7.10a). YOY abundance declined initially at Site 501 (Figure 7.11a), but changes in the total abundance of bass at this site were not detected until 2013 (Figure 7.11a). By 2014, total abundance and numbers of YOY bass at both downstream treatment sites had returned to pre-removal baseline levels. However, stock-sized bass have not been collected at Site 502 since 2011 (Figure 7.12). Bass abundance at the upstream treatment site (504) showed trends similar to but greater in magnitude than the upstream control site in 2012 and 2013 (Figure 7.11b). YOY abundance increased by over 500 percent in 2013 – an increase far greater than that observed at the control site (Figure 7.11b).

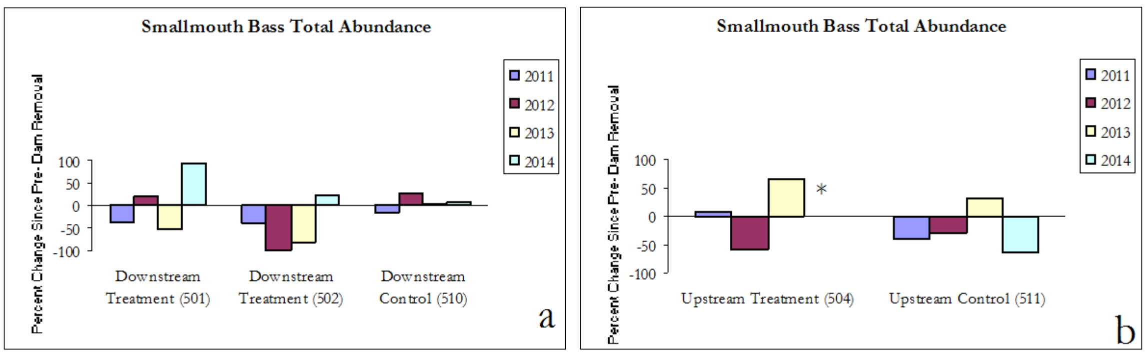

Figure 7.10: Post-removal percent change in Smallmouth Bass abundance at Patapsco sites upstream of Simkins Dam (504), downstream of Simkins Dam (501, 502) and control sites. * Site504 was not sampled in 2014.

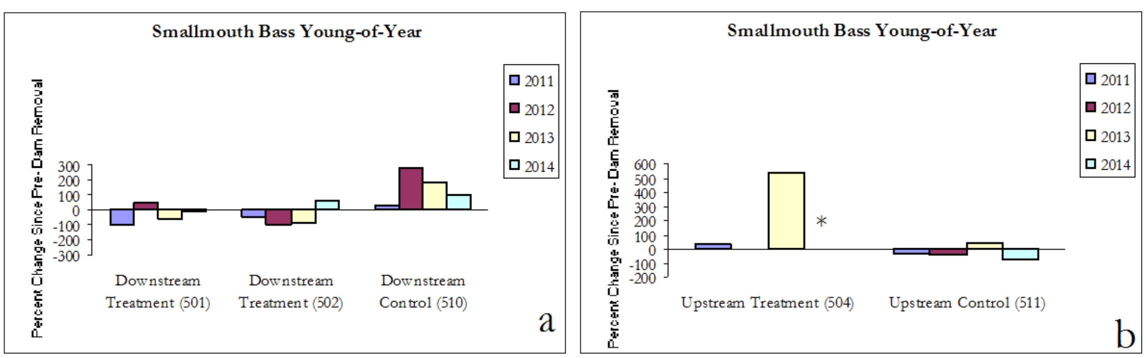

Figure 7.11: Post-removal percent change in Smallmouth Bass young-of-year abundance at Patapsco sites upstream of Simkins Dam (504), downstream of Simkins Dam (501, 502) and control sites. * Site504 was not sampled in 2014.

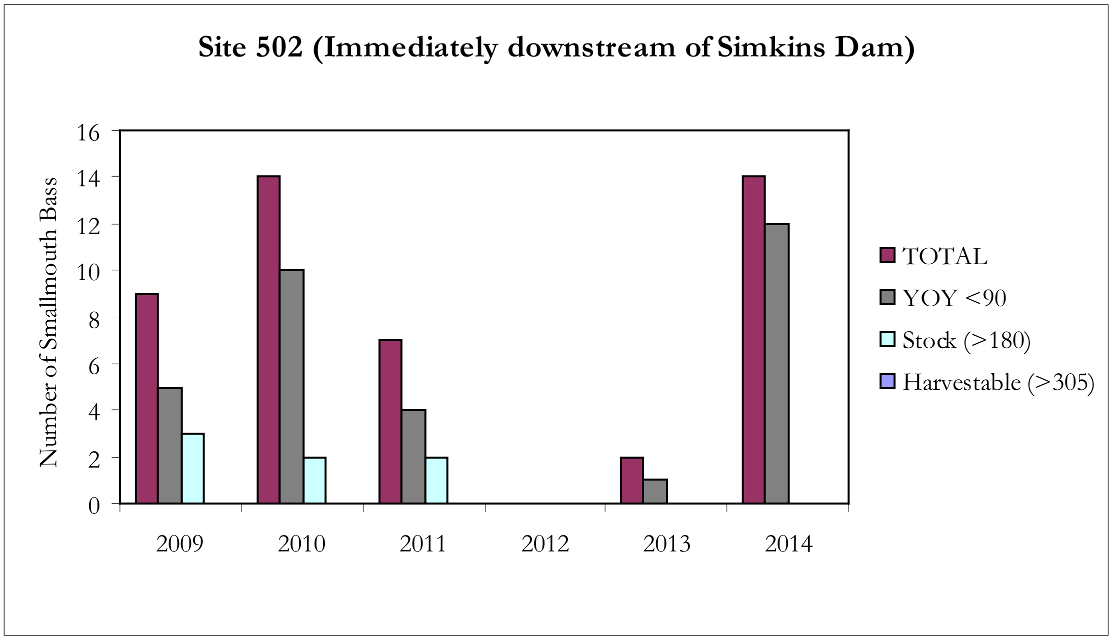

Figure 7.12: Smallmouth Bass total abundance and abundance per size class at site 502, located immediately downstream of Simkins Dam, from 2009-2014. No bass within the Harvestable size class were collected during the study.

Discussion

Fish Assemblage Response

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. The dam removal process causes immediate geomorphologic and hydrologic changes, especially in areas adjacent to the dam. These changes can elicit subsequent responses in fish assemblages in downstream and upstream reaches.

As was first documented by Harbold et al. (2013), the most pronounced changes in fish assemblages observed during our study occurred in the river downstream of Simkins Dam. We noted declines in fish density, biomass, and changes in the ecological composition of assemblages at each downstream treatment site as fish habitat quality and quantity declined due to the influx of sand and sediment previously stored behind the dam. Downstream sedimentation had the greatest affect on benthic riffle fishes that require clean, coarse substrate as refuge and for feeding (Jenkins and Burkhead 1994; Bushaw-Newton et al. 2002). Sedimentation of downstream areas also negatively affected intolerant species – those sensitive to disturbance – and lithophilic spawning species – those that require coarse substrate free of fine sediment for successful reproduction. We also documented changes in Smallmouth Bass populations, especially in the young-of year size class, in reaches downstream of the former Simkins Dam site following its removal. Declines observed in young-of year bass likely reflected reduced survival of this susceptible age class due to stress associated with sedimentation (Doeg and Koehn 1994; Waters 1995). Changes in bass populations and in the ecological composition of assemblages (e.g., benthic riffle species, lithophilic spawners) may reflect direct mortality or displacement of these species from the affected portion of the river, or a combination thereof.

Changes in fish assemblages did not always occur simultaneously (i.e., in the same year) at both downstream treatment sites. We noted changes in fish density, number of tolerant species, and smallmouth bass populations at Site 502, located closest to Simkins Dam, a full year before similar changes were observed at Site 501. Similarly, fish assemblages at sites 502 and 501 were most dissimilar from pre-removal baseline in 2012 and 2013, respectively. The lag in response observed at Site 501 is likely due to both the greater distance between this site and Simkins Dam and the influence of Bloede Dam, located between both treatment sites, on the downstream movement of fine sediments. Although the timing of the response to dam removal was not always the same between treatment sites, the greatest impact attributable to dam removal occurred at both sites in 2012 – two years following dam removal. For example, the largest changes in a number of metrics examined including fish density, biomass, and benthic riffle species occurred in 2012 at both downstream treatment sites.

Dam removal can alter fish species richness and composition, and increase abundance of non-game and game fishes in formerly impounded stream reaches as they return to free-flowing conditions (Kanehl et al. 1997; Catalano et al. 2007; Maloney et al. 2008). We documented an increase in fish density and the numbers of lithophilic spawning and intolerant species at the upstream treatment site (504) following dam removal. We also documented a temporary decrease in the density of non-native fishes at this site in the first two years (2011-2012) following the removal of Simkins Dam; a pattern consistent with other dam removal studies (Kanehl et al. 1997; Bushaw-Newton et al. 2002). Previous studies have documented dramatic shifts from assemblages dominated by lentic species to those dominated by lotic species in formerly impounded reaches (Maloney et al. 2008). We did not detect such a shift in the fish assemblage at the upstream treatment site, but this was most likely due to the location of Site 504. This site 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.

Fish Assemblage Recovery

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 have been shown to recover within one year following dam removal (Catalano et al. 2007). However, recovery of some taxa can take several years (Maloney et al. 2008). Monitoring over the six year study period, including four years following dam removal, allowed us to assess the response in fish assemblages and also to examine for evidence of recovery as habitat conditions in the river improved.

Site 502, located closest to Simkins Dam, was the first of the downstream treatment sites to be impacted by sedimentation associated with dam removal. This is reflected in the initial response measured in the fish assemblage at this site within the first two years following dam removal. Given its proximity to Simkins Dam, this site was also the first to recover. Extensive sedimentation that occurred at this site following dam removal largely abated in 2013 as most fine sediment initially deposited was flushed further downstream. As substrate composition and habitat quality returned to pre-removal conditions, many of the fish assemblage metrics examined in this study showed signs of recovery. As of 2014, fish assemblage species composition and fish density have recovered to pre-removal levels measured from 2009-2010. Benthic riffle species at this site have nearly recovered to pre-removal levels. Similarly, Smallmouth Bass abundance and young-of-year have also recovered, however size structure of the bass population (specifically, stock-sized bass) at this site remains altered. Fish biomass at this site also remains lower than that measured prior to the removal of Simkins Dam.

As mentioned previously, there was a lag time in the response to sedimentation associated with dam removal in the fish assemblage at Site 501 due to the combined influence of distance (from Simkins Dam) and the attenuating effects on sediment transport of Bloede Dam. Recovery at this site has also been slower than that measured at Site 502. As of 2014, fish density and biomass remain below pre-removal levels. The number of benthic riffle species, lithophilic spawners, and intolerant species remain below pre-removal levels as well, but show signs of recovery – approaching pre-removal levels in 2013 and 2014. Recovery of the fish assemblage at this site may be slowed due to the continued presence of Bloede Dam. As a fish blockage, this dam has altered hydrologic patterns in the river and fragmented fish habitat and populations (Pringle 1997; Gardner et al. 2011). Bloede Dam likely limits the potential for downstream re-colonization of fish species (e.g., benthic riffle species, lithophilic spawners) most affected by the removal of Simkins Dam thereby hindering recovery of the fish assemblage at this site.

The majority of studies conducted to date that have examined fish assemblage response and recovery to dam removal have been conducted over short time periods (<3 years) with minimal post-removal monitoring (Maloney et al. 2008). To our knowledge, our study is one of the only dam removal studies that have monitored both response and recovery over six years. Monitoring of the five Patapsco River sites discussed in this chapter as planned for 2015 and beyond will continue to provide important information on the short-term impacts and long-term benefits of dam removal on riverine fishes in the Patapsco River.

Literature Cited

Bednarek, A.T. 2001. Undamming rivers: A review of the ecological impacts of dam removal. Environmental Management 27:8032-814.

Bushaw-Newton, K.L., D.D. Hart, J.E. Pizzuto, J.R. Thomson, J. Egan, J.T. Ashley, T.E. Johnson, R.J. Horwitz, M. Keeley, J. Lawrence, D. Charles, C. Gatenby, D.A. Kreeger, T. Nightengale, R.L. Thomas, and D.J. Velinsky. 2002. An integrative approach towards understanding ecological responses to dam removal: The Manatawny Creek study. Journal of the American Water Resources Association 38:1581-1599.

Catalano, M.J., M.A. Bozek, and T.D. Pellett. 2007. Effects of dam removal on fish assemblage structure and spatial distributions in the Baraboo River, Wisconsin. North American Journal of Fisheries Management 27:519-530.

Doeg, T.J., and J.D. Koehn. 1994. Effects of draining and desilting a small weir on downstream fish and macroinvertebrates. Regulated Rivers: Research and Management 9:263-277.

Doyle, M.W., E.H. Stanley, and J.M. Harbor. 2003. Channel adjustments following two dam removals in Wisconsin. Water Resources Research 39:1-15.

Doyle, M.W., E.H. Stanley, C.H. Orr, A.R. Selle, S.A. Sethi, and J.M. Harbor. 2005. Stream ecosystem response to small dam removal: Lessons from the heartland. Geomorphology 71:227-244.

Gabelhouse, D. W., Jr . 1984. A length-categorization system to assess fish stocks. North American Journal of Fisheries Management 4:273-285.

Gardner, C., S.M. Coghland, Jr., J. Zydlewski, and R. Saunders. 2011. Distribution and abundance of stream fishes in relation to barriers: Implications for monitoring stream recovery after barrier removal. River Research and Applications.

Harbold, W., S. Stranko, J.V. Kilian, M. Ashton, and P. Graves. 2013. Patapsco River dam removal study: Assessing changes in American Eel distribution and aquatic communities. Final Report. Prepared for: American Rivers under NOAA American Recovery and Reinvestment Act award NA09NMF4630327.

Jenkins, R.E. and N.M. Burkhead. 1994. Freshwater fishes of Virginia. American Fisheries Society, Bethesda, Maryland.

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

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27.

Karr, J.R. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey Special Publication 5:1-28.

Krebs C.J. 1989. *Ecological Methodology*. Addison-Welsey Educational Publishers, Inc. New York.

Maloney, K.O., H.R. Dodd, S.E. Butler, and D.H. Wahl. 2008. Changes in macroinvertebrate and fish assemblages in a medium-sized river following a breach of a low-head dam. Freshwater Biology 53:1055-1068.

Pringle, C.M. 1997. Exploring how disturbance is transmitted upstream: going against the flow. Journal of the North American Benthological Society 16:425-438.

Southerland, M.T., G.M. Rogers, M.J. Kline, R.P. Morgan, D.M. Boward, P.F. Kazyak, R.J. Klauda, and S.A. Stranko. 2007. Improving biological indicators to better assess the condition of streams. Ecological Indicators 7:751-767.

Waters, T.F. 1995. Sediment in streams: Sources, biological effects, and control. American Fisheries Society Monograph 7.

Chapter 8 : Results of the 2013 and 2014 Mainstem Patapsco River Macroinvertebrate Sampling

Introduction

This report summarizes the results of the latest benthic macroinvertebrate sampling in the Patapsco River conducted since 2012. These 2013 and 2014 monitoring results were compared to the pre-dam removal (2009-2010) and the first two years' post-dam removal (2011- 2012) results to determine if the Patapsco River macroinvertebrate communities had continued to change in the years since the removal of Simkins Dam. Macroinvertebrate data collected at sites impacted by dam removal are compared to data from a control site outside the impact area. For an observed change in the macroinvertebrate community at a given site to be attributed to dam removal impacts, observed changes would need to exceed the range of any change observed at the control site during the same time period.

Methods

During the 2013-2014 monitoring period we followed the same methods for data collection and analysis outlined in Harbold et al. 2013 with three exceptions: the graphical representation of figures changed in order to allow for better interpretation of the data, two sampling sites were abandoned, and the number of metrics examined was reduced.

Figures depicting changes in macroinvertebrate metrics in this report contain the values recorded at the upstream control site as a series of horizontal lines. The average value at the control site over the post-dam removal period (2011-2014) is displayed as a solid red line bordered above and below by dashed black lines representing that average plus and minus one standard error. These lines overlay bars representing the year to year values for impact sites, allowing for easier determination of whether or not values recorded at impact sites are within the variability observed at the control site.

Fourteen sites, including one control site, were sampled in both 2013 and 2014. Sampling at 507 and 508 - two sites in the vicinity of Union Dam that were sampled from 2009-2012 - was discontinued after 2012. The locations of the sites used in this study, as well as the Patapsco River dams (both extant and removed) are shown in Figure 8.1 below.

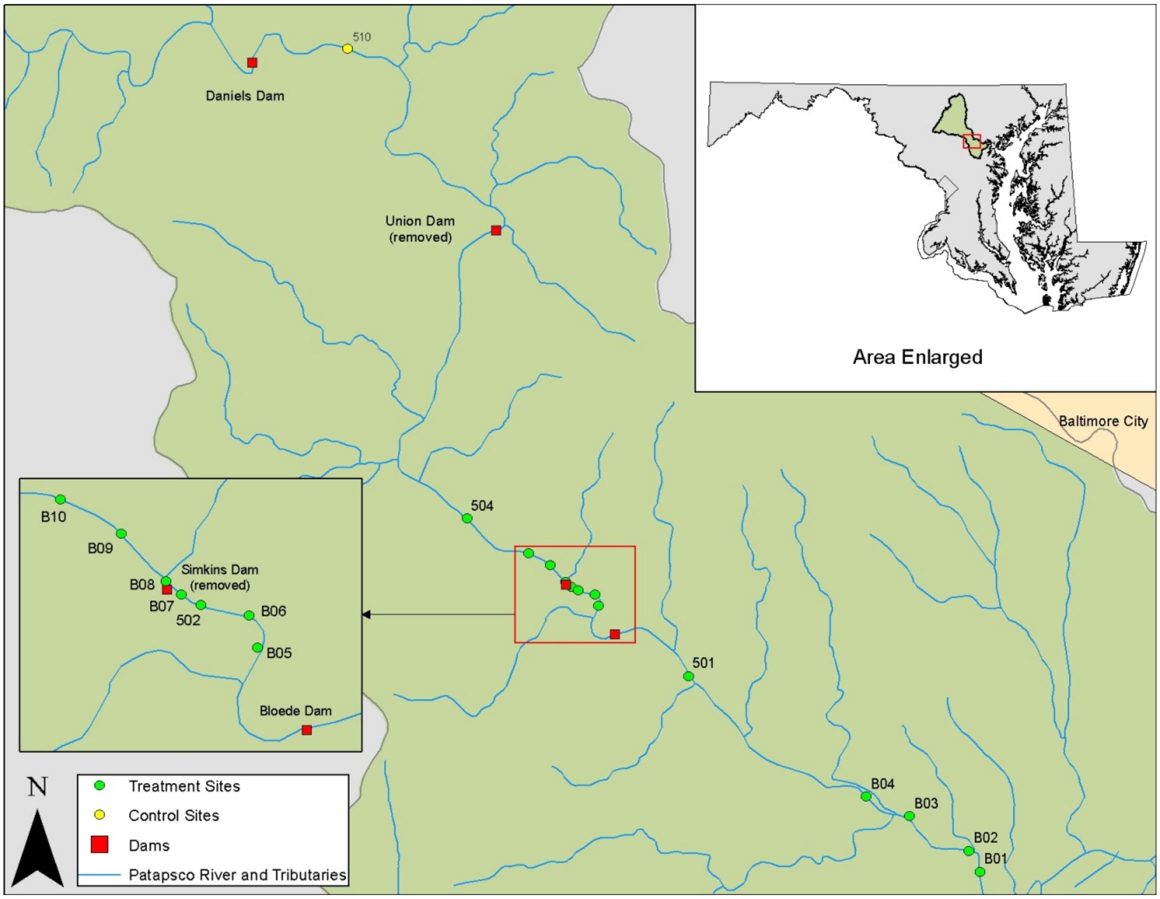

Figure 8.1: Sites sampled for benthic macroinvertebrates on the main stem Patapsco River in 2013 and 2014.

While nine metrics were examined in the last report, just two- %Burrower and %EPT- were found to be sufficient for this latest effort to document the changes taking place in the Patapsco River. Changes to the benthic macroinvertebrate community in the Patapsco River are primarily the result of habitat change, and these two metrics effectively illustrate the communities' response to habitat. Explanations of these metrics, first reported in Harbold et al. 2013, are included below:

%Burrower

This metric refers to the proportion of a macroinvertebrate sample that is comprised of burrowing taxa. 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.

%EPT

This metric refers to the proportion of a macroinvertebrate sample that is comprised of EPT taxa. 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 taxa are a commonly used metric 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.

Results

Changes in the benthic macroinvertebrate community appear to be driven by changes in habitat, particularly in the vicinity of Simkins Dam. The changes in the types of habitat sampled, as well as changes in two macroinvertebrate taxa metrics- %Burrower and %EPT - do a particularly good job of illustrating this observation. These are discussed in detail below.

Change in Habitat Types Before and After the Removal of Simkins Dam

With the removal of Simkins Dam, the habitat available for benthic macroinvertebrates changed significantly. This is exemplified by the changing conditions at two sites, B08 and B09 (Figure 8.1). Following dam removal, the habitat at B08 and B09 changed from a deep pool/glide system with sandy substrate to a more shallow system with exposed rock substrate. This change can be shown by comparing the breakdown of the habitats sampled during the pre-removal (2009-2010) and post removal (2011-2014) time periods (Table 8-1). The habitat sampled at B08 and B09 changed from a mixture of macrophytes, sand/gravel, and rootwad/woody debris before Simkins was removed to 95% riffle after the removal. In contrast, sites downstream of the dam, which were composed primarily of coarse, heterogeneous substrate prior to dam removal, temporarily shifted to finer substrates as impounded sediment moved downstream through the study area. These changes in habitat sampled are reflected in the observed changes in the macroinvertebrate metrics during the same time period. By and large, the changes observed in 2013- 2014 remain consistent with those observed when comparing the changes 2011-2012 to the pre-removal period.

Table 8-1:Habitat proportions sampled during the pre- (2009-2010) and post-dam removal (2013-2014) periods at the control site, 510, and two impacts sites, B08 and B09, immediately upstream of Simkins Dam.

Site		Riffle (%)	Leaf pack (%)	Rootwad/Woody debris (%)	Macrophytes (%)	Other: sand (%)
Impact	B08 Pre	0	0	15	50	35
	B08 Post	95	2.5	2.5	0	0
	BO9 Pre	0	0	17.5	50	32.5
	B09 Post	97.5	1.25	1.25	0	0
Control	510 Pre	92.5	0	0	7.5	0
	510 Post	90	0	10	0	0

Change in the Macroinvertebrate Community

%Burrower

2013 %Burrower values were the lowest recorded during this study at all sites except for B03 (Figure 8.2). The largest decrease from pre-removal levels in a %Burrower score observed during this study (-37) was recorded at B08 during 2013. The primary reason for the decreased 2013 %Burrower values is the lack of burrower individuals, primarily burrowing chironomids, in the 2013 collection (366) compared to the Pre, 2011, 2012, and 2014 collections which all contained 788 or more individuals.

2014 % Burrower values were more similar to the baseline Pre %Burrower than any other sampling period. Similar to observations from 2011- 2013, the largest decrease in %Burrower during 2014 occurred at B08, and, as was the case with all post-removal periods, was due to the absence of the aquatic clam *Corbicula* in the samples.

With the exception of B03, the %Burrower increase we observed in 2011 and 2012 at sites downstream of Simkins Dam was not observed in the 2013 and 2014 data. At all sites downstream of the dam, with the exception of B03, %Burrower values were either close to or less than pre-removal values, suggesting that conditions leading to increases observed in 2011-2012 may have abated.

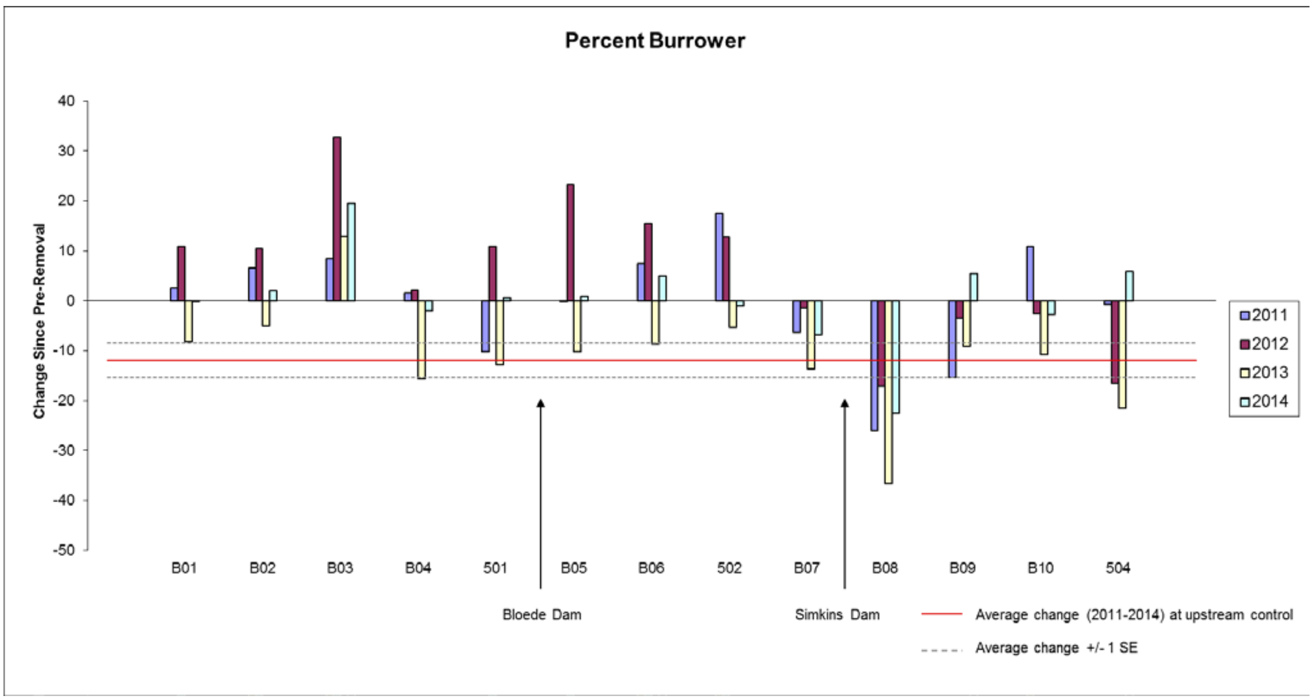

Figure 8.2: Change since pre-removal levels in %Burrower at Patapsco River sites during the time periods 2011-2014.

%EPT

%EPT values (Figure 8.3) were almost universally higher than pre-removal values throughout the study area during 2013 and 2014. The greatest increases were primarily at sites upstream of Simkins Dam, while the one site with a %EPT value less than pre-removal levels (B04) was downstream. All other instances of %EPT values decreasing after dam removal (values less than pre-removal values) were also downstream of Simkins Dam, during 2011 and 2012.

The highest %EPT values of the study so far were observed in 2013. Ten out of fourteen sites saw their highest recorded %EPT values during this year, including three out of the four sites upstream of Simkins Dam. These results are due to an unexplained irruption of stonefly individuals in the 2013 samples. 2014 %EPT values were still higher than pre-removal levels but lower than the 2013 levels. This decrease is likely due to a return of stonefly densities to more normal levels following the spike in 2013.

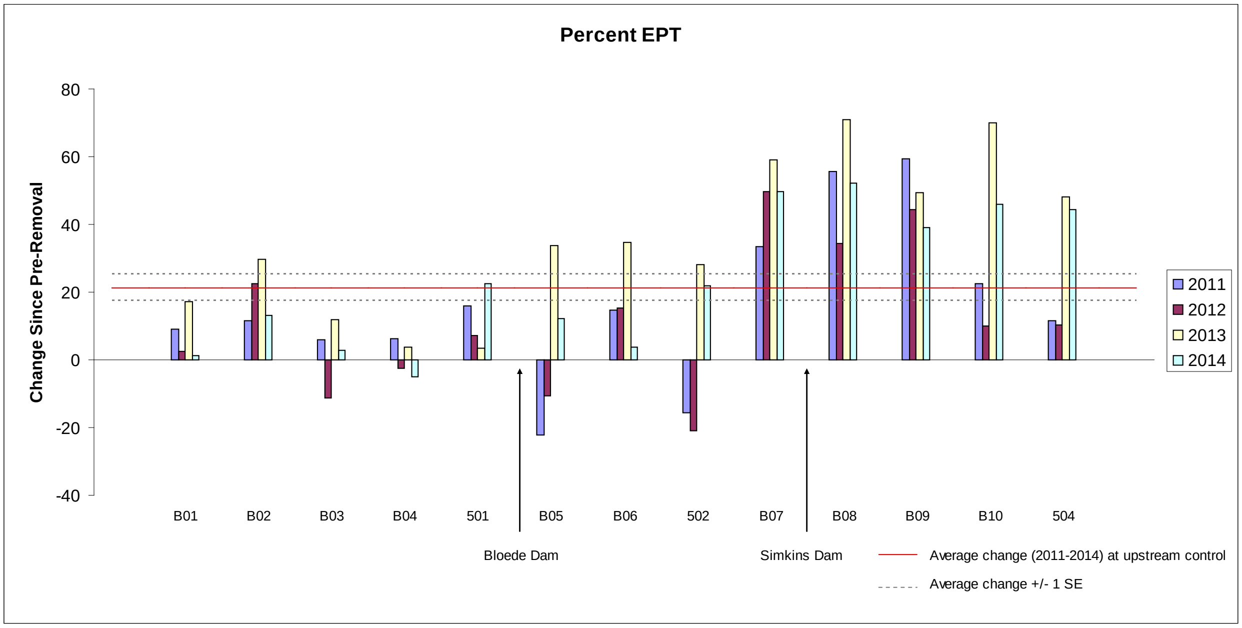

Figure 8.3: Change since pre-removal levels in %EPT at Patapsco River sites during the time periods 2011-2014

Discussion

Throughout the duration of this study period (2009-2014), the most notable changes in both macroinvertebrate community metrics have been recorded at sites closest to Simkins Dam (B06 upstream to B10, Figure 8.1). The sites where these changes have been observed are those at which the dominant habitat and substrate types shifted after the removal of Simkins Dam.

The removal of Simkins Dam changed the habitat and substrate types that were available for macroinvertebrate colonization. At sites upstream of Simkins Dam there was a change from slow-flowing, sand-dominated substrate to faster-flowing, heterogeneous rock substrate. The macroinvertebrate taxa followed suit, as burrowing taxa declined while EPT taxa became more abundant. Conversely**,** the habitat and substrate downstream of Simkins temporarily shifted from faster-flowing, heterogeneous rock substrate to slow-flowing, sand-dominated substrate following dam removal. Again, biological changes followed physical ones, with burrowing taxa increasing while EPT taxa declined.

The influx of EPT taxa into sites upstream of Simkins Dam following its removal, coupled with the change in habitat proportions sampled due to newly created conditions (Table 8-1, for example), led to the significant decreases of %Burrower taxa (Figure 8.2) that had formerly dominated in the impounded sites but were not as well suited for the new conditions as the EPT taxa groups. At sites immediately downstream of the dam, the results were, for the most part, reversed. It is in these areas that the largest decreases in %EPT (Figure 8.3) and the other EPT-influenced metrics were observed.

Despite the change in habitat downstream of the dam, the changes in the downstream macroinvertebrate communities were not as pronounced as those upstream. These results can be attributed to taxa, particularly the EPT, that are adapted to faster-flowing waters and heterogeneous rock substrates being more mobile and better able to recolonize habitats, especially after a large disturbance. Furthermore, the downstream sites were faster-flowing, heterogeneous rock substrate prior to the removal of Simkins, and after the initial pulse of sediment following the removal of Simkins, were able to return back to these conditions as the fine sediments were pushed downstream over time. Upstream sites, however, experienced a more dramatic shift to fast-flowing, heterogeneous substrate that has persisted to date.

Beyond the habitat-driven changes in macroinvertebrate taxa, the discovery in 2013 of a new stonefly record for the Patapsco River bears mentioning. One *Pteronarcys* individual identified as *P. dorsata* was collected at Site 511 during 2013. This individual represents the first MBSS record of this genus in the Patapsco River mainstem. *Pteronarcys*, in the family Pteronarcyidae, is the largest stonefly in MD, is pollution intolerant (MBSS tolerance value of 1.1, Southerland et al. 2005), and is typically found in streams and rivers colder than the Patapsco. The presence of *Pteronarcys* generally indicates the system in which they are found has had a period of good water quality, at least for the time period *Pteronarcys* has been present, as these individuals require high water quality during their 1-2 year development.

Future Macroinvertebrate Sampling in the Patapsco River

In 2013, an annual quantitative benthic macroinvertebrate sampling protocol was implemented at six Patapsco River mainstem sites in an effort to observe changes that could occur in the macroinvertebrate community as a result of the removal of Bloede Dam. Of these six sites (see Figure 8.4), two are immediately downstream of Bloede (501 and 516), two are immediately upstream of Bloede (515 and B05), and two serve as upstream controls (510 and 511). These sites will be sampled annually in September both before and after the anticipated removal of Bloede Dam.

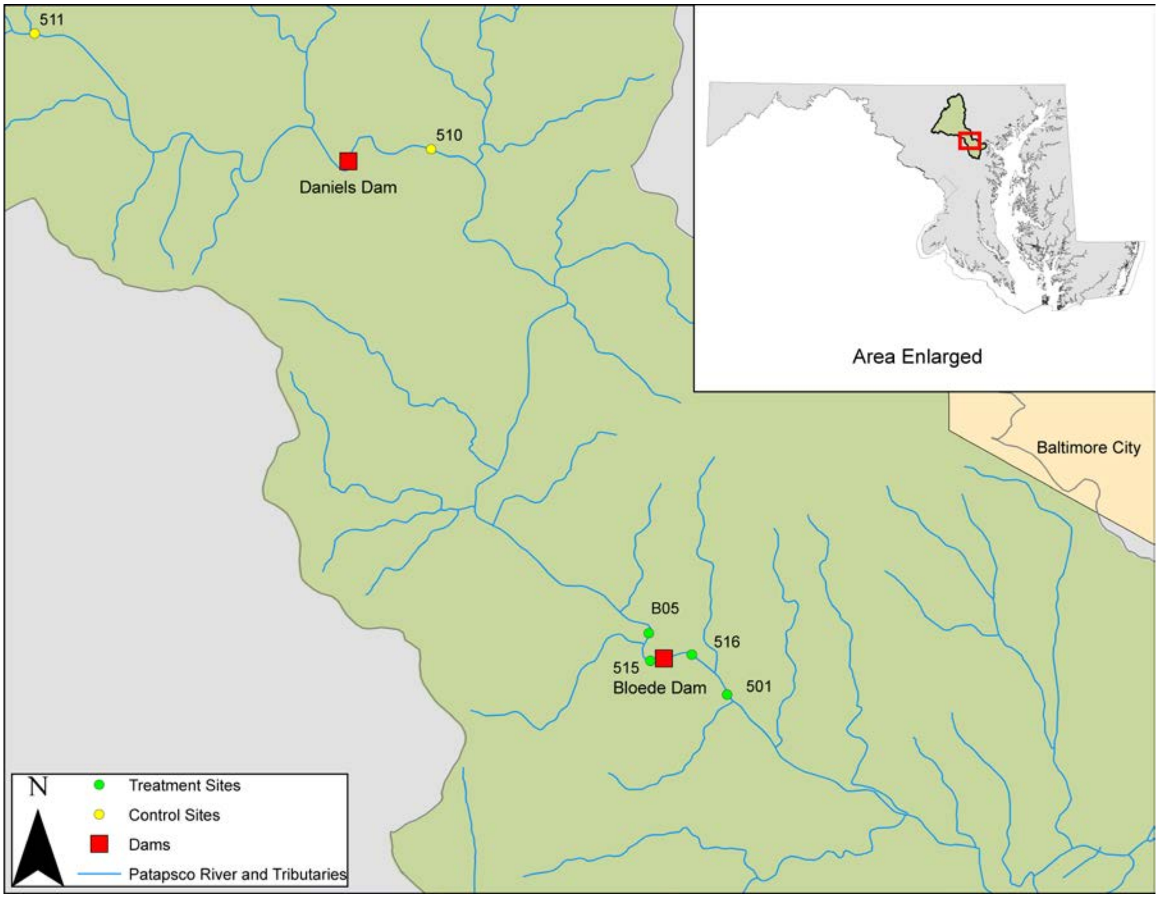

Figure 8.4: Sites sampled quantitatively for benthic macroinvertebrates and physical habitat in the vicinity of Bloede Dam since 2013.

At each site, five 15 meter transects are marked with a flag at mid-channel, creating two grids per transect and 10 grids overall at each site. In each grid, one Hess sample (area sampled = 0.086 m^2) is taken in an area that is representative of the average flow, average depth, dominant habitat (i.e., riffle, glide, etc.) and substrate type in the grid. This results in a total of ten samples (0.086 m^2 x 10 = 0.86 m^2) per site. Every organism collected in a sample is sorted in the lab and identified to genus level or to the lowest taxonomic level possible.

In addition to a macroinvertebrate sample, select physical habitat data are collected at each site. Prior to sampling, the wetted width of each grid transect is recorded. Additionally, stream velocity is measured with a flow meter immediately upstream of where the Hess sample is collected. Embeddedness is estimated as light, moderate or heavy in each grid sampled and both the dominant and subdominant substrate types are recorded.

This quantitative analysis of macroinvertebrate community data will be coupled with the substrate and habitat data recorded at the location of each sample to better understand the relationships, if any, that exist between the two. This will be especially helpful in the event that Bloede Dam is removed, as there will be both pre- and post-removal data (assuming funding continues) quantitatively linking the response of the macroinvertebrate community to any changes in the physical habitat around the dam that may occur.

The spring and summer macroinvertebrate D-net sampling protocol that has been employed since 2009 will continue at 17 mainstem sites and 6 tributary sites. These data will continue to be analyzed and compared to data collected from previous years to provide a better understanding of changes occurring in the macroinvertebrate community at these sites.

Literature Cited

Harbold, W., S. Stranko, J. Kilian, M. Ashton, P. Graves. 2013. Patapsco River Dam Removal Study: Assessing Changes in American Eel Distribution and Aquatic Communities. Report submitted to American Rivers by the Maryland Department of Natural Resources.

Merritt, R.W., K.W. Cummins, and M.B. Berg (eds.). 2008. An Introduction to the Aquatic Insects of North America (4th edition). Kendall/Hunt Publishing Company,Dubuque, Iowa. 1214 pp.

Southerland, M.T., Rogers, G.M., Kline, M.J., Morgan, R.P., Boward, D.M., Kayzak, P.F., Klauda, R.J., and S.A. Stranko. 2005. New biological indicators to better assess the condition of Maryland streams. Report prepared for Maryland Department of Natural Resources, Monitoring and Non-tidal Assessment Division, Annapolis, MD.

Chapter 9 : Conclusions and Recommendations

The conclusions based on our work in the Patapsco River come from four 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 as long as possible to document the long-term ecological response to dam removals on the Patapsco River. Restoration on the Patapsco River is not complete- Bloede Dam will likely be removed within the next one to two years, before the major ecological changes in the Patapsco River resulting from the removal of Simkins Dam have been able to reach a new dynamic equilibrium. Once Bloede Dam has been removed it will not be possible to tease the impacts of its removal from those of Simkins Dam before it. Analyses of both will then need to be linked together, treating both removals as one restoration process. Documenting changes as they occur is the best way to demonstrate the benefits of dam removal, and continued monitoring will allow this to occur. 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 in future years.

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 and the one preceding it will provide six 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 should 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.