An Ecological Study of Gunston Cove

2016

FINAL REPORT
August 2017

by

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Back of Title Page

Table of Contents

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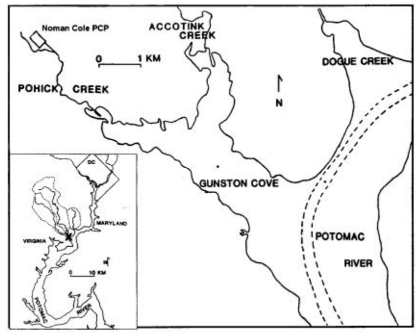
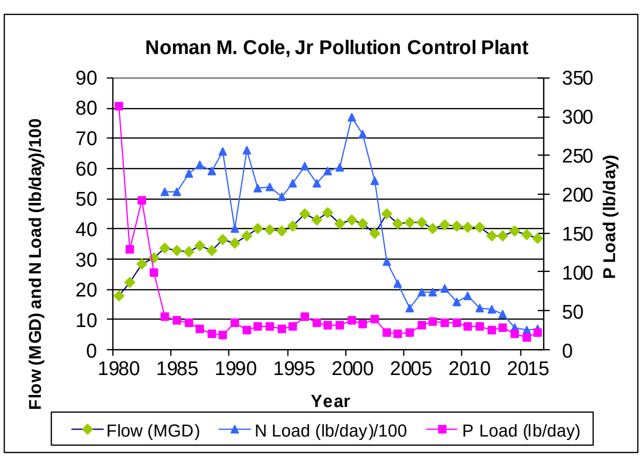
Executive Summary



Gunston Cove is an embayment of the tidal freshwater Potomac River located in Fairfax County, Virginia about 12 miles (20 km) downstream of the I-95/I-495 Woodrow Wilson Bridge. The Cove receives treated wastewater from the Noman M. Cole, Jr. Pollution Control Plant and inflow from Pohick and Accotink Creeks which drain much of central and southern Fairfax County. The Cove is bordered on the north by Fort Belvoir and on the south by the Mason Neck. Due to its tidal nature and shallowness, the Cove does not seasonally stratify vertically, and its water mixes gradually with the adjacent tidal Potomac River mainstem. Thermal stratification can make nutrient management more difficult, since it can lead to seasonal oxygen-diminished bottom waters that may result in fish mortality. Since 1984 George Mason University personnel, with funding and assistance from the Wastewater Management Program of Fairfax County, have been monitoring water quality and biological communities in the Gunston Cove area including stations in the Cove itself and the adjacent River mainstem. This document presents study findings from 2016 in the context of the entire data record.

The Chesapeake Bay, of which the tidal Potomac River is a major subestuary, is the largest and most productive coastal system in the United States. The use of the bay as a fisheries and recreational resource has been threatened by overenrichment with nutrients which can cause nuisance algal blooms, hypoxia in stratified areas, and a decline of fisheries. As a major discharger of treated wastewater into the tidal Potomac River, particularly Gunston Cove, Fairfax County has been proactive in decreasing nutrient loading since the late 1970's. Due to the strong management efforts of the County and the robust monitoring program, Gunston Cove has proven an extremely valuable case study in eutrophication recovery for the bay region and even internationally. The onset of larger areas of SAV coverage in Gunston Cove will have further effects on the biological resources and water quality of this part of the tidal Potomac River.

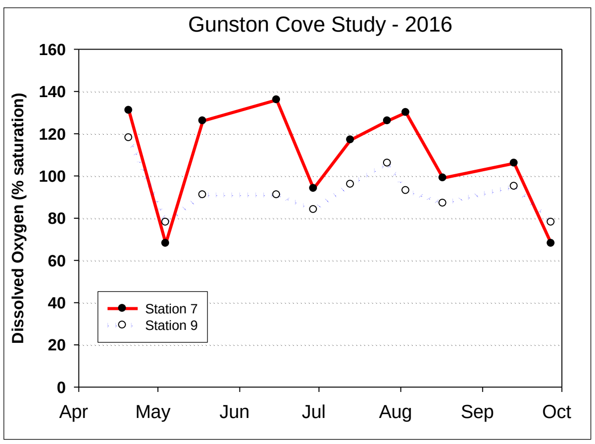


As shown in the figure to the left, phosphorus loadings were dramatically reduced in the early 1980's. In the last several years, nitrogen, and solids loadings as well as effluent chlorine concentrations have also been greatly reduced or eliminated. These reductions have been achieved even as flow through the plant has slowly increased.

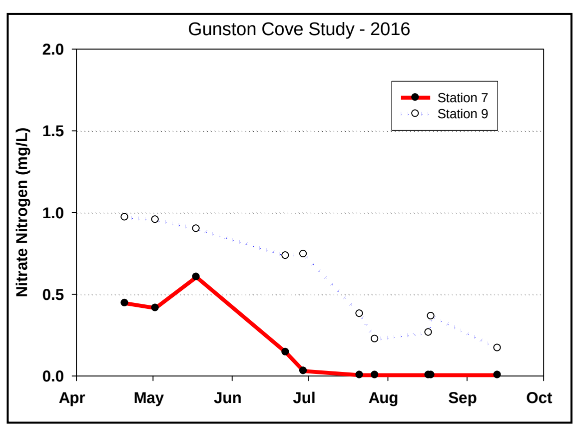
The ongoing ecological study reported here provides documentation of major improvements in water quality and biological resources which can be attributed to those efforts. Water quality improvements have been substantial in spite of the increasing population and volume of wastewater produced. The 30 plus year record of data from Gunston Cove and the nearby Potomac River has revealed many important long-term trends that validate the effectiveness of County initiatives to improve treatment and will aid in the continued management and improvement of the watershed and point source inputs.

The year 2016 was characterized by well above normal temperatures for the summer months. Monthly precipitation was well above normal in May, but close to normal in other months. Two sampling dates (early May and late June) occurred following significant flow events.

Mean water temperature was similar at the two stations reaching a maximum over 30°C in late July. Specific conductance declined substantially at both stations in the wake of the early May flow events, then gradually increased through the remainder of the year.

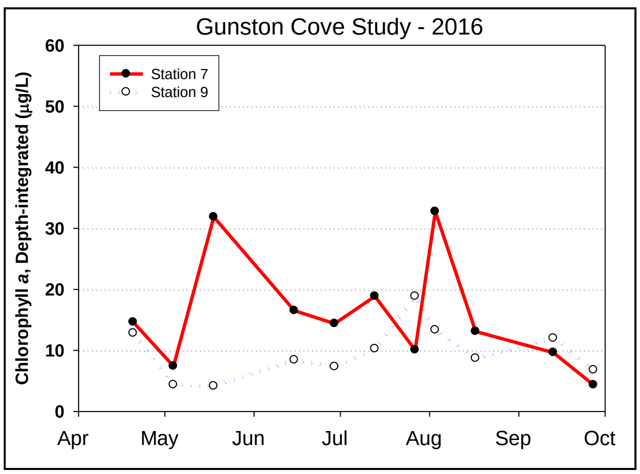


Chloride showed a similar pattern, but was consistently somewhat higher in the cove. Dissolved oxygen saturation (DO) was normally substantially higher in the cove than in the river due to photosynthetic activity of phytoplankton and submersed aquatic vegetation (SAV) (figure at left). An exception to this occurred in the wake of the early May flow event when both areas showed a depression in DO which was very marked in the cove. A second, lesser decline in late June was in the wake of a second flow event. Field pH patterns mirrored those in DO: higher values in the cove than the river and strong response to the early May flow event. Total alkalinity was generally higher in the river than in the cove and was fairly constant seasonally. Secchi disk transparency was generally lower in the cove in spring and showed a depression in the early May sampling as well as the late June sampling. By late summer Secchi disk transparency in the cove increased above river values and approached 1.8 m by late September. Light attenuation coefficient and turbidity followed a similar pattern.



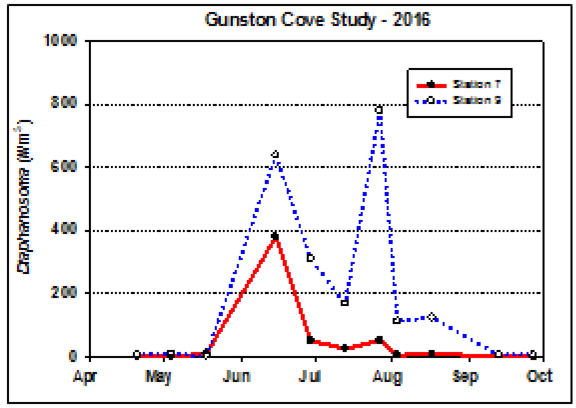
Ammonia nitrogen was consistently low in the study area during 2016. All but one value was below the limits of detection which makes analyzing any temporal or spatial trends impossible. Un-ionized ammonia remained below values that would cause toxicity issues, but exact values were not possible due to the high incidence of non-detects on total ammonia. Nitrate values declined seasonally at both sites due to algal and plant uptake and possibly denitrification. By late July nitrate nitrogen in the cove was below detection limits where it remained through the remainder of the year (see figure above). River nitrate nitrogen levels reached a low of about 0.2 mg/L. Organic nitrogen exhibited substantial variability with a decline in values in the cove through the course of the year. Total phosphorus was similar at both sites and showed little seasonal change. Soluble reactive phosphorus was very low and consistently below detection limits in the cove and higher in the river. N to P ratio declined strongly at both stations reaching a minimum of about 12 in September which is still indicative of P limitation of phytoplankton and SAV. Biochemical oxygen demand (BOD) was generally higher in the cove than in the river. Total suspended solids (TSS) was fairly constant throughout the year. Peak value in the river was observed in late June; interestingly, the early May flow event did not seem to affect TSS. Volatile suspended solids (VSS) was also fairly constant between sites and seasonally.

In the cove algal populations as measured by chlorophyll *a* declined strongly in the wake of the early May flow event. A strong rebound was observed in late May followed by a gradual decline and another peak in early August. In the river, the early May decline was observed, but levels recovered only gradually reaching a late July peak. Both cell density and biovolume indicated the flow-induced decline in phytoplankton in early May in the cove and in the river. Values in the cove also showed a decline in late June, the time of the second flow event. The early August peak in the cove in chlorophyll was not seen in the cell count data. Due to a cutback in phytoplankton count frequency, cell counts were not done on the late July sample from the river. Cell density data from the cove was dominated by cyanobacteria, the principal species being *Oscillatoria*. In the river, diatoms dominated cell density data for most of the year, first Pennate 2, then Pennate 1, and finally *Melosira*. In late summer other groups were important with *Anabaena* numerous in August and *Dictyospherium* important in September. Cell biovolume was more evenly distributed among various taxa in the cove than was cell density. In the river cell biovolume was dominated by diatoms with discoid centrics most important in the first half of the year and *Melosira* in the second half. Rotifers continued to be the most numerous zooplankton in 2016.



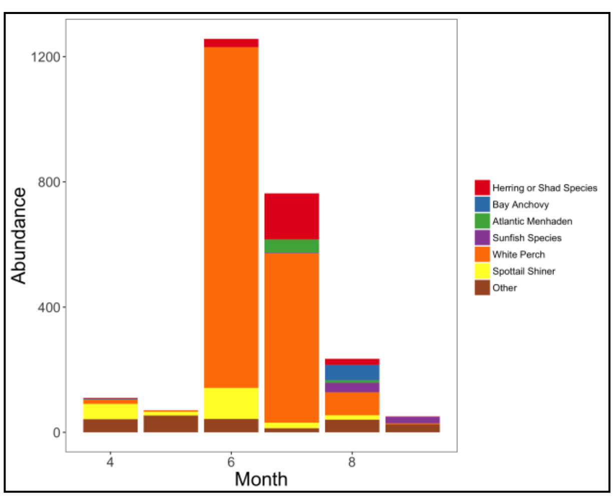
Rotifer densities were unusually high in April in both areas, but declined dramatically in early May in response to the flow event. Another peak was observed in late June. *Brachionus*, *Filinia*, and *Keratells* shared dominance in the cove; *Filinia* was not common in the river, but *Brachionus* and *Keratella* were. *Bosmina*, a small cladoceran that was often common was only present at low densities in 2016. *Diaphanosoma*, a larger cladoceran was found in both area at moderate densities. Both peaked in late June and then declined after in the wake of the flow event. A subsequent higher peak in

Diaphanosoma in the river was not found in the cove (see graph below). Surprisingly, *Daphnia* and *Ceriodaphnia* exhibited their one strong peak in the cove in early May. *Moina* was only found in substantial number is late June in the river. *Leptodora* also seemed to respond positively to the early May flow event in the cove and reached even higher levels in early June in both areas. Copepod nauplii densities reached a peak in both study areas in early June and then declined. A second peak was found in the river in late August. The calanoid copepod *Eurytemora* was very abundant in the cove in early May whereas the river maximum was found in early June. A second calanoid *Diaptomus* was restricted to the river at lower levels. Cyclopoid copepods had a strong maximum in early May in the cove and a mid-July maximum in the river.



In 2016 ichthyoplankton was dominated by clupeids, most of which were Gizzard Shad and Alewife, and to a lesser extent, Blueback Herring, American Shad, and Hickory Shad. White Perch was a dominant species as well, with the same relative contribution to the total ichthyoplankton community as Gizzard Shad. Striped Bass and Inland Silverside was found in relatively high densities as well. *Morone* species (White Perch and Striped Bass) were mostly found in the Potomac mainstem, confirming their affinity for open water. Other taxa were found in very low densities similar to the previous year. The highest density of fish larvae occurred in mid-May, which was driven by a high density of Clupeid larvae.

A total of 2484 fishes comprising 24 species were collected in all trawl samples combined (see Figure below). The dominant species of the fish collected in the trawls was White Perch (69.4%, numerically). In the spring, adult White Perch were primarily caught in the nets while later in the summer juveniles dominated. Other abundant taxa included herring or shad (7.9%), Spottail Shiner (7.7%), Sunfishes (2.2%), and Bay Anchovy (2.1%). Other species were observed sporadically and at low abundances. A total of 35 seine samples were conducted, comprising 3885 fishes of 26 species. This is a little lower than the number of individuals and species collected last year. Similar to last year, the most dominant species in seine catches was Banded Killifish, with a relative contribution to the catch of 56.4%. Other dominant species (with >5% of relative abundance) were White Perch (10.2%) followed by Inland Silverside (7.1%), eastern Silvery Minnow (6.2%), and Alosa sp. (5.2%). In 2016 we collected a total number of 456



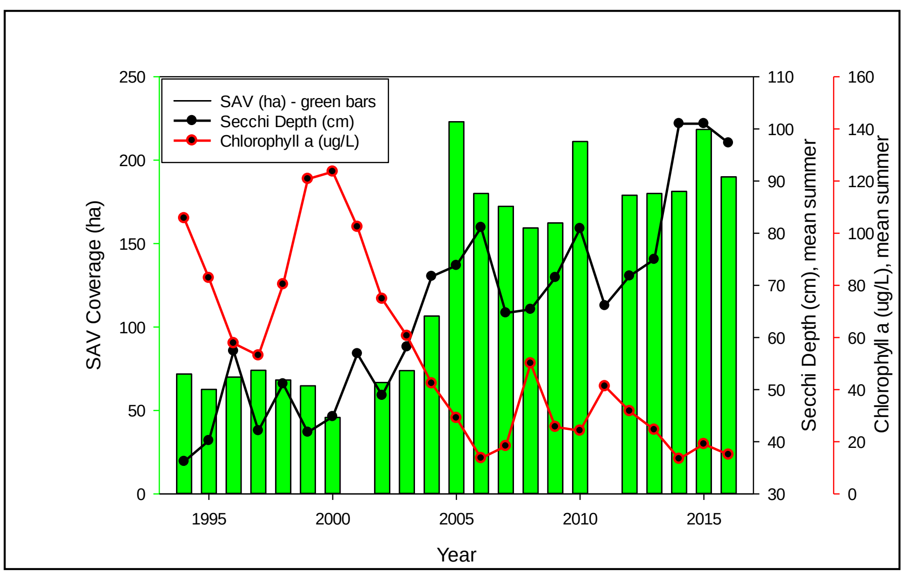
specimens of 15 species in the two fyke nets, which is a little bit less than last year. While Banded Killifish is abundant here as well (23% of the catch), the fyke nets show a high contribution of sunfishes too.

The coverage of submersed aquatic vegetation (SAV) in 2016 was similar to recent years. In 2016, species distributions were mapped. The exotic plant *Hydrilla* was the most dense and widespread species, but the native species *Ceratophyllum* (coontail) was also widespread. As in most previous years, oligochaetes were the most common invertebrates collected in ponar samples in 2016. Chironomids were the second most abundant in the cove, but were found at much lower levels in the river. Amphipods were the second most abundant taxon at Station 9 with isopods also very common.

In the anadromous creek survey (of fish migrating from salt water to spawn in fresh water), Alewife was the dominant species in both larval and adult collections in both Pohick and Accotink Creeks. In the hoop net sets, 170 Alewife, 89 Blueback Herring, and 21 Hickery Shad adults were collected. While these numbers were lower than observed in 2015, they are still strong relative to previous years. In a notable sign of recovery Pohick Creek, which was totally devoid of spawning fish in the early years of the study, now typically harbors more spawners than Accotink Creek. In fact, almost all of the Blueback Herring and Hickory Shad spawning was in Pohick Creek.

Two literature-based special reports were commissioned in 2016, both concerned with benthic macroinvertebrates. These reports were included as separate chapters at the end of this full annual report. One involved development of a benthic index of biotic integrity for the tidal freshwater Potomac River. Progress was made in compiling a complete list of potential macroinvertebrate taxa and features of their ecology like pollution tolerance which are required for index development. Additional data needs such as reference site data were identified. The second special report compiled relevant information on the status and diversity of native freshwater mussels in the tidal freshwater Potomac River. This lays the ground for enhanced efforts to sample these valuable indicator organisms.

Data from 2016 generally reinforced the major trends which were reported in previous years. First, phytoplankton algae populations (which can cause nuisance algal blooms, hypoxia in stratified areas, and a decline of fisheries) in Gunston Cove have shown a clear pattern of declined since 1989. Accompanying this decline have

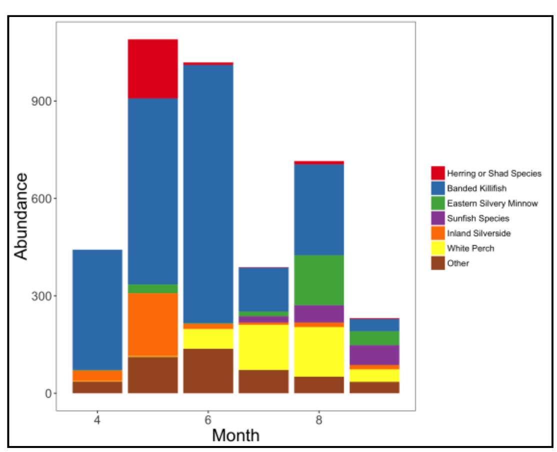


been more normal levels of pH and dissolved oxygen, and increased water clarity which are critical for a life-sustaining aquatic habitat. Data available through 2016 from Virginia Institute of Marine Science for SAV (submersed aquatic vegetation) assessment have indicated that the coverage by plants has remained at elevated levels observed since 2005 (green bars in figure above). The increased water clarity in the Cove has brought the rebound of SAV which provides increased habitat value for fish and fish food organisms. The SAV also filters nutrients and sediments and itself will inhibit the overgrowth of phytoplankton algae. This trend is undoubtedly the result of phosphorus removal practices at Noman M. Cole Pollution Control Plant which were initiated in the late 1970's (see first figure in Executive Summary). This lag period of 10-15 years between phosphorus control and phytoplankton decline has been observed in many freshwater systems resulting at least partially from sediment loading to the water column which can continue for a number of years. Gunston Cove is now an internationally recognized case study for ecosystem recovery due to the actions that were taken and the subsequent monitoring to validate the response.

A second significant change in water quality documented by the study has been the removal of chlorine and ammonia from the Noman M. Cole, Jr. Pollution Control Plant effluent. A decline of over an order of magnitude in ammonia nitrogen has been observed in the Cove as compared to earlier years. The declines in ammonia and the elimination of chlorine from the effluent (to values well below those that may result is toxicity problems) have allowed fish to recolonize tidal Pohick Creek which now typically has more spawning activity than tidal Accotink Creek. Monitoring of creek fish allowed us to observe recovery of this habitat which is very important for spawning species such as shad. The decreased ammonia, suspended solids, and phosphorus loading from the plant have contributed to overall Chesapeake Bay cleanup.

Another trend of significance which is indicative of the Cove recovery is changes in the relative abundance of fish species. While it is still the dominant species in trawls, White Perch has gradually been displaced in seines by Banded Killifish. This trend continued in 2016 with Banded Killifish being much more abundant in seines than White Perch. In general this is a positive development as the net result has been a more diverse fish community. Blue Catfish have entered the area recently, and brown bullhead has decreased greatly in the Cove. Blue Catfish are regarded as rather voracious predators and may negatively affect the food web.

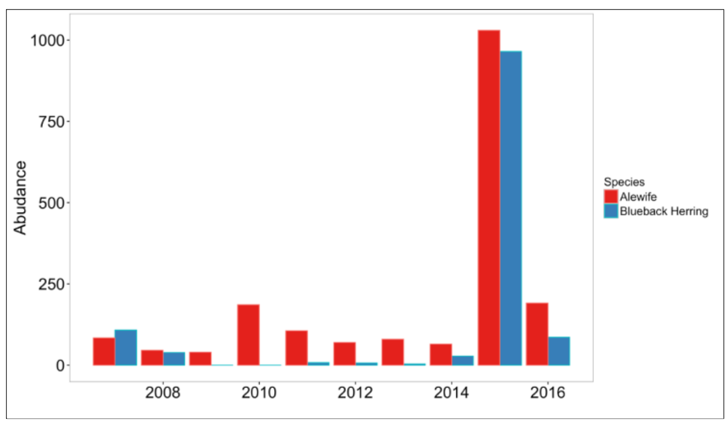
Clearly, recent increases in SAV provide refuge and additional spawning habitat for Banded Killifish and Sunfish. Analysis shows that White Perch dominance was mainly indicative of the community present when there was no SAV; increased abundances of Bay Anchovy indicative for the period with some SAV; and Banded Killifish and Largemouth Bass indicative of the period when SAV beds were expansive. In 2016 seine collections were dominated by Banded Killifish (see graph to the right).



While the seine does not sample these SAV areas directly, the enhanced growth of SAV provides a large bank of Banded Killifish that spread out into the adjacent unvegetated shoreline areas and are sampled in the seines. The fyke nets that do sample the SAV areas directly documented a dominance of Sunfish and Banded Killifish in the SAV beds. In addition to SAV expansion, the invasive Blue Catfish may also have both direct (predation) and indirect (competition) effects, especially on species that occupy the same niche such as Brown Bullhead and Channel Catfish. Overall, these results indicate that the fish assemblage in Gunston Cove is dynamic and supports a diversity of commercial and recreational fishing activities.

Juvenile anadromous species continue to be an important component of the fish assemblage in Gunston Cove. We have seen declines in "river herring" (a multispecies group that includes both Alewife and Blueback Herring) since the mid-1990s, which is in concordance with other surveys around the Potomac and Chesapeake watersheds. In January 2012, a moratorium on river herring was put in effect to alleviate fishing pressure in an effort to help stocks rebound. We reported last year that the larval abundances of the *Alosa* genus was high in 2014, possibly resulting in higher adult abundances in 2015. We indeed saw higher numbers of juvenile Blueback Herring and Alewife in trawls in 2015, but this was not repeated in 2016.

The most direct indication we have of the status of river herring spawning populations is the anadromous study in Pohick and Accotink Creeks (which included Dogue Creek and Quantico Creek up to 2008). We witnessed a one to two orders of magnitude increase in catches from Accotink and Pohick Creeks of Alewife and Blueback Herring (the two species that are considered river herring) in 2015; 2016 catches were somewhat lower, but still substantial (figure to the right). The shad moratorium has been in place in Virginia and neighboring states for four years, which means this is likely the first cohort protected by this moratorium for one full life cycle. Through meetings with the Technical Expert Working group (TEWG) for river herring (http://www.greateratlantic.fisheries.noaa.gov/protected/riverherring/tewg/index.html), it has become clear that not all tributaries of the Chesapeake Bay, in Virginia and elsewhere, have seen increased abundances in 2015; some surveyors even reported declines. Since the decline in river herring was related both to overfishing and habitat degradation, it could be the case that habitat in those areas has not recovered sufficiently to support a larger spawning population now that fishing pressure is released. Thus, the habitat in the Gunston Cove may be of suitable quality to support a larger spawning population now that reduced fishing pressure allows for more adults to return to their natal streams. Continued monitoring in years after this large spawning population was observed, will determine if this spawning season results in a successful year class, and if this is the first year of continued high river herring abundances.



In summary, it is important to continue the data record that has been established to allow assessment of how the continuing increases in volume and improved efforts at wastewater treatment interact with the ecosystem as SAV increases and plankton and fish communities change in response. Furthermore, changes in the fish communities from the standpoint of habitat alteration by SAV and introductions of exotics like snakeheads and blue catfish need to be followed.

Global climate change is becoming a major concern worldwide. Since 2000 a slight, but consistent increase in summer water temperature has been observed in the Cove which may reflect the higher summer air temperatures documented globally. Other potential effects of directional climate change remain very subtle and not clearly differentiated given seasonal and cyclic variability.

We recommend that:

1. Long term monitoring should continue. The revised schedule initiated in 2004 which focuses sampling in April through September has captured the major trends affecting water quality and the biota. The Gunston Cove study is a model for long term monitoring which is necessary to document the effectiveness of management actions. This process is sometimes called adaptive management and is recognized as the most successful approach to ecosystem management.
2. Two aspects of the program should be reviewed.
    a. In 2016 phytoplankton cell counts frequency was decreased from twice monthly to monthly as a cost-saving step. But it does result in some sampling dates not having phytoplankton data to go along with the other variables. If funds are available, we recommend reinstituting twice monthly phytoplankton counts.
    b. As nutrient concentrations have decreased in the river and cove due to management successes, we are now encountering a substantial number of samples which are below detection limits. This becomes a problem in data analysis. To date we have set “below dection limits” values at ½ the detection limit, but this becomes less defensable the greater the proportion of these values. This is particularly true of nitrate and ammonia nitrogen. We recommend reviewing analytical protocols to try to lower detection limits for these two variables.
3. The fyke nets have proven to be a successful addition to our sampling routine. Even though a small, non-quantitative sample is collected due to the passive nature of this gear, it provides us with useful information on the community within the submersed aquatic vegetation beds. Efficient use of time allows us to include these collections in a regular sampling day with little extra time or cost. We recommend continuing with this gear as part of the sampling routine in future years.
4. Anadromous fish sampling is an important part of this monitoring program and has gained interest now that the stock of river herring has collapsed, and a moratorium on these taxa has been established in 2012. We recommend continued monitoring, and we plan to use the collections before and during the moratorium to help determine the effect of the moratorium. Our collections will also form the

basis of a population model that can provide information on the status of the stock.

5. GMU's Potomac Environmental Research and Education Center instituted a continuous water quality monitoring site at Pohick Bay marina in May 2011. This program was suspended in 2014 due to ramp construction near the monitor, but we will consider reinstituting the program in 2017 should the County consider it valuable.
6. As river restoration continues, the benthic community including native mussels is showing signs of rejuvenation. We recommend that more use be made of the benthos in tracking recovery of the River. To that end we recommend that the initative to construct a Benthic Index of Biotic Integrity (B-IBI) for the tidal Potomac River be continued with the goal of having a trial index available by the end of the next contract.
7. The assessment of native river mussel populations which was completed in 2016 found that there is a substantial pool of potential mussel species in the river, but we are not using effective methods to sample them. We propose to try out a new sampling system called a brail with the goal of accurately and comprehensively inventorying the current status of river mussels in the tidal freshwater Potomac.
8. Recent work has raised awareness that some pollutants may be causing sublethal stress on fish populations which are manifest in higher incidences of disease and abnormalities. We recommend that that a pilot study be done to establish a baseline of the incidence of these impacts in specific Gunston Cove taxa and explore the feasibility of routine assessment of fish abnormalities as part of the monitoring program.

List of Abbreviations

BOD	Biochemical oxygen demand
cfs	cubic feet per second
DO	Dissolved oxygen
ha	hectare
l	liter
LOWESS	locally weighted sum of squares trend line
m	meter
mg	milligram
MGD	Million gallons per day
NS	not statistically significant
NTU	Nephelometric turbidity units
SAV	Submersed aquatic vegetation
SRP	Soluble reactive phosphorus
TP	Total phosphorus
TSS	Total suspended solids
um	micrometer
VSS	Volatile suspended solids
#	number

THE ONGOING AQUATIC MONITORING PROGRAM

FOR THE GUNSTON COVE AREA

OF THE TIDAL FRESHWATER POTOMAC RIVER

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INTRODUCTION

This section reports the results of the on-going aquatic monitoring program for Gunston Cove conducted by the Potomac Environmental Research and Education Center at George Mason University and Fairfax County's Environmental Monitoring Branch. This study is a continuation of work originated in 1984 at the request of the County's Environmental Quality Advisory Committee and the Department of Public Works. The original study design utilized 12 stations in Gunston Cove, the Potomac mainstem, and Dogue Creek. Due to budget limitations and data indicating that spatial heterogeneity was not severe, the study has evolved such that only two stations are sampled, but the sampling frequency has been maintained at semimonthly during the growing season. This sampling regime provides reliable data given the temporal variability of planktonic and other biological communities and is a better match to other biological sampling programs on the tidal Potomac including those conducted by the Maryland Department of Natural Resources and the District of Columbia. Starting in 2004, the sampling period was reduced to April through September and photosynthesis determinations were ended.

The 1984 report entitled "An Ecological Study of Gunston Cove – 1984" (Kelso et al. 1985) contained a thorough discussion of the history and geography of the cove. The reader is referred to that document for further details.

This work's primary objective is to determine the status of biological communities and the physico-chemical environment in the Gunston Cove area of the tidal Potomac River for evaluation of long-term trends. This will facilitate the formulation of well-grounded management strategies for maintenance and improvement of water quality and biotic resources in the tidal Potomac. Important byproducts of this effort are the opportunities for faculty research and student training which are integral to the educational programs at GMU.

The authors wish to thank the numerous individuals and organizations whose cooperation, hard work, and encouragement have made this project successful. We wish to thank the Fairfax County Department of Public Works and Environmental Services, Wastewater Planning and Monitoring Division, Environmental Monitoring Branch, particularly Juan Reyes and Shahram Mohsenin for their advice and cooperation during the study. Benny Gaines deserves recognition for field sample collection on days when Fairfax County collected independent samples. The entire analytical staff at the Noman Cole lab are gratefully acknowledged. The Northern Virginia Regional Park Authority facilitated access to the park and boat ramp. Without a dedicated group of field and laboratory workers this project would not have been possible. PEREC field and lab technician Laura Birsa deserves special recognition for day-to-day operations. Dr. Joris van der Ham headed up field fish collecting. Dr. Saiful Islam conducted phytoplankton counts. Thanks also go to C.J. Schlick, Beverly Bachman, Sammy Alexander, Katie Saalbach, Amanda Sills, Lauren Cross, Chelsea Gray, Larin Isdell, Tabitha King, Casey Pehrson, Kali Rauhe, Kristen Reck, and Chelsea Saber, Claire Buchanan served as a voluntary consultant on plankton identification. Roslyn Cress, Natasha Heinrich, and Lisa Bair were vital in handling personnel and procurement functions.

METHODS

A. Profiles and Plankton: Sampling Day

Sampling was conducted on a semimonthly basis at stations representing both Gunston Cove and the Potomac mainstem (Figures 1a,b). One station was located at the center of Gunston Cove (Station 7) and the second was placed in the mainstem tidal Potomac channel off the Belvoir Peninsula just north of the mouth of Gunston Cove (Station 9). Dates for sampling as well as weather conditions on sampling dates and immediately preceding days are shown in Table 1. Gunston Cove is located in the tidal freshwater section of the Potomac about 20 km (13 miles) downstream from Washington, DC.

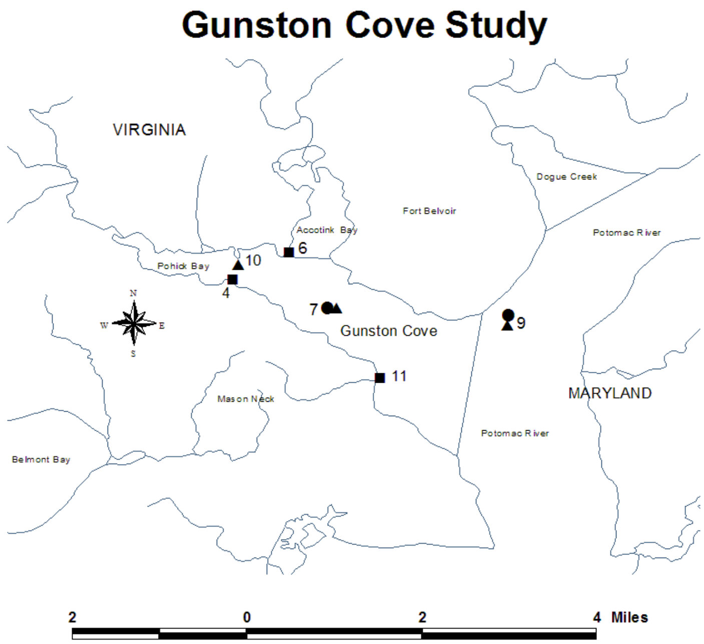


Figure 1a. Gunston Cove area of the Tidal Potomac River showing sampling stations. Circles (●) represent Plankton/Profile stations, triangles (▲) represent Fish Trawl stations, and squares (■) represent Fish Seine stations.

Figure 1b. Fish sampling stations including location and image of the fyke nets.

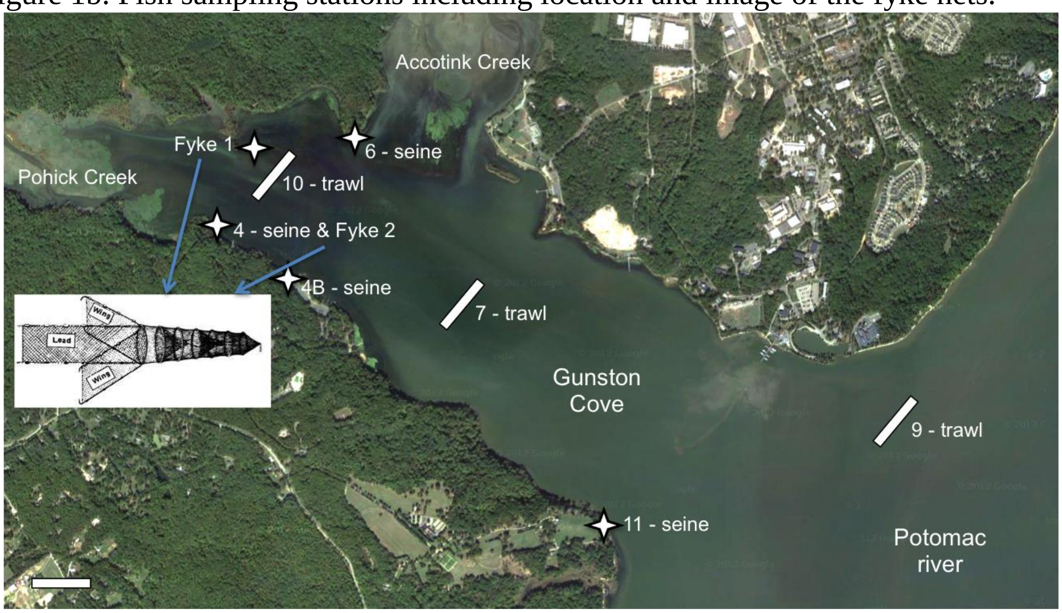


Table 1
Sampling Dates and Weather Data for 2016

	Type of Sampling					Avg Daily Temp (ºC)		Precipitation (cm)	
Date	G	F	T	S	Y	1-Day	3-Day	1-Day	3-Day
April 20	G	F				11.1	13.0	0.03	1.57
April 21			T	S	Y	17.2	18.1	0	0
May 2		F*				18.9	15.0	3.15	4.42
May 4	GB					13.9	17.2	0.08	3.66
May 5			T	S	Y	12.8	15.2	0.05	0.56
May 18	G	F				15.6	14.3	T	1.05
May 19			T	S	Y	17.2	15.6	0.03	1.08
June 15	GB					23.9	23.3	T	T
June 16			T	S	Y	26.1	24.4	1.17	1.17
June 22		F*				25.6	26.7	0	2.51
June 29	G	F	T	S	Y	23.9	25.0	0	3.10
July 13	GB		T	S	Y	28.3	27.0	T	T
July 21		F*				27.2	27.0	0	1.17
July 27	G	F	T	S	Y	31.7	32.0	0	0
August 3	GB		T	S	Y	25.6	27.8	0	0
August 17	G	F	T	S	Y	28.9	29.8	2.24	5.18
August 18		F*				28.3	29.1	T	2.25
Sept 13	G	F	T	S	Y	26.1	25.4	0	0
Sept 27	G	B				21.7	20.4	1.09	1.40

Type of Sampling: B: Benthic, G: GMU profiles and plankton, F: nutrient and lab water quality by Fairfax County Laboratory, T: fish collected by trawling, S: fish collected by seining, Y: fish collected by fyke net. Except as indicated by asterisk, all samples collected by GMU personnel.
*Samples collected by Fairfax County Lab Personnel

Sampling was initiated at 10:30 am. Four types of measurements or samples were obtained at each station : (1) depth profiles of temperature, conductivity, dissolved oxygen, pH, and irradiance (photosynthetically active radiation) measured directly in the field; (2) water samples for GMU lab determination of chlorophyll *a* and phytoplankton species composition and abundance; (3) water samples for determination of nutrients, BOD, alkalinity, suspended solids, chloride, and pH by the Environmental Laboratory of the Fairfax County Department of Public Works and Environmental Services; (4) net sampling of zooplankton and ichthyoplankton.

Profiles of temperature, conductivity, dissolved oxygen, and pH were conducted at each station using a YSI 6600 datasonde. Measurements were taken at 0.3 m, 1.0 m, 1.5 m, and 2.0 m in the cove. In the river measurements were made with the sonde at depths of 0.3 m, 2 m, 4 m, 6 m, 8 m, 10 m, and 12 m. Meters were checked for calibration before and after sampling. Profiles of irradiance (photosynthetically active radiation, PAR) were collected with a LI-COR underwater flat scalar PAR probe. Measurements were taken at 10 cm intervals to a depth of 1.0 m. Simultaneous measurements were made with a terrestrial probe in air during each profile to correct for changes in ambient light if needed. Secchi depth was also determined. The readings of at least two crew members were averaged due to variability in eye sensitivity among individuals.

A 1-liter depth-composited sample was constructed from equal volumes of water collected at each of three depths (0.3 m below the surface, middepth, and 0.3 m off of the bottom) using a submersible bilge pump. A 100-mL aliquot of this sample was preserved immediately with acid Lugol's iodine for later identification and enumeration of phytoplankton. The remainder of the sample was placed in an insulated cooler with ice. A separate 1-liter sample was collected from 0.3 m using the submersible bilge pump and placed in the insulated cooler with ice for lab analysis of surface chlorophyll *a*. These samples were analyzed by Mason.

Separate 4-liter samples were collected monthly at each site from just below the surface (0.3 m) and near the bottom (0.3 m off bottom) at each site using the submersible pump. This water was promptly delivered to the nearby Fairfax County Environmental Laboratory for determination of nitrogen, phosphorus, BOD, TSS, VSS, pH, total alkalinity, and chloride.

Microzooplankton was collected by pumping 32 liters from each of three depths (0.3 m, middepth, and 0.3 m off the bottom) through a 44 µm mesh sieve. The sieve consisted of a 12-inch long cylinder of 6-inch diameter PVC pipe with a piece of 44 µm nitex net glued to one end. The 44 µm cloth was backed by a larger mesh cloth to protect it. The pumped water was passed through this sieve from each depth and then the collected microzooplankton was backflushed into the sample bottle. The resulting sample was treated with about 50 mL of club soda and then preserved with formalin containing a small amount of rose bengal to a concentration of 5-10%.

Macrozooplankton was collected by towing a 202 µm net (0.3 m opening, 2 m long) for 1 minute at each of three depths (near surface, middepth, and near bottom). Ichthyoplankton was sampled by towing a 333 µm net (0.5 m opening, 2.5 m long) for 2 minutes at each of the same depths. In the cove, the boat made a large arc during the tow while in the river the net was towed in a more linear fashion along the channel. Macrozooplankton tows were about 300 m and ichthyoplankton tows about 600 m. Actual distance depended on specific wind conditions and tidal current intensity and direction, but an attempt was made to maintain a constant slow forward speed through the water during the tow. The net was not towed directly in the wake of the engine. A General Oceanics flowmeter, fitted into the mouth of each net, was used to establish the exact towing distance. During towing the three depths were attained by playing out rope equivalent to about 1.5-2 times the desired depth. Samples which had obviously scraped bottom were discarded and the tow was repeated. Flowmeter readings taken before and after towing allowed precise determination of the distance towed and when multiplied by the area of the opening produced the total volume of water filtered.

Macrozooplankton and ichthyoplankton were backflushed from the net cup and immediately preserved. Rose bengal formalin with club soda pretreatment was used for macrozooplankton. Ichthyoplankton were preserved in 70% ethanol. Macrozooplankton was collected on each sampling trip; ichthyoplankton collections ended after July because larval fish were normally not found after this time. On dates when water samples were not being collected for water quality analysis by the Fairfax County laboratory, benthic macroinvertebrate samples were collected. Three samples were collected at each site using a petite ponar grab. The bottom material was sieved through a 0.5 mm stainless steel sieve and resulting organisms were preserved in rose bengal formalin for lab analysis.

Samples were delivered to the Fairfax County Environmental Services Laboratory by 2 pm on sampling day and returned to GMU by 3 pm. At GMU 10-15 mL aliquots of both depth-integrated and surface samples were filtered through 0.45 µm membrane filters (Gelman GN-6 and Millipore MF HAWP) at a vacuum of less than 10 lbs/in^2 for chlorophyll a and pheopigment determination. During the final phases of filtration, 0.1 mL of $MgCO_3$ suspension (1 g/100 mL water) was added to the filter to prevent premature acidification. Filters were stored in 20 mL plastic scintillation vials in the lab freezer for later analysis. Seston dry weight and seston organic weight were measured by filtering 200-400 mL of depth-integrated sample through a pretared glass fiber filter

(Whatman 984AH).

Sampling day activities were normally completed by 5:30 pm.

B. Profiles and Plankton: Follow-up Analyses

Chlorophyll *a* samples were extracted in a ground glass tissue grinder to which 4 mL of dimethyl sulfoxide (DMSO) was added. The filter disintegrated in the DMSO and was ground for about 1 minute by rotating the grinder under moderate hand pressure. The ground suspension was transferred back to its scintillation vial by rinsing with 90% acetone. Ground samples were stored in the refrigerator overnight. Samples were removed from the refrigerator and centrifuged for 5 minutes to remove residual particulates.

Chlorophyll *a* concentration in the extracts was determined fluorometrically using a Turner Designs Model 10 field fluorometer configured for chlorophyll analysis as specified by the manufacturer. The instrument was calibrated using standards obtained from Turner Designs. Fluorescence was determined before and after acidification with 2 drops of 10% HCl. Chlorophyll *a* was calculated from the following equation which corrects for pheophytin interference:

Chlorophyll *a* (µg/L) = $F_s R_s (R_b - R_a)/(R_s - 1)$

where F_s=concentration per unit fluorescence for pure chlorophyll *a*
R_s=fluorescence before acid / fluorescence after acid for pure chlorophyll *a*
R_b=fluorescence of sample before acid
R_a=fluorescence of sample after acid

All chlorophyll analyses were completed within one month of sample collection.

Phytoplankton species composition and abundance was determined using the inverted microscope-settling chamber technique (Lund et al. 1958). Ten milliters of well-mixed algal sample were added to a settling chamber and allowed to stand for several hours. The chamber was then placed on an inverted microscope and random fields were enumerated. At least two hundred cells were identified to species and enumerated on each slide. Counts were converted to number per mL by dividing number counted by the volume counted. Biovolume of individual cells of each species was determined by measuring dimensions microscopically and applying volume formulae for appropriate solid shapes.

Microzooplankton and macrozooplankton samples were rinsed by sieving a well-mixed subsample of known volume and resuspending it in tap water. This allowed subsample volume to be adjusted to obtain an appropriate number of organisms for counting and for formalin preservative to be purged to avoid fume inhalation during counting. One mL subsamples were placed in a Sedgewick-Rafter counting cell and whole slides were analyzed until at least 200 animals had been identified and enumerated. A minimum of two slides was examined for each sample. References for identification were: Ward and Whipple (1959), Pennak (1978), and Rutner-Kolisko (1974). Zooplankton counts were converted to number per liter (microzooplankton) or per cubic meter (macrozooplankton) with the following formula:

$$\text{Zooplankton } (\#/\text{L or } \#/\text{m}^3) = NV_s/(V_cV_f)$$

where N = number of individuals counted
V_s = volume of reconstituted sample, (mL)
V_c = volume of reconstituted sample counted, (mL)
V_f = volume of water sieved, (L or m^3)

Ichthyoplankton sample processing began with removal and sorting of larval fish speciments from the sample with the aid of a stereo dissecting microscope, and the total number of larval fish was counted. Identification of ichthyoplankton was made to family and further to genus and species where possible. The works of Hogue et al. (1976), Jones et al. (1978), Lippson and Moran (1974), and Mansueti and Hardy (1967) were used for identification. The number of ichthyoplankton in each sample was expressed as number per 10 m^3 using the following formula:

$$\text{Ichthyoplankton } (\#/10\text{m}^3) = 10N/V$$

where N = number ichthyoplankton in the sample
V = volume of water filtered, (m^3)

C. Adult and Juvenile Fish

Fishes were sampled by trawling at stations 7, 9, and 10, seining at stations 4, 4B, 6, and 11, and setting fyke nets at stations 4-fyke and 10-fyke (Figure 1a and b). For trawling, a try-net bottom trawl with a 15-foot horizontal opening, a ¾ inch square body mesh and a ¼ inch square cod end mesh was used. The otter boards were 12 inches by 24 inches. Towing speed was 2-3 miles per hour and tow length was 5 minutes. In general, the trawl was towed across the axis of the cove at stations 7 and 10 and parallel to the channel at station 9. The direction of tow should not be crucial. Dates of sampling and weather conditions are found in Table 1. Due to extensive SAV cover, station 10 could not be sampled in June, July, and August. Since this thick SAV cover is now annually recurring, we have adjusted our sampling regime since 2012 by adding fyke nets (Figure 1b).

Seining was performed with seine net that was 50 feet long, 4 feet high, and made of knotted nylon with a ¼ inch square mesh. The seining procedure was standardized as much as possible. The net was stretched out perpendicular to the shore with the shore end in water no more than a few inches deep. The net was then pulled parallel to the shore for a distance of 100 feet by a worker at each end moving at a slow walk. Actual distance was recorded if in any circumstance it was lower than 100 feet. At the end of the prescribed distance, the offshore end of the net was swung in an arc to the shore and the net pulled up on the beach to trap the fish. Dates for seine sampling were generally the same as those for trawl sampling. 4B was added to the sampling stations since 2007 because extensive SAV growth interferes with sampling station 4 in late summer. Sampling with a fyke net near station 4 has been added since 2012 (Figure 1b).

Due to the permanent recovery of the SAV cover in station 4 and station 10, we adjusted our sampling regime in 2012, and have continued with this approach in 2014. Fyke nets were now set in station 4-fyke and station 10-fyke during the entire sampling season. Setting fyke nets when seining and trawling is still possible will allow for gear comparison. Fyke nets were set within the SAV to sample the fish community that uses the SAV cover as habitat. Moving or discontinuing the trawl and seine collections when sampling with those gear types becomes impossible may underrepresent the fish community that lives within the dense SAV cover. Fyke nets were set for 5 hours to passively collect fish. The fyke nets have 5 hoops, a 1/4 inch mesh size, 16 feet wings and a 32 feet lead. Fish enter the net by actively swimming and/or due to tidal motion of the water. The lead increases catch by capturing the fish swimming parallel to the wings (see insert Figure 1b).

After collection with various gear types, the fishes were measured for standard length to the nearest mm. Standard length is the distance from the front tip of the snout to the end of the vertebral column and base of the caudal fin. This is evident in a crease perpendicular to the axis of the body when the caudal fin is pulled to the side.

If the identification of the fish was not certain in the field, the specimen was preserved in 70% ethanol and identified later in the lab. Identification was based on characteristics in dichotomous keys found in several books and articles, including Jenkins and Burkhead (1983), Hildebrand and Schroeder (1928), Loos et al (1972), Dahlberg (1975), Scott and Crossman (1973), Bigelow and Schroeder (1953), Eddy and Underhill (1978), Page and Burr (1998), and Douglass (1999).

D. Submersed Aquatic Vegetation

Data on coverage and composition of submersed aquatic vegetation (SAV) were obtained from the SAV webpage of the Virginia Institute of Marine Science (http://www.vims.edu/bio/sav). Information on this web site was obtained from aerial photographs near the time of peak SAV abundance as well as ground surveys which were used to determine species composition. SAV abundances were also surveyed on August 29. As the research vessel slowly transited the cove, a weighted garden rake was dragged for 10-15 seconds along the bottom and retrieved. Adhering plants were identified and their relative abundance determined. About 40 such measurements were made on that date.

E. Benthic Macroinvertebrates

Benthic macroinvertebrates were sampled using a petite ponar sampler at Stations 7 and 9. Triplicate samples were collected at each site on dates when water samples for Fairfax County lab analysis were not collected. Bottom samples were sieved on site through a 0.5 mm stainless steel sieve and preserved with rose bengal formalin. In the laboratory benthic samples were rinsed with tap water through a 0.5 mm sieve to remove formalin preservative and resuspended in tap water. All organisms were picked, sorted, identified and enumerated.

F. Data Analysis

Several data flows were merged for analysis. Water quality data emanating from the Noman Cole laboratory was used for graphs of both current year seasonal and spatial patterns and long term trends. Water quality, plankton, benthos and fish data were obtained from GMU samples. Data for each parameter were entered into spreadsheets (Excel or SigmaPlot) for graphing of temporal and spatial patterns for the current year. Long term trend analysis was conducted with Systat by plotting data for a given variable by year and then constructing a LOWESS trend line through the data. For water quality parameters the trend analysis was conducted on data from the warmer months (June-September) since this is the time of greatest microbial activity and greatest potential water quality impact. For zooplankton and fish all data for a given year were used. When graphs are shown with a log axis, zero values have been ignored in the trend analysis. JMP v8.0.1was used for fish graphs. Linear regression and standard parametric (Pearson) correlation coefficients were conducted to determine the statistical significance of linear trends over the entire period of record.

RESULTS

A. Climatic and Hydrologic Factors - 2016

In 2016 air temperature was above average for most of the year including all but one of the months when sampling occurred (Table 2). July and August were the warmest months, 2°C and 3°C above normal, respectively. September was also 3°C higher than the long term average. There were 52 days with maximum temperature above 32.2^{o}C (90^{o}F) during 2016 compared with 41 in 2015. 2016 had the second highest number of these days since 2004. Precipitation was well above normal during May, but close to normal in the other months when sampling occurred. The largest daily rainfall total was on May 2 with over 3 cm on top of May 1 with 1.2 cm.

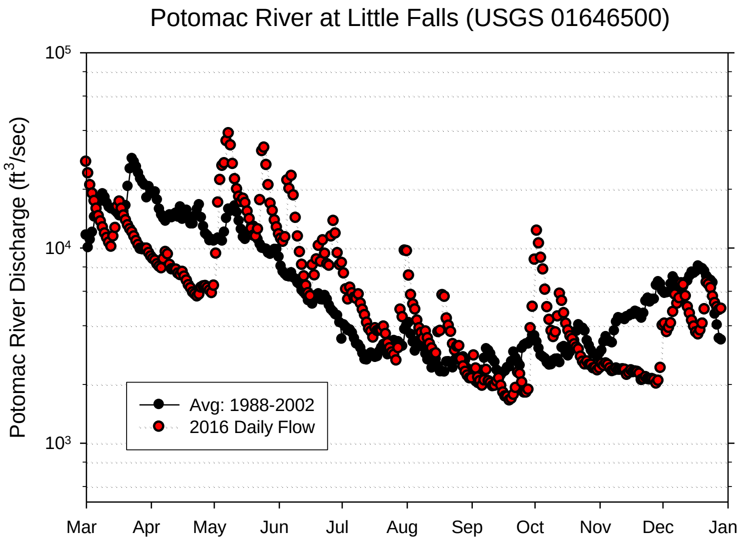
Table 2. Meteorological Data for 2016. National Airport. Monthly Summary.

MONTH	Air Temp (^{o}C)		Precipitation (cm)	
March	11.9	(8.1)	3.0	(9.1)
April	13.8	(13.4)	5.2	(7.0)
May	17.7	(18.7)	14.4	(9.7)
June	24.6	(23.6)	9.4	(8.0)
July	28.2	(26.2)	8.0	(9.3)
August	28.2	(25.2)	7.1	(8.7)
September	24.4	(21.4)	6.4	(9.6)
October	17.3	(14.9)	2.3	(8.2)
November	11.4	(9.3)	1.9	(7.7)
December	5.4	(4.2)	6.6	(7.8)

Note: 2016 monthly averages or totals are shown accompanied by long-term monthly averages (1971-2000).
Source: Local Climatological Data. National Climatic Data Center, National Oceanic and Atmospheric Administration.

Table 3. Monthly mean discharge at USGS Stations representing freshwater flow into the study area. (+) 2016 month > 2x Long Term Avg. (-) 2016 month < ½ Long Term Avg.

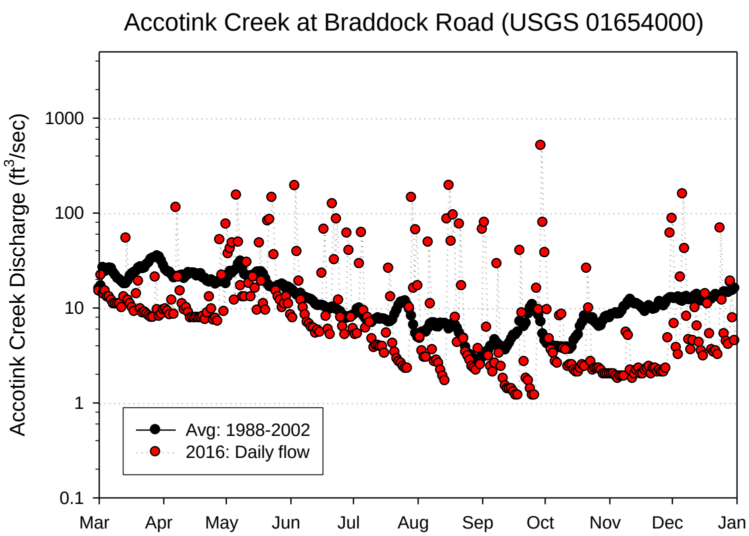
	Potomac River at Little Falls (cfs)		**Accotink Creek at Braddock Rd (cfs)**	
	2016	**Long Term Avg.**	**2016**	**Long Term Avg.**
March	13844	23600	13 (-)	42
April	7258 (-)	20400	15 (-)	36
May	19954	15000	35	34
June	11016	9030	28	28
July	4674	4820	15	22
August	3925	4550	21	22
September	2177 (-)	5040	29	27
October	7504	5930	5.3 (-)	19



In a tidal freshwater system like the Potomac River, river flow entering from upstream is important in maintaining freshwater conditions and also serves to bring in dissolved and particulate substances from the watershed. High freshwater flows may also flush planktonic organisms downstream and bring in suspended sediments that decrease water clarity. The volume of river flow per unit time is referred to as "river discharge" by hydrologists. Note the long term seasonal pattern of higher discharges in winter and spring and lower discharges in summer and fall.

Figure 2. Mean Daily Discharge: 2016. Potomac River at Little Falls (USGS Data). Month tick is at the beginning of the month.

Potomac River discharge during 2016 was below normal in March and April, but generally above normal in May and June (Table 3, Figure 2). July and August had periods of higher flows. Accotink Creek flows followed a similar pattern with most sampling months near normal (Figure 3). Throughout the year there were large, short lived flow peaks due to individual storms.



In the Gunston Cove region of the tidal Potomac, freshwater discharge is occurring from both the major Potomac River watershed upstream (measured at Little Falls) and from immediate tributaries. The cove tributary for which stream discharge is available is Accotink Creek. Accotink Creek delivers over half of the stream water which directly enters the cove. While the gauge at Braddock Road only covers the upstream part of the watershed it is probably representative.

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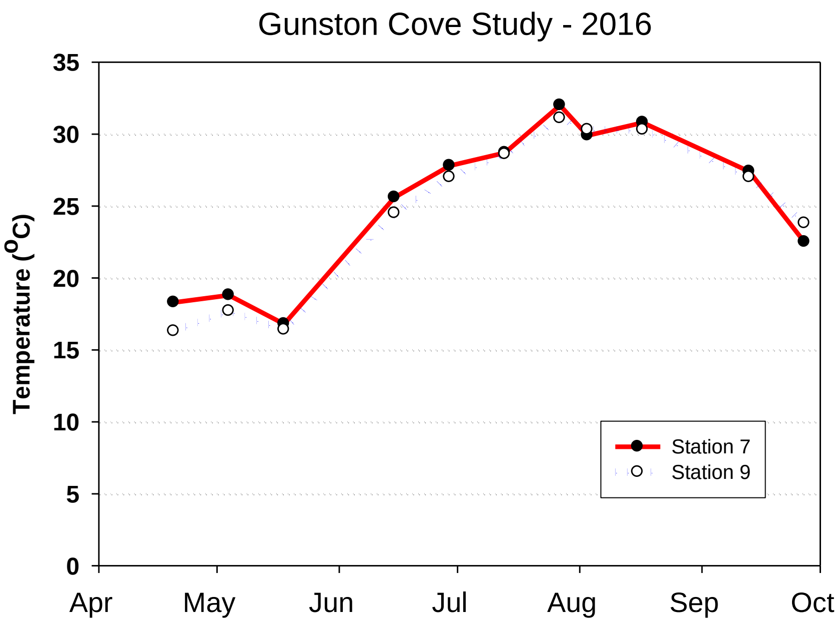
Figure 3. Mean Daily Discharge: 2016. Accotink Creek at Braddock Road (USGS Data).

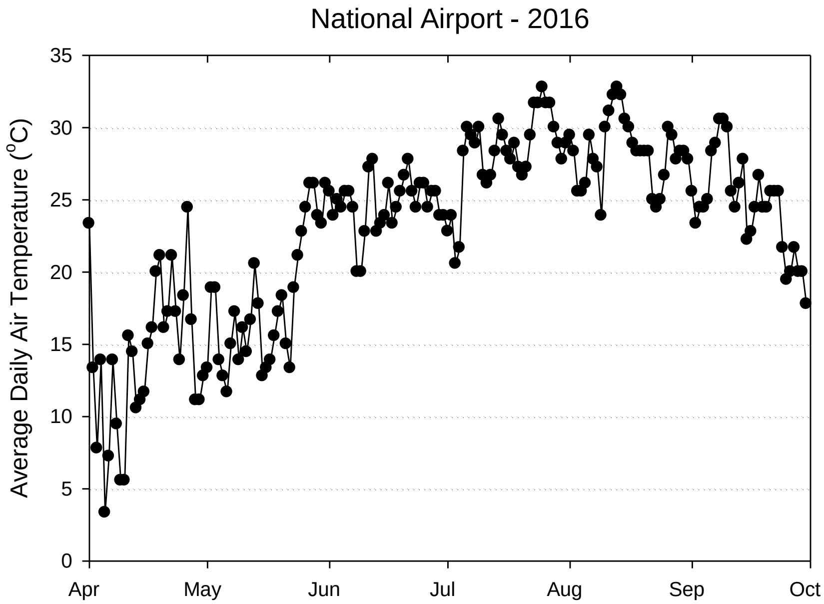
B. Physico-chemical Parameters – 2016



Water temperature is an important factor affecting both water quality and aquatic life. In a well-mixed system like the tidal Potomac, water temperatures are generally fairly uniform with depth.
In a shallow mixed system such as the tidal Potomac, water temperature often closely tracks daily changes in air temperature.

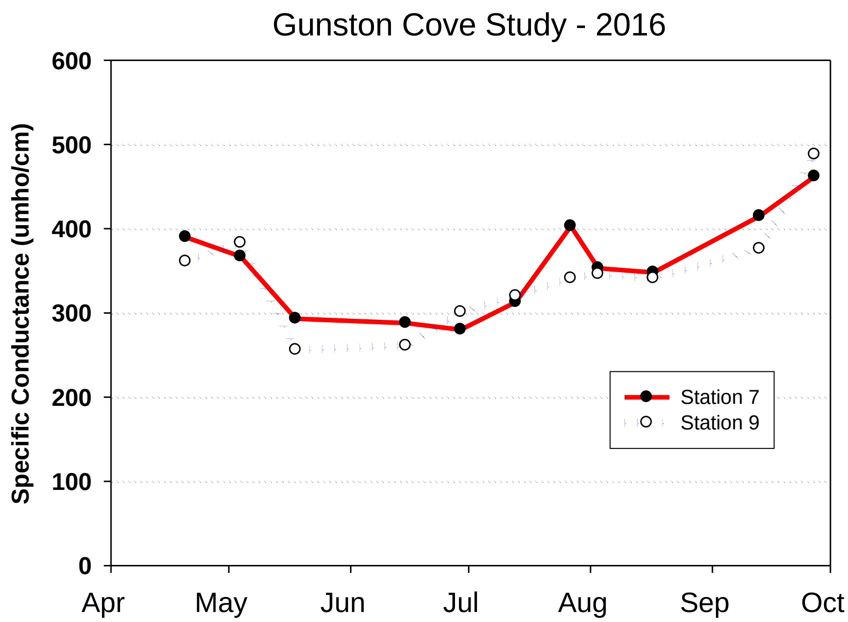
Figure 4. Water Temperature (°C). GMU Field Data. Month tick is at first day of month.

In 2016, water temperature followed the typical seasonal pattern at both sites (Figure 4). Both sites showed an early spring increase which leveled off through May. Both sites exceeded 30°C in late July and early August, the warmest months for air temperature. For most of the summer, the two stations showed very similar water temperatures. Water temperature declined in September.



Mean daily air temperature (Figure 5) was a good predictor of water temperature (Figure 4). Variations in daily air temperature were more pronounce in the spring than in the summer.

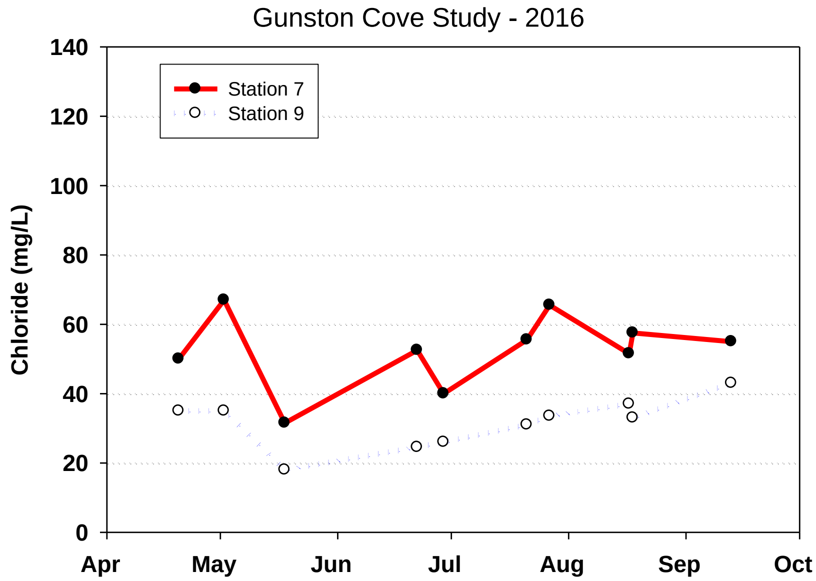
Figure 5. Average Daily Air Temperature (°C) at Reagan National Airport.



Specific conductance measures the capacity of the water to conduct electricity standardized to 25ºC. This is a measure of the concentration of dissolved ions in the water. In freshwater, conductivity is relatively low. Ion concentration generally increases slowly during periods of low freshwater inflow and decreases during periods of high freshwater inflow. In years of low freshwater inflow during the summer and fall, conductance may increase dramatically if brackish water from the estuary reaches the study area.

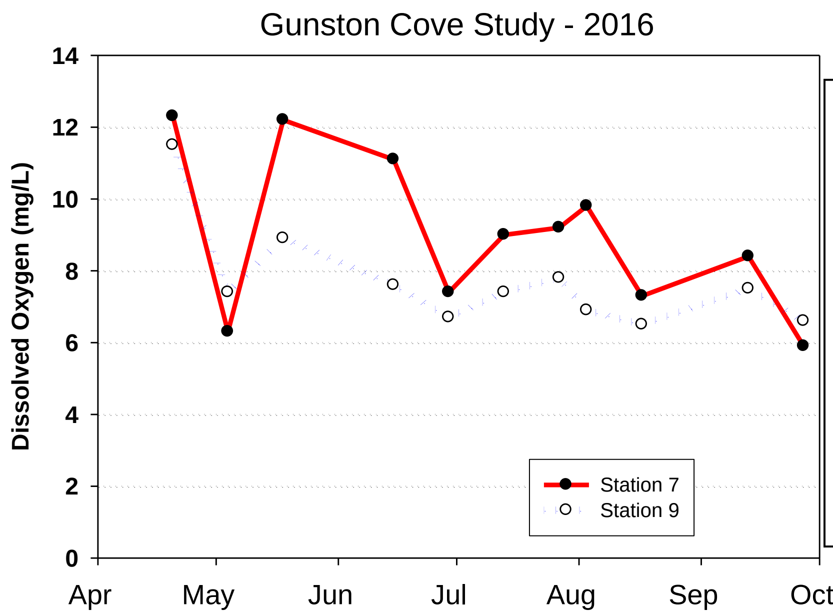
Figure 6. Specific Conductance (uS/cm). GMU Field Data. Month tick is at first day of month.

Specific conductance decreased from April through mid June due to the wet period during May (Figure 6). From June through September specific conductance increased steadily at both sites reaching a similar maximum in late September at both sites. Chloride ion was consistently higher at Station 7 and exhibited a less marked seasonal pattern (Figure 7). Perhaps the higher levels of chloride in the cove were due to inflows from the Noman Cole plant.



Chloride ion (Cl-) is a principal contributor to conductance. Major sources of chloride in the study area are sewage treatment plant discharges, road salt, and brackish water from the downriver portion of the tidal Potomac. Chloride concentrations observed in the Gunston Cove area are very low relative to those observed in brackish, estuarine, and coastal areas of the Mid-Atlantic region. Chloride often peaks markedly in late summer or fall when brackish water from down estuary may reach the cove as freshwater discharge declines.

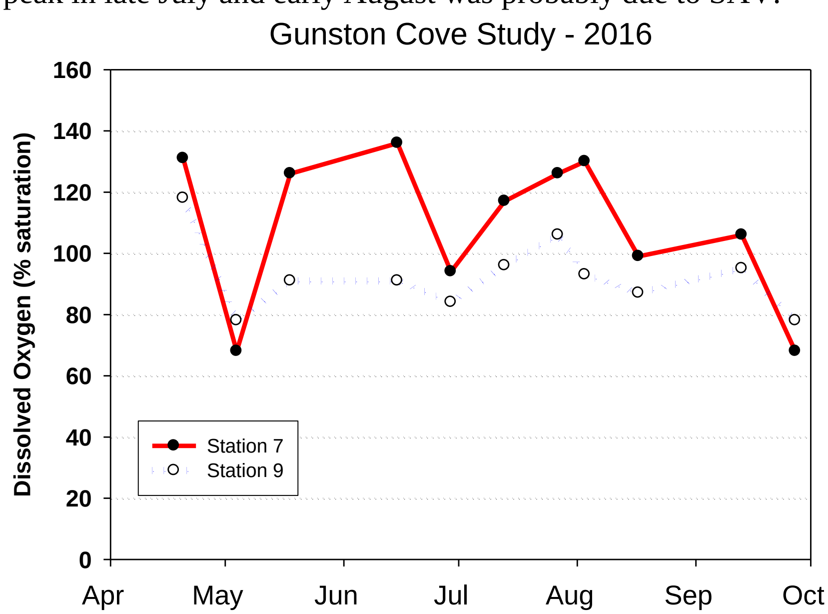
Figure 7. Chloride (mg/L). Fairfax County Lab Data. Month tick is at first day of month.



Oxygen dissolved in the water is required by freshwater animals for survival. The standard for dissolved oxygen (DO) in most surface waters is 5 mg/L. Oxygen concentrations in freshwater are in balance with oxygen in the atmosphere, but oxygen is only weakly soluble in water so water contains much less oxygen than air. This solubility is determined by temperature with oxygen more soluble at low temperatures.

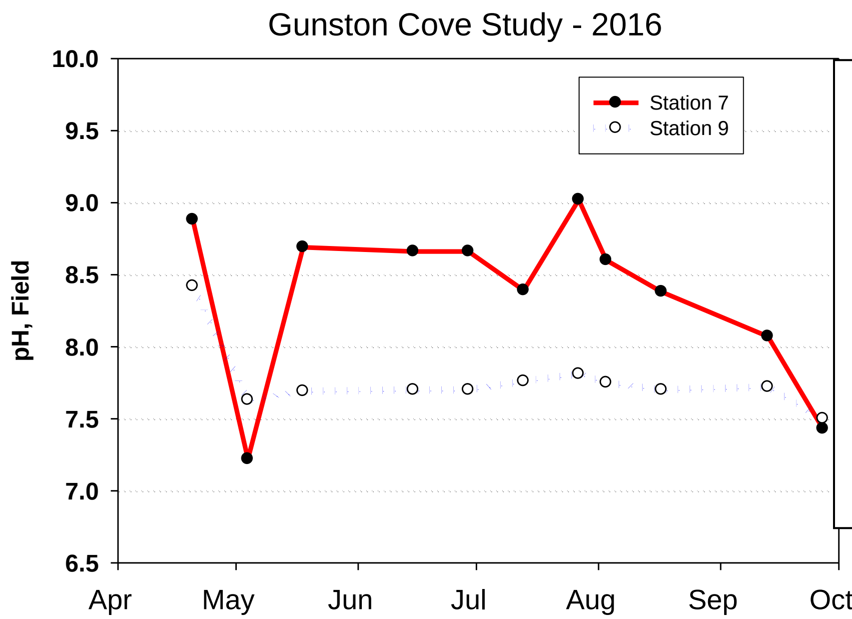
Figure 8. Dissolved Oxygen (mg/L). GMU Field Data. Month tick is at first day of month.

Dissolved oxygen showed substantial differences between the two stations for most of the year (Figure 8). From late May through early September the two sites diverged with Station 7 in Gunston Cove consistently exhibiting much higher values. Figure 9 shows that dissolved oxygen levels in the cove were often substantially above 100% indicating abundant photosynthesis by SAV and phytoplankton. In the river values were generally equal or less than 100% indicating lower photosynthesis and an excess of respiration. A major peak in early June in the cove was probably attributable to phytoplankton while the peak in late July and early August was probably due to SAV.



The temperature effect on oxygen concentration can be removed by calculating DO as percent saturation. This allows examination of the balance between photosynthesis and respiration both of which also impact DO. Photosynthesis adds oxygen to the water while respiration removes it. Values above 120% saturation are indicative of intense photosynthesis while values below 80% reflect a preponderance of respiration or decomposition.

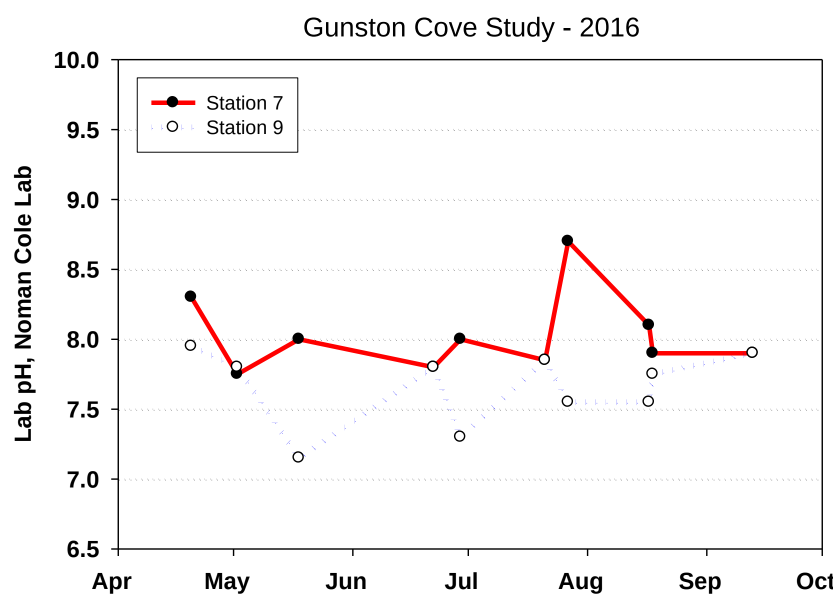
Figure 9. Dissolved Oxygen (% saturation). GMU Field Data. Month tick is at first day of month.



pH is a measure of the concentration of hydrogen ions (H+) in the water. Neutral pH in water is 7. Values between 6 and 8 are often called circumneutral, values below 6 are acidic and values above 8 are termed alkaline. Like DO, pH is affected by photosynthesis and respiration. In the tidal Potomac, pH above 8 indicates active photosynthesis and values above 9 indicate intense photosynthesis.

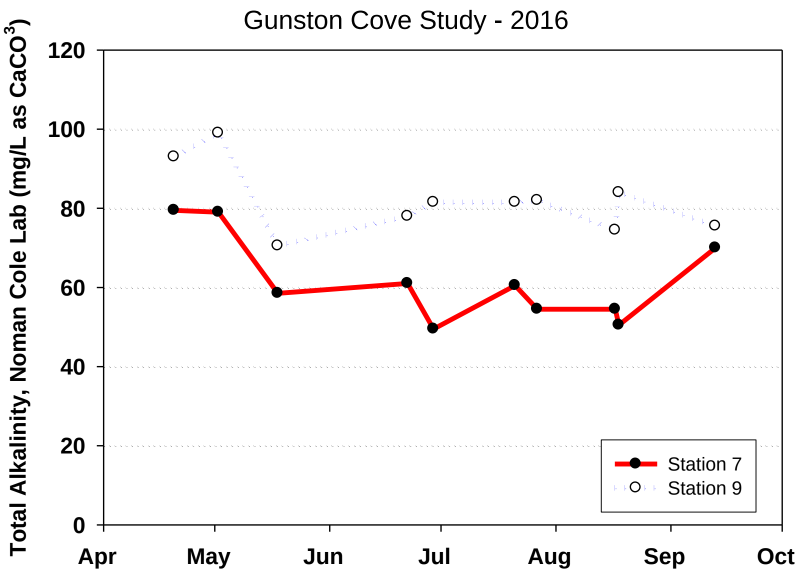
Figure 10. pH. GMU Field Data. Month tick is at first day of month.

Field pH was consistently greater in the cove than in the river again reflecting differences in photosynthetic activity (Figure 10). Times of pH peaks generally corresponded to those in dissolved oxygen. Lab pH was collected less frequently, but generally showed similar patterns (Figure 11).



pH may be measured in the field or in the lab. Field pH is more reflective of in situ conditions while lab pH is done under more stable and controlled laboratory conditions and is less subject to error. Newer technologies such as the Hydrolab and YSI sondes used in GMU field data collection are more reliable than previous field pH meters and should give results that are most representative of values actually observed in the river.

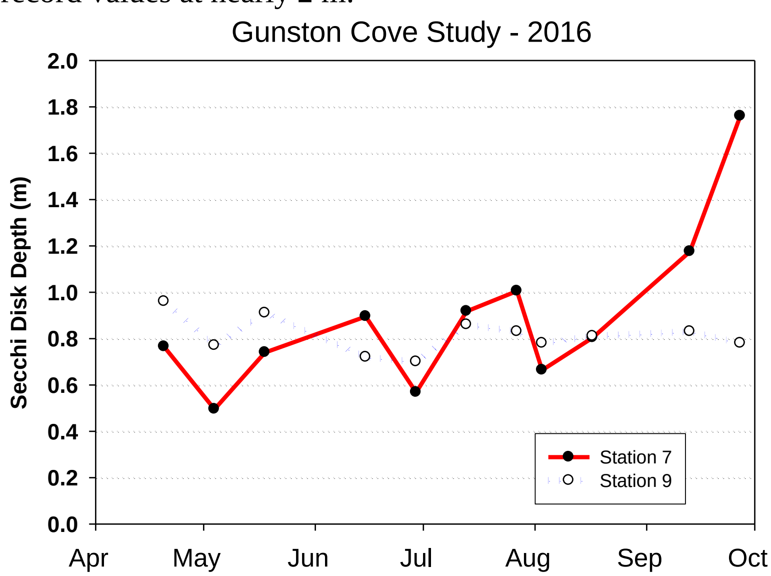
Figure 11. pH. Noman Cole Lab Data. Month tick is at first day of month.



Total alkalinity measures the amount of bicarbonate and carbonate dissolved in the water. In freshwater this corresponds to the ability of the water to absorb hydrogen ions (acid) and still maintain a near neutral pH. Alkalinity in the tidal freshwater Potomac generally falls into the moderate range allowing adequate buffering without carbonate precipitation.

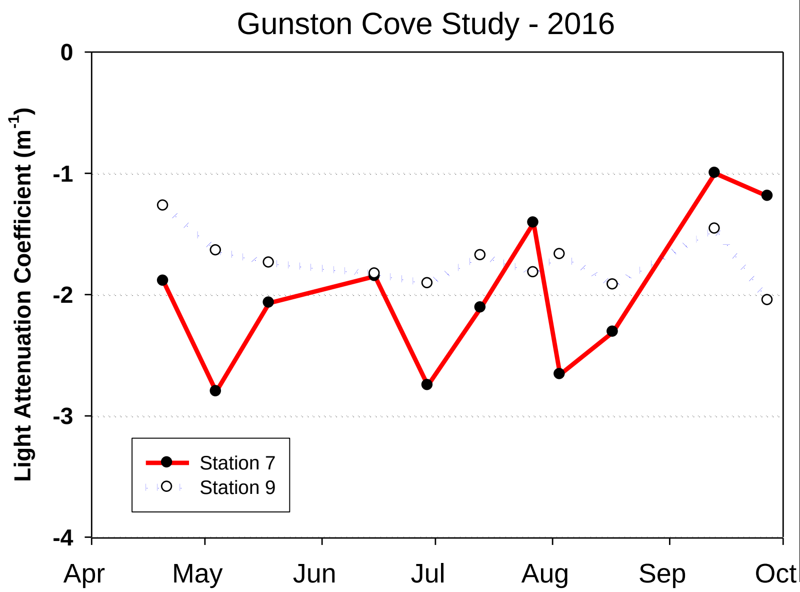
Figure 12. Total Alkalinity (mg/L as $CaCO_3$). Fairfax County Lab data. Month tick is at first day of month.

Total alkalinity was consistently higher in the river than in the cove by about 20 units (Figure 12). Water clarity as reflected by Secchi disk depth was generally similar at both sites, but in September it was much greater in the cove (Figure 13). On these two dates, summer Secchi exceeded 1 m consistently and in fact in early September approached record values at nearly 2 m.



Secchi Depth is a measure of the transparency of the water. The Secchi disk is a flat circle or thick sheet metal or plywood about 6 inches in diameter which is painted into alternate black and white quadrants. It is lowered on a calibrated rope or rod to a depth at which the disk disappears. This depth is termed the Secchi Depth. This is a quick method for determining how far light is penetrating into the water column. Light is necessary for photosynthesis and thereby for growth of aquatic plants and algae.

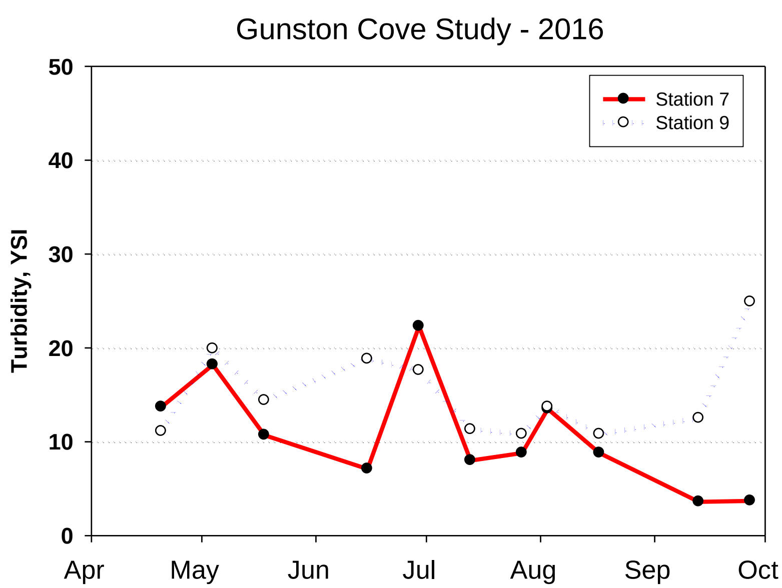
Figure 13. Secchi Disk Depth (m). GMU Field Data. Month tick is at first day of month.



Light Attenuation is another approach to measuring light penetration. This is determined by measuring light levels at a series of depths starting near the surface. The resulting relationship between depth and light is fit to a semi-logarithmic curve and the resulting slope is called the light attenuation coefficient. This relationship is called Beer's Law. It is analogous to absorbance on a spectrophotometer. The greater the light attenuation, the faster light is absorbed with depth. More negative values indicate greater attenuation. Greater attenuation is due to particulate and dissolved material which absorbs and deflects light.

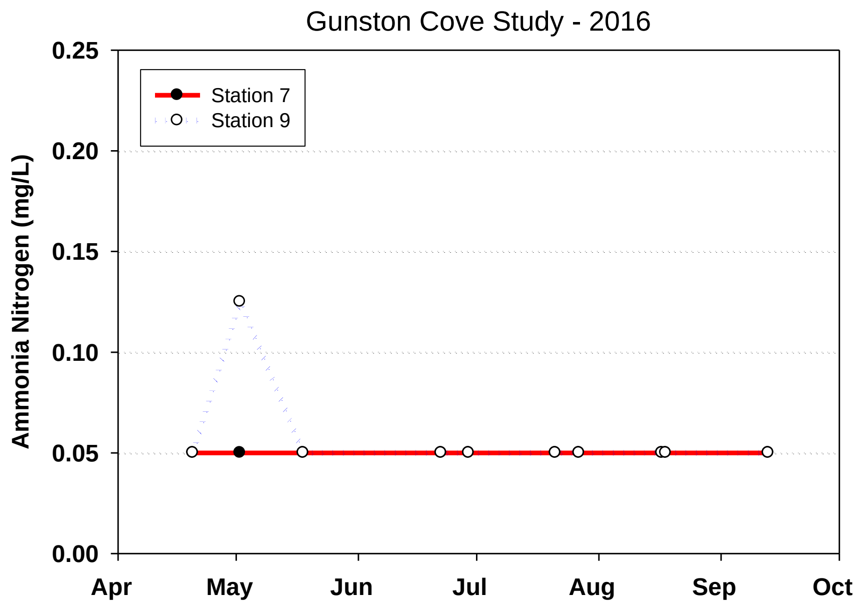
Figure 14. Light Attenuation Coefficient (m^{-1}). GMU Field Data. Month tick is at first day of month.

Light attenuation coefficient generally fell in the range -1.0 to -3.0 m^{-1} (Figure 14). Temporal and spatial trends were similar to those for Secchi depth. Light attenuation was less variable in the river than in the cove. Turbidity was generally slightly lower in the cove than in the river except in September when turbidity was very low in the cove (high turbidity corresponds to low transparency) (Figure 15).



Turbidity is yet a third way of measuring light penetration. Turbidity is a measure of the amount of light scattering by the water column. Light scattering is a function of the concentration and size of particles in the water. Small particles scatter more light than large ones (per unit mass) and more particles result in more light scattering than fewer particles.

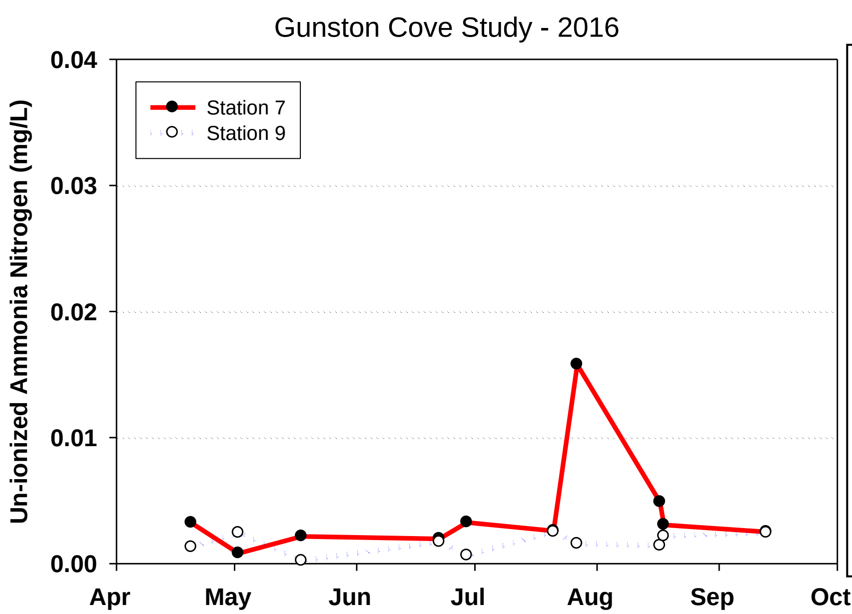
Figure 15. Turbidity (NTU). GMU Lab Data. Month tick is at first day of month.



Ammonia nitrogen measures the amount of ammonium ion (NH_4^+) and ammonia gas (NH_3) dissolved in the water. Ammonia nitrogen is readily available to algae and aquatic plants and acts to stimulate their growth. While phosphorus is normally the most limiting nutrient in freshwater, nitrogen is a close second. Ammonia nitrogen is rapidly oxidized to nitrate nitrogen when oxygen is present in the water.

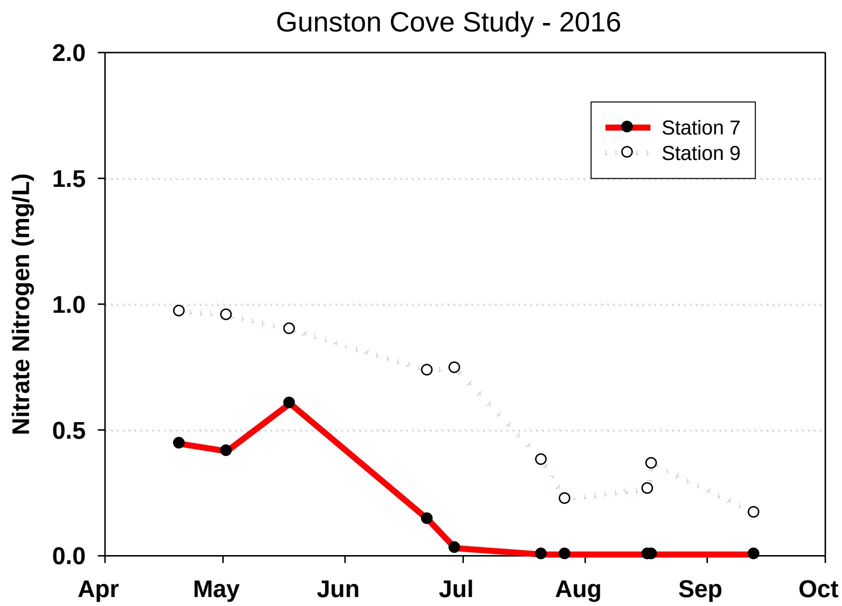
Figure 16. Ammonia Nitrogen (mg/L). Fairfax County Lab Data. Month tick is at first day of month. (Limit of detection: 0.10 mg/L, LD values graphed as 0.05 mg/L)

Ammonia nitrogen was consistently low in the study area during 2016 (Figure 16). All but one value was below the limits of detection which makes analyzing any temporal or spatial trends impossible. Un-ionized ammonia was very low at both stations through the entire year although these are based on approximations of ammonia N (Figure 17). Values were well below those causing toxicity problems.



Un-ionized ammonia nitrogen refers to ammonia gas (NH_3) dissolved in the water. This form is of interest because of its toxicity to aquatic life. The amount of un-ionized ammonia is a function of total ammonia, pH, and temperature. pH is especially important since as pH rises above 9, un-ionized ammonia rapidly increases. Un-ionized ammonia concentrations above 1 mg/L, well in excess of those observed here, are considered toxic to aquatic life.

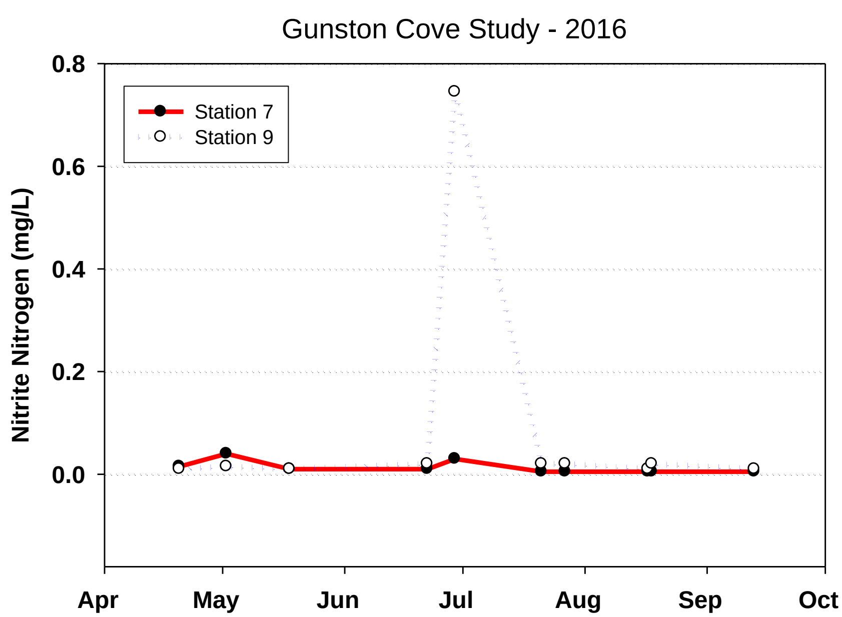
Figure 17. Un-ionized Ammonia Nitrogen (mg/L). Fairfax County Lab Data. Month tick is at first day of month.



Nitrate Nitrogen refers to the amount of N that is in the form of nitrate ion (NO_3^-). Nitrate ion is the most common form of nitrogen in most well oxidized freshwater systems. Nitrate concentrations are increased by input of wastewater, nonpoint sources, and oxidation of ammonia in the water. Nitrate concentrations decrease when algae and plants are actively growing and removing nitrogen as part of their growth.

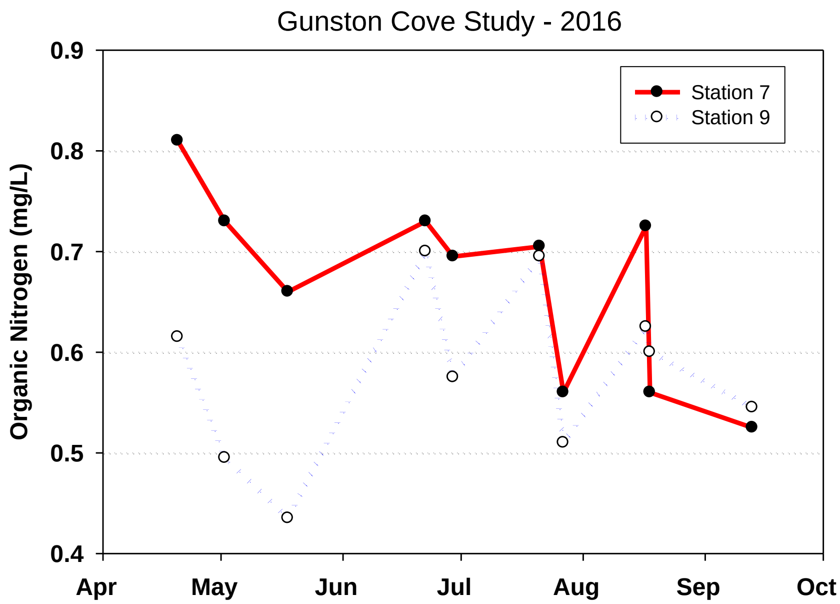
Figure 18. Nitrate Nitrogen (mg/L). Fairfax County Lab Data. Month tick is at first day of month. (Limit of detection: 0.01 mg/L; LD values graphed as 0.005 mg/L)

Nitrate nitrogen levels were highest at both sites in early spring and declined through the year (Figure 18). The decline was much quicker in the cove. This decline corresponded to the upswing in phytoplankton and SAV and was probably due to algal and SAV uptake. Nitrite nitrogen remained low throughout the year, often being below the limit of detection in the cove, but being consistently somewhat higher in the river (Figure 19). One exceptionally high value was reported in early July in the river.



Nitrite nitrogen consists of nitrogen in the form of nitrite ion (NO_2^-). Nitrite is an intermediate in the oxidation of ammonia to nitrate, a process called nitrification. Nitrite is usually in very low concentrations unless there is active nitrification.

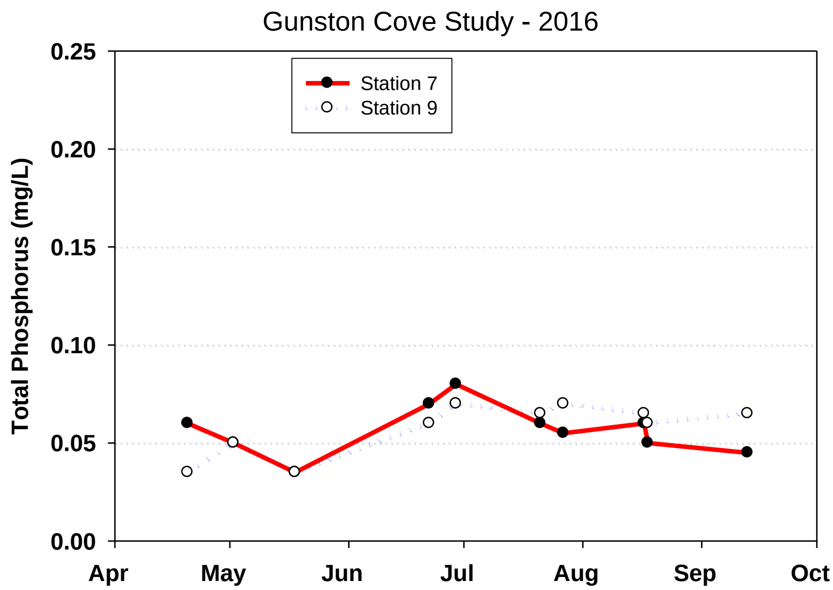
Figure 19. Nitrite Nitrogen (mg/L). Fairfax County Lab Data. Month tick is at first day of month. (limit of detection = 0.01 mg/L).



Organic nitrogen measures the nitrogen in dissolved and particulate organic compounds in the water. Organic nitrogen comprises algal and bacterial cells, detritus (particles of decaying plant, microbial, and animal matter), amino acids, urea, and small proteins. When broken down in the environment, organic nitrogen results in ammonia nitrogen. Organic nitrogen is determined as the difference between total Kjeldahl nitrogen and ammonia nitrogen.

Figure 20. Organic Nitrogen (mg/L). Fairfax County Lab Data. Month tick is at first day of month.

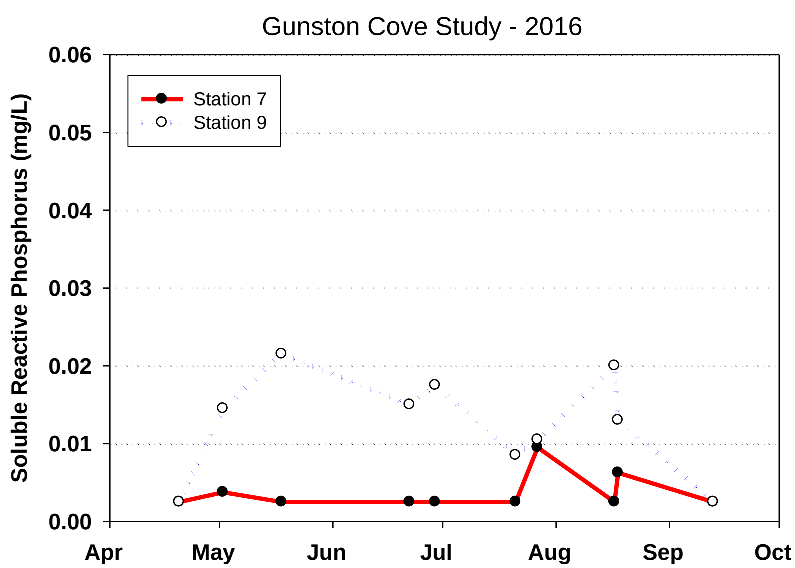
Organic nitrogen was highest in the cove in the spring and similar at both sites from late June through September (Figure 20).



Phosphorus (P) is often the limiting nutrient in freshwater ecosystems. As such the concentration of P can set the upper limit for algal growth. Total phosphorus is the best measure of P availability in freshwater since much of the P is tied up in biological tissue such as algal cells. Total P includes phosphate ion (PO_4^{-3}) as well as phosphate inside cells and phosphate bound to inorganic particles such as clays.

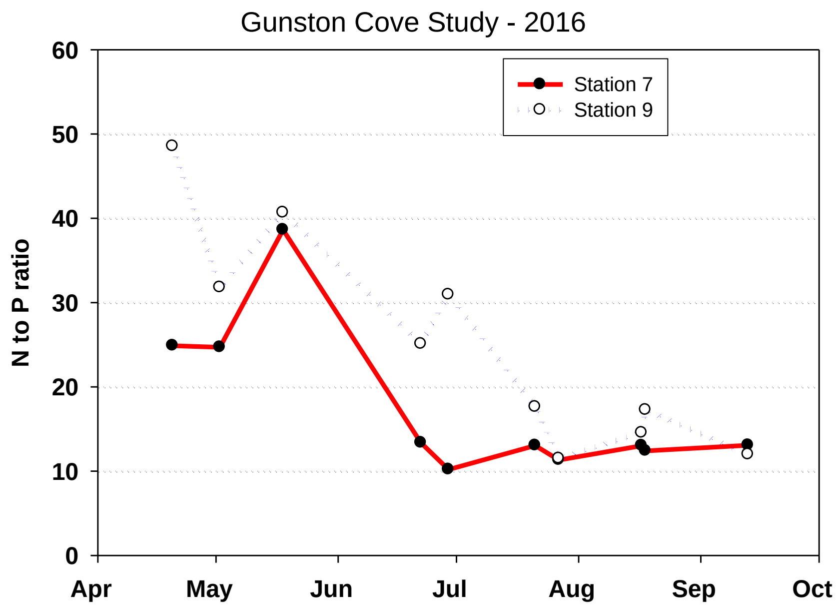
Figure 21. Total Phosphorus (mg/L). Fairfax County Lab Data. Month tick is at first day of month. (Limit of detection: 0.03 mg/L)

Total phosphorus was similar at both sites on almost all dates and did not show much seasonal variation (Figure 21). Soluble reactive phosphorus was consistently higher in the river while being quite low in almost all cove samples (Figure 22).



Soluble reactive phosphorus (SRP) is a measure of phosphate ion (PO_4^{-3}). Phosphate ion is the form in which P is most available to primary producers such as algae and aquatic plants in freshwater. However, SRP is often inversely related to the activity of primary producers because they tend to take it up so rapidly. So, higher levels of SRP indicate either a local source of SRP to the waterbody or limitation by a factor other than P.

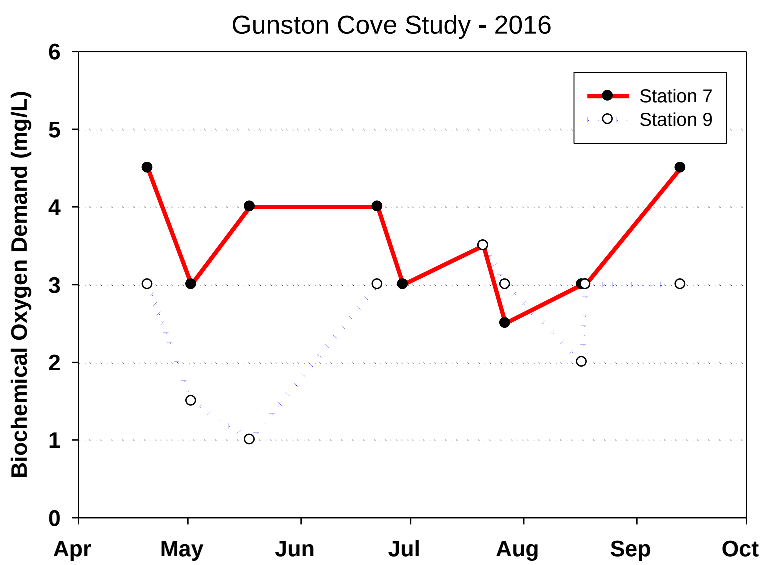
Figure 22. Soluble Reactive Phosphorus (mg/L). Fairfax County Lab Data. Month tick is at first day of month. (Limit of detection = 0.005 mg/L)



N:P ratio is determined by summing all of the components of N (ammonia, nitrate, nitrite, and organic nitrogen) and dividing by total P. This ratio gives an indication of whether N or P is more likely to be limiting primary production in a given freshwater system. Generally, values above 7.2 are considered indicative of P limitation while values below 7.2 suggest N limitation. N limitation could lead to dominance by cyanobacteria who can fix their own N from the atmosphere.

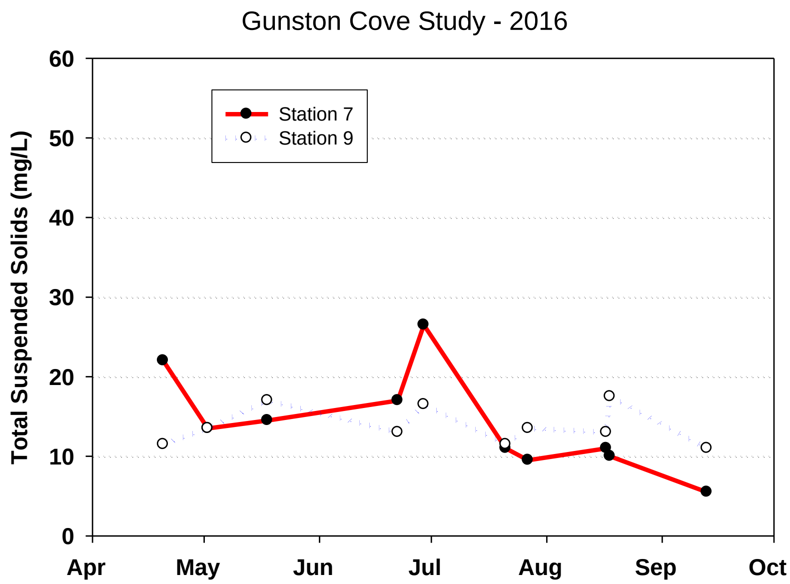
Figure 23. N/P Ratio (by mass). Fairfax County Lab Data. Month tick is at first day of month.

N/P ratio exhibited a distinct seasonal decline at both sites (Figure 23). Values bottomed out at about 10 in late June in the cove and remained there for the rest of the year. Values in the river reached this level in late July. Biochemical oxygen demand (BOD) was consistently higher in the cove than in the river (Figure 24). Values in the cove did not show much seasonal change, but exhibited a minimum in May in the river.



Biochemical oxygen demand (BOD) measures the amount of decomposable organic matter in the water as a function of how much oxygen it consumes as it breaks down over a given number of days. Most commonly the number of days used is 5. BOD is a good indicator of the potential for oxygen depletion in water. BOD is composed both dissolved organic compounds in the water as well as microbes such as bacteria and algae which will respire and consume oxygen during the period of measurement.

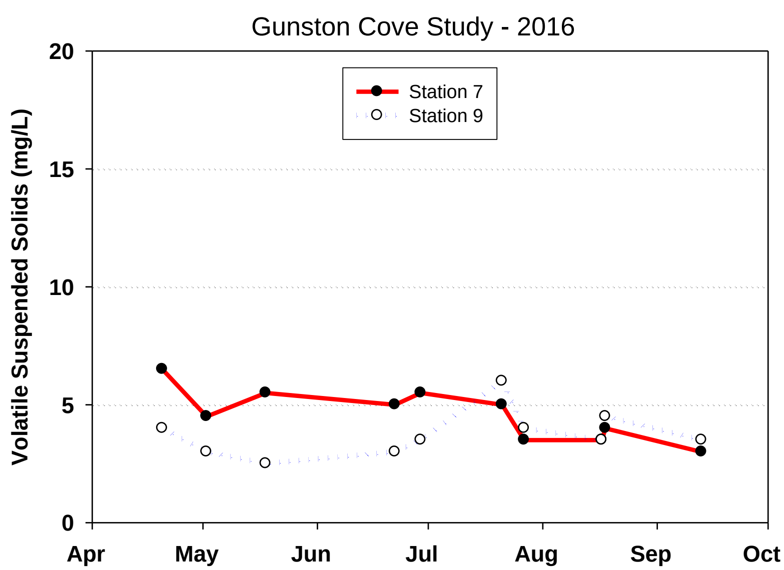
Figure 24. Biochemical Oxygen Demand (mg/L). Fairfax County Lab Data. Month tick is at first day of month.



Total suspended solids (TSS) is measured by filtering a known amount of water through a fine filter which retains all or virtually all particles in the water. This filter is then dried and the weight of particles on the filter determined by difference. TSS consists of both organic and inorganic particles. During periods of low river and tributary inflow, organic particles such as algae may dominate. During storm flow periods or heavy winds causing resuspension, inorganic particles may dominate.

Figure 25. Total Suspended Solids (mg/L). Fairfax County Lab Data. Month tick is at first day of month.

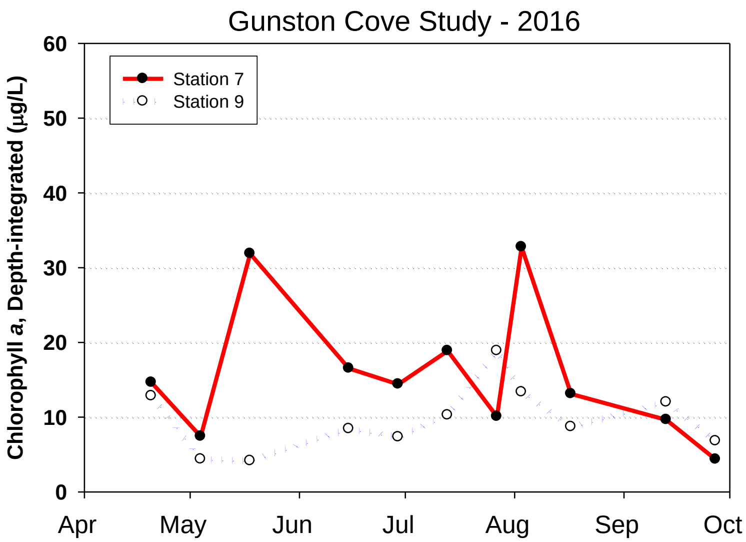
Total suspended solids was generally in the range 10-20 mg/L at both stations (Figure 25). There was little seasonal pattern, but cove values did spike in early July and reached very low values in September. Volatile suspended solids was generally higher in the cove with little seasonal pattern (Figure 26).



Volatile suspended solids (VSS) is determined by taking the filters used for TSS and then ashing them to combust (volatilize) the organic matter. The organic component is then determined by difference. VSS is a measure of organic solids in a water sample. These organic solids could be bacteria, algae, or detritus. Origins include sewage effluent, algae growth in the water column, or detritus produced within the waterbody or from tributaries. In summer in Gunston Cove a chief source is algal (phytoplankton) growth.

Figure 26. Volatile Suspended Solids (mg/L). Fairfax County Lab Data. Month tick is at first day of month.

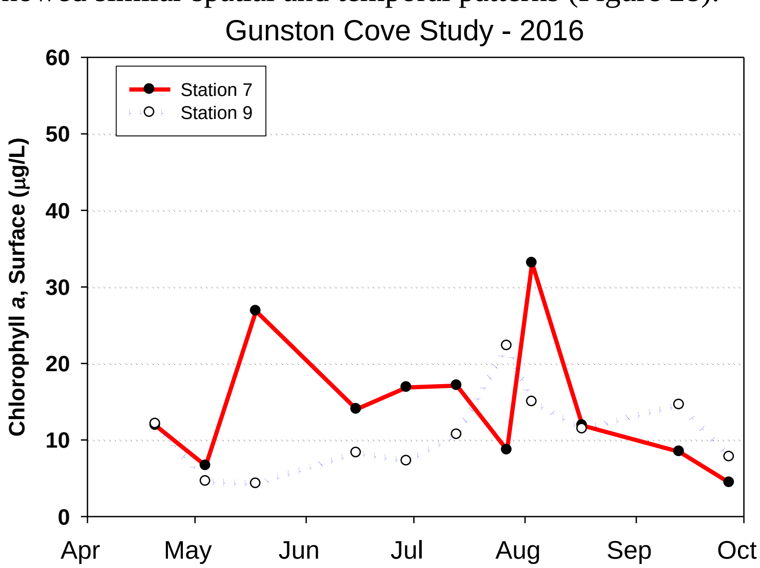
C. Phytoplankton -2016



Chlorophyll *a* is a measure of the amount of algae growing in the water column. These suspended algae are called phytoplankton, meaning "plant wanderers". In addition to the true algae (greens, diatoms, cryptophytes, etc.) the term phytoplankton includes cyanobacteria (sometimes known as "blue-green" algae). Both depth-integrated and surface chlorophyll values are measured due to the capacity of phytoplankton to aggregate near the surface under certain conditions.

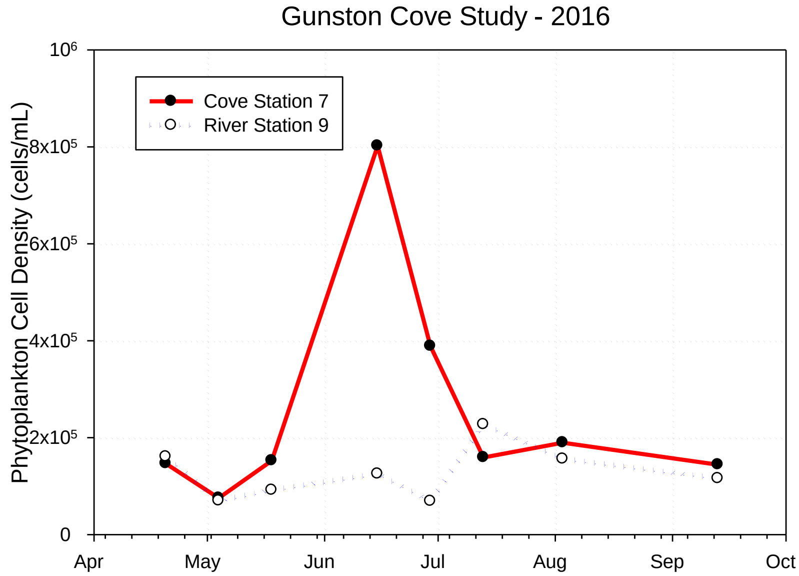
Figure 27. Chlorophyll *a* (ug/L). Depth-integrated. GMU Lab Data. Month tick is at the first day of month.

Chlorophyll *a* in the cove displayed a distinct seasonal pattern in 2016 (Figure 27). A decline in early May was probably due to flushing by high flows. A marked increase was observed in late May under favorable growing conditions followed by a slow decline in June and July and another peak in early August. In the river chlorophyll values were lower showing maxima in April and late July. Depth-integrated and surface chlorophyll showed similar spatial and temporal patterns (Figure 28).



In the Gunston Cove, there is very little difference in surface and depth-integrated chlorophyll levels because tidal action keeps the water well-mixed which overcomes any potential surface aggregation by the phytoplankton. Summer chlorophyll concentrations above 30 ug/L are generally considered characteristic or eutrophic conditions.

Figure 28. Chlorophyll *a* (ug/L). Surface. GMU Lab Data. Month tick is at first day of month.



Phytoplankton cell density provides a measure of the number of algal cells per unit volume. This is a rough measure of the abundance of phytoplankton, but does not discriminate between large and small cells. Therefore, a large number of small cells may actually represent less biomass (weight of living tissue) than a smaller number of large cells. However, small cells are typically more active than larger ones so cell density is probably a better indicator of activity than of biomass. The smaller cells are mostly cyanobacteria.

Figure 29. Phytoplankton Density (cells/mL).

In the cove phytoplankton density exhibited a strong peak in early June (Figure 29). In the river there was an increase from April through the end of July. The river showed a much less distinct seasonal pattern with values near that of the cove except in June. Total biovolume at both stations showed a distinct drop in early May. During May and June, cove values were distinctly higher than in the river. This was the general time when chlorophyll values were elevated in the cove relative to the river (Figure 30).

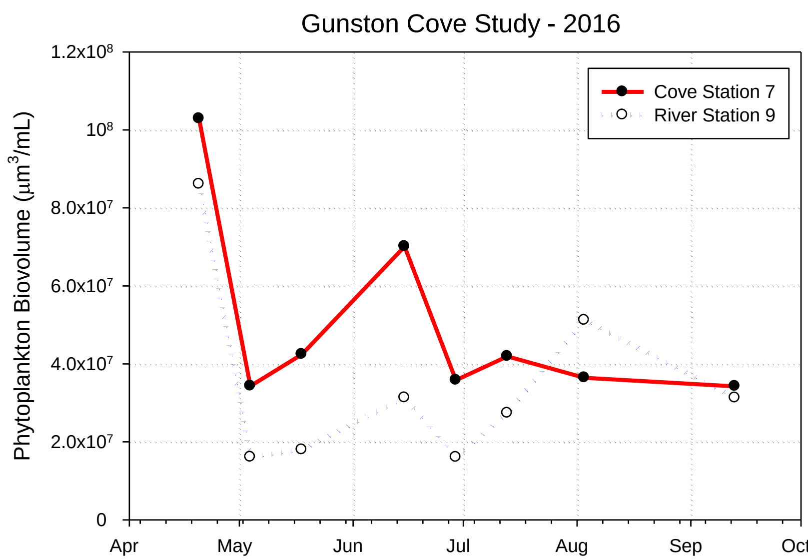
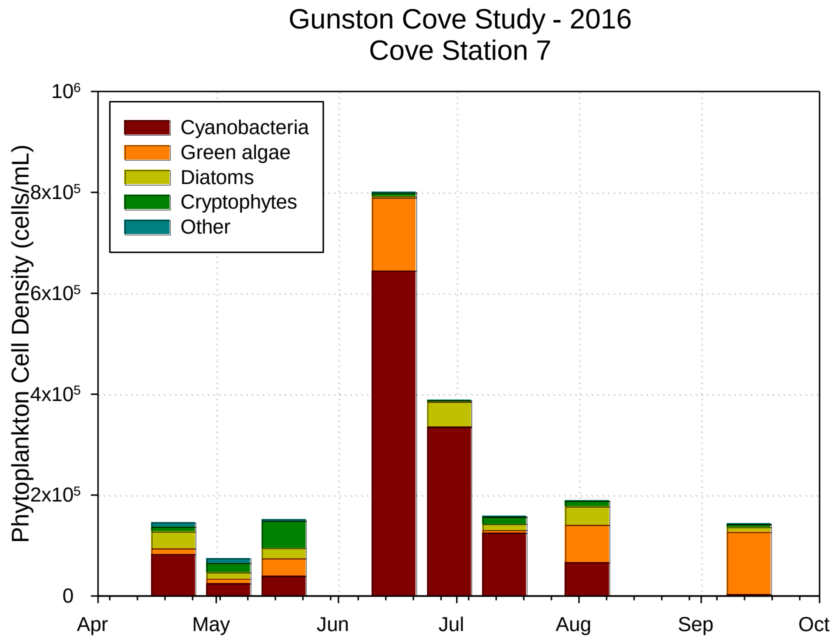


Figure 30. Phytoplankton Biovolume (um^3/mL).

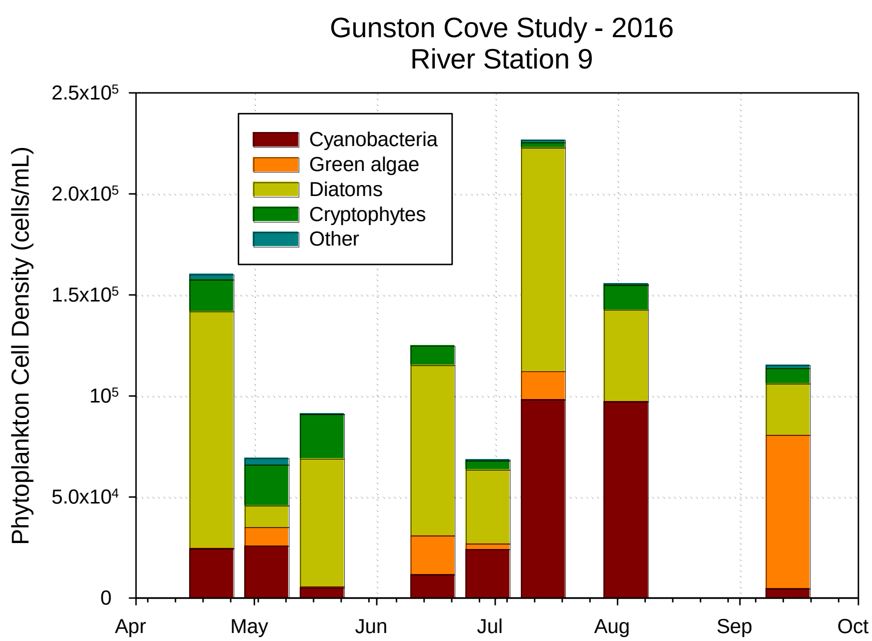
The volume of individual cells of each species is determined by approximating the cells of each species to an appropriate geometric shape (e.g. sphere, cylinder, cone, etc.) and then making the measurements of the appropriate dimensions under the microscope. Total phytoplankton biovolume (shown here) is determined by multiplying the cell density of each species by the biovolume of each cell of that species. Biovolume accounts for the differing size of various phytoplankton cells and is probably a better measure of biomass. However, it does not account for the varying amount of water and other nonliving constituents in cells.



Total phytoplankton cell density can be broken down by major group. The top four groups represent those which are generally most abundant. **Other** includes euglenoids and dinoflagellates. Due to their small size cyanobacteria typically dominate cell density numbers. Their numbers are typically highest in the late summer reflecting an accumulation of cells during favorable summer growing conditions.

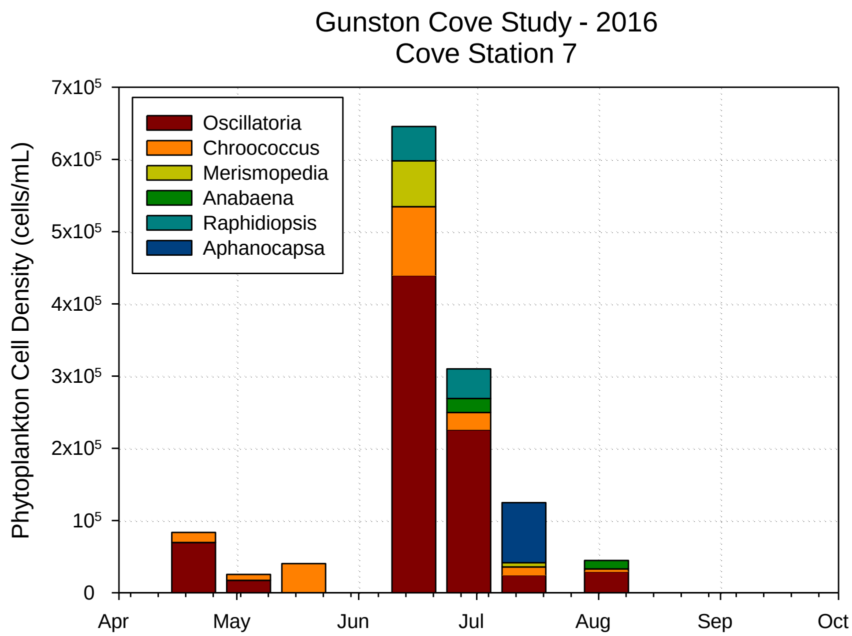
Figure 31. Phytoplankton Density by Major Group (cells/mL). Gunston Cove.

Phytoplankton density in the cove was dominated by cyanobacteria during most of the year (Figure 31). In the river diatoms were clearly most numerous on most dates with cyanobacteria important in July and August and green algae dominant in September (Figure 32). Due to their small size, cyanobacteria usually are often the most abundant group, but do not necessarily represent the greatest biomass.



In the river cyanobacteria normally follow similar patterns as in the cove, but attaining lower abundances. This is probably due to the deeper water column which leads to lower effective light levels and greater mixing. Other groups such as diatoms and green algae tend to be more important on a relative basis than in the cove.

Figure 32. Phytoplankton Density by Major Group (cells/mL). River.



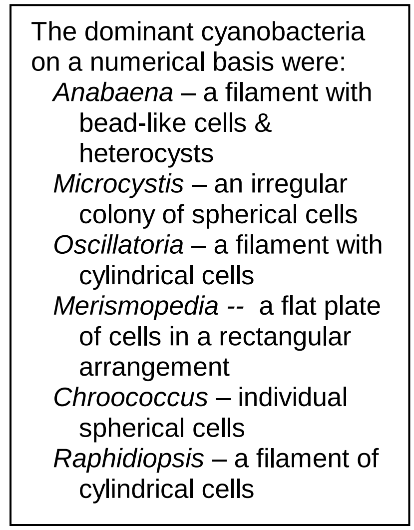
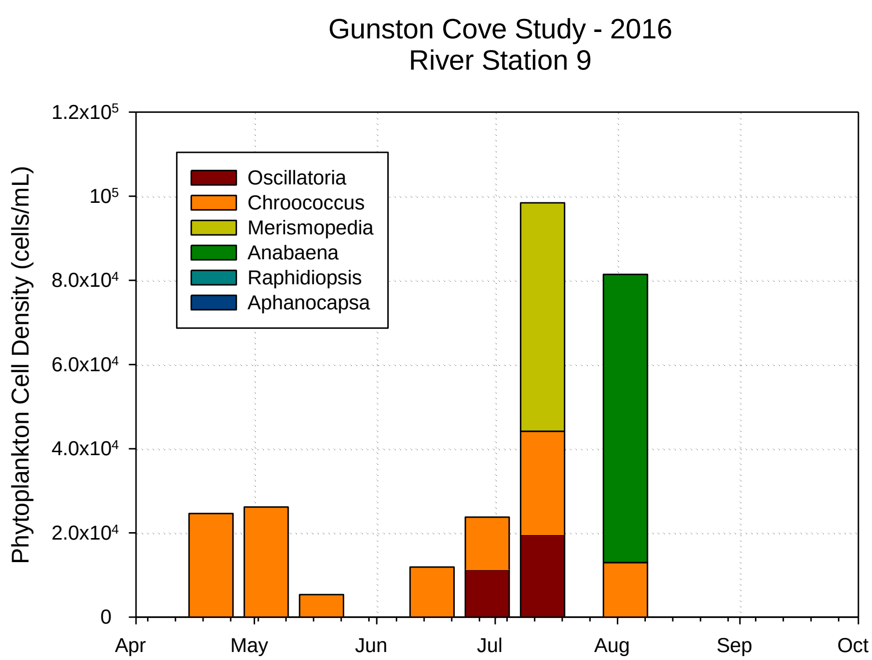


Figure 33. Phytoplankton Density by Dominant Cyanobacteria (cells/mL). Gunston Cove.

Oscillatoria was the most abundant cyanobacterium on most dates (Figure 33). *Chroococcus* was a substantial contributor on most dates. In the river *Oscillatoria* was much less prominent and *Chroococcus* was typically most abundant (Figure 34). *Anabaena* made a strong showing in early August.



Anabaena

Microcystis

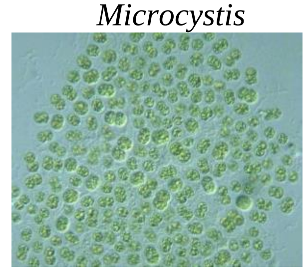
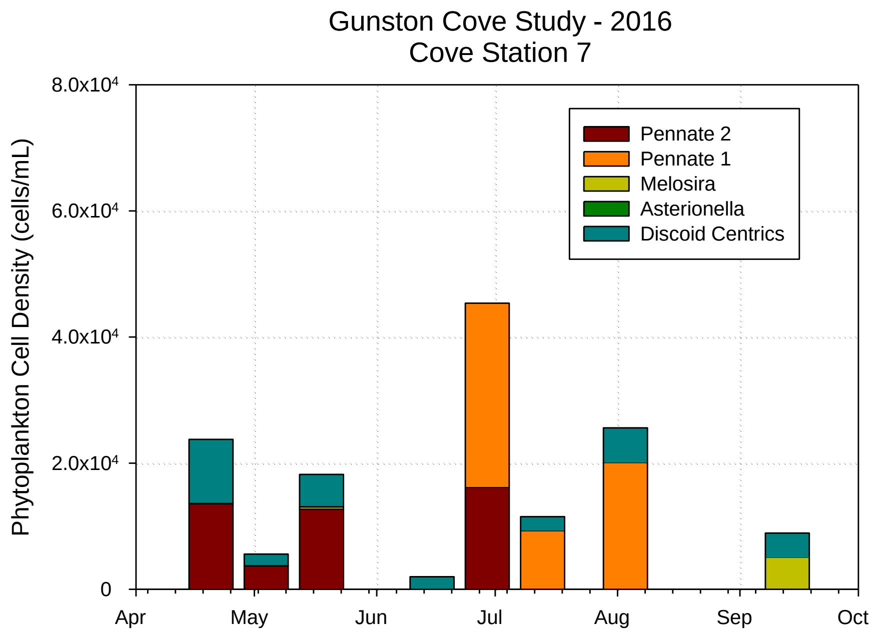


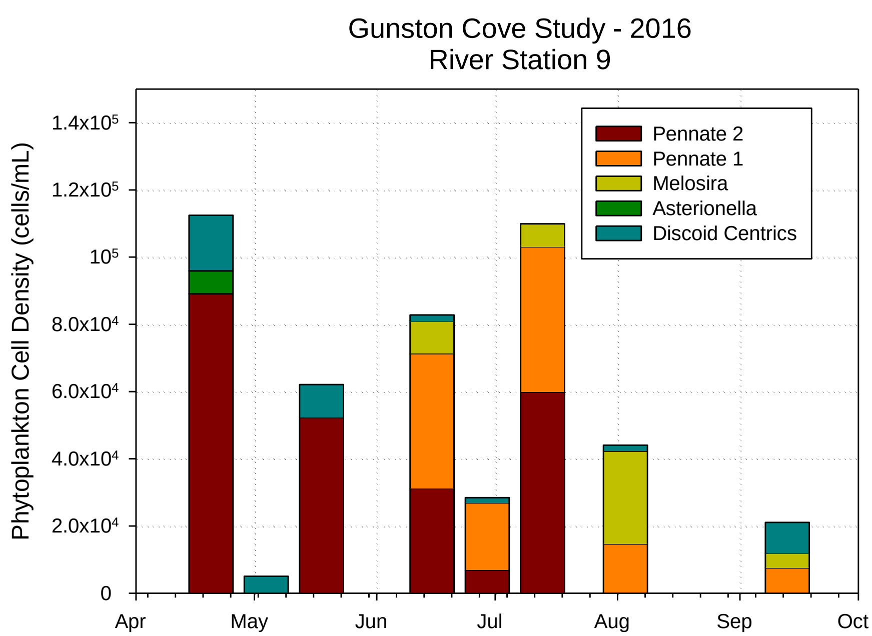
Figure 34. Phytoplankton Density by Dominant Cyanobacteria (cells/mL). River.



The most numerous diatoms in the phytoplankton were:
Melosira – a filamentous centric diatom
Centrics – discoid centric diatoms
Pennate 2 – a small pinnate diatom
Pennate 1 – a small pinnate diatom
Asterionella – a small pennate diatom that forms spoked - wheel colony

Figure 35. Phytoplankton Density by Dominant Diatoms (cells/mL). Gunston Cove.

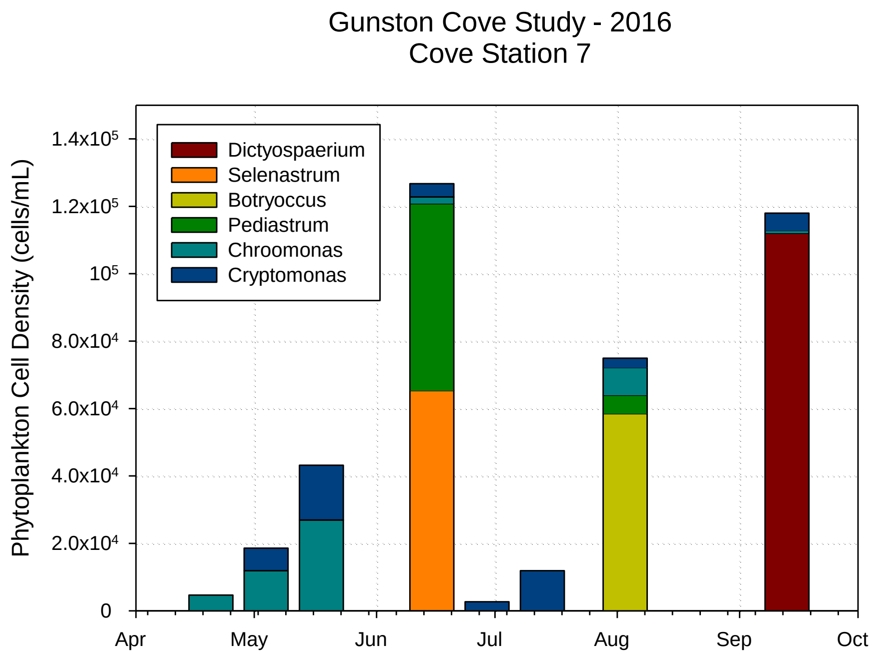
Diatom cell density was dominated by Pennate 2 in spring and Pennate 1 in summer. Discoid centrics were also prominent in most cove samples (Figure 35). In the river a similar pattern was observed with the addition of substantial numbers of *Melosira* in summer and fall (Figure 36).



Melosira

Discoid Centrics

Figure 36. Phytoplankton Density by Dominant Diatoms (cells/mL). River.

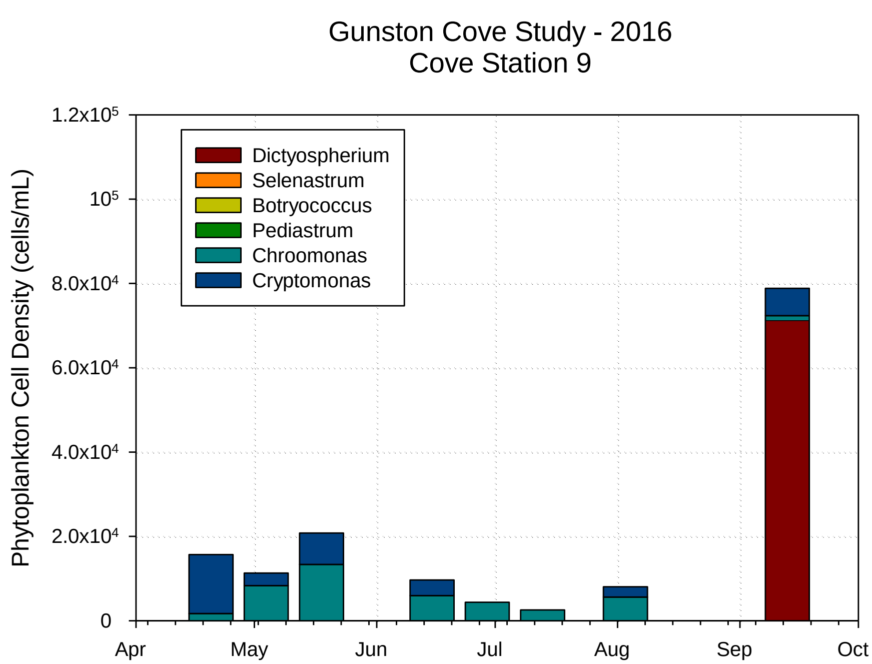


The most numerous phytoplankton among the cryptophytes, green algae and others were:

- *Cryptomonas* – an ellipsoidal, flagellated unicell
- *Chroomonas* – a flagellated cryptomonad unicell
- *Dictyosphaerium* – colony of small green unicells
- *Selenastrum* – single green algal cell as curved rod
- *Pediastrum* – green algal colony
- *Botryococcus* – single green coccoid cell

Figure 37. Phytoplankton Density (#/mL) by Dominant Other Taxa. Gunston Cove.

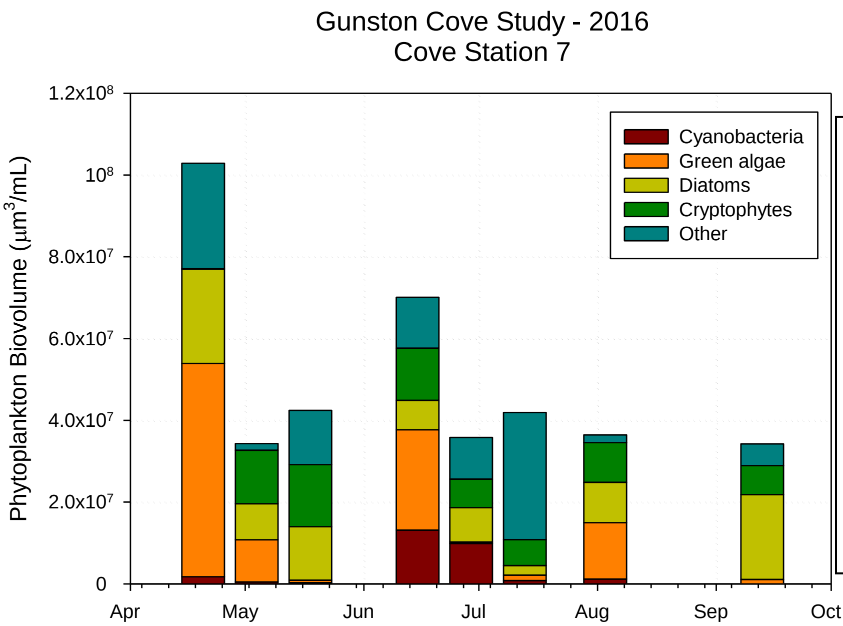
In the cove numerous other taxa were important and there was a lot of variation between dates (Figure 37). *Cryptomonas* and *Chroomonas* were generally the most abundant other taxa in spring and early summer with a large number of *Dictyospherium* in September at both sites. In the cove there were large numbers of *Selenastrum*, *Pediastrum*, and *Botryococcus* on certain dates (Figure 38).




Cryptomonas


Scenedesmus

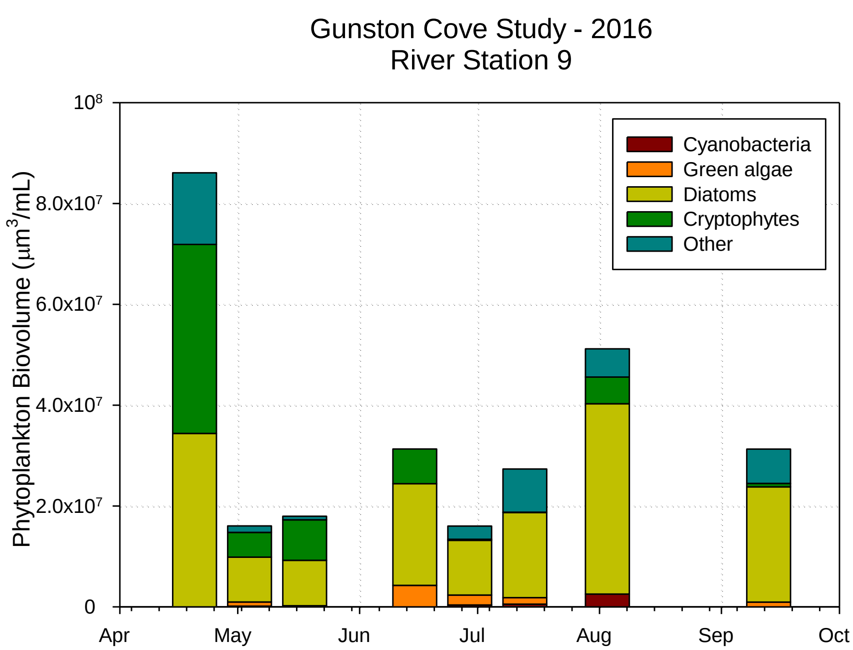
Figure 38. Phytoplankton Density (#/mL) by Dominant Other Taxa. River.



Total phytoplankton biovolume can be broken down into groups:
- **Cyanobacteria** ("blue-green" algae)
- **Green algae**
- **Diatoms** – includes both centric and pinnate
- **Cryptophytes**
- **Other** – includes euglenoids, chrysophytes, and dinoflagellates

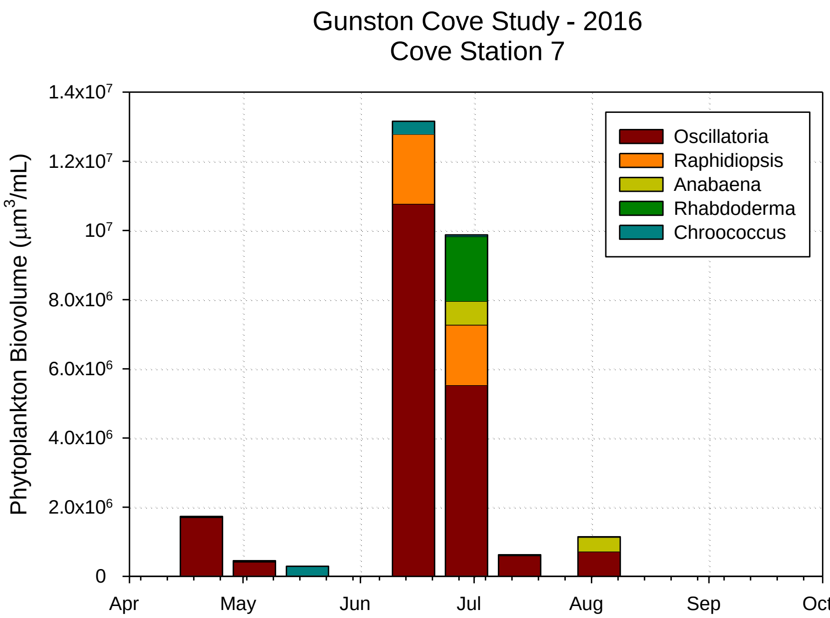
Figure 39. Phytoplankton Biovolume (um^3/mL) by Major Groups. Gunston Cove.

In the cove biovolume dominance was variable and often multiple groups had a significant presence (Figure 39). Green algae and diatoms were the most important on most dates. Cryptophytes and other algae were also prominent. Cyanobacteria were generally at low levels. In the river, diatoms were dominant in biovolume for most of the year with cryptophytes also important on most dates (Figure 40).



While dominating cell density, cyanobacteria typically make up a much smaller portion of phytoplankton biovolume. As with cell density, biovolume is generally greater in the cove.

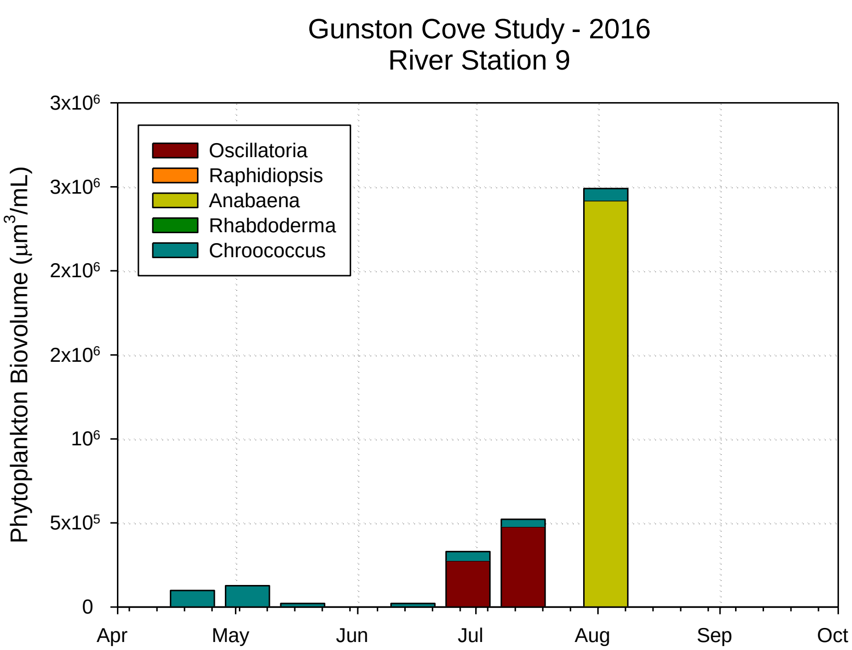
Figure 40. Phytoplankton Biovolume (um^3/mL) by Major Groups. River.



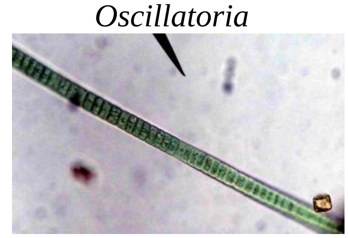
The dominant cyanobacteria on a biovolume basis were:
- *Anabaena* – a filament with bead-like cells & heterocysts
- *Rhabdoderma* – rod-shaped cells in small packets
- *Oscillatoria* – a filament with cylindrical cells
- *Chroococcus* – individual spherical cells
- *Raphidiopsis* – a filament of cylindrical cells

Figure 41. Phytoplankton Biovolume (um^3/mL) by Cyanobacteria Taxa. Gunston Cove.

Oscillatoria accounted for almost all of the cyanobacterial biovolume in the cove (Figure 41). *Raphiopsis* and *Rhabdoderma* were of importance in mid summer. In the river cyanobacteria were much less abundant with the only major event being a peak of *Anabaena* in early August (Figure 42).



Oscillatoria



Chroococcus

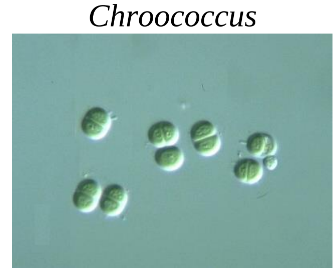
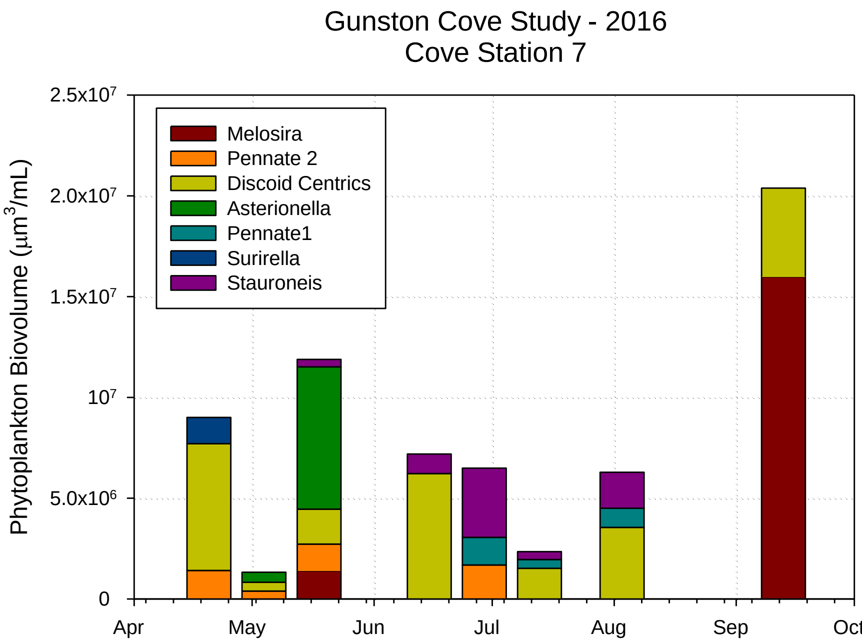


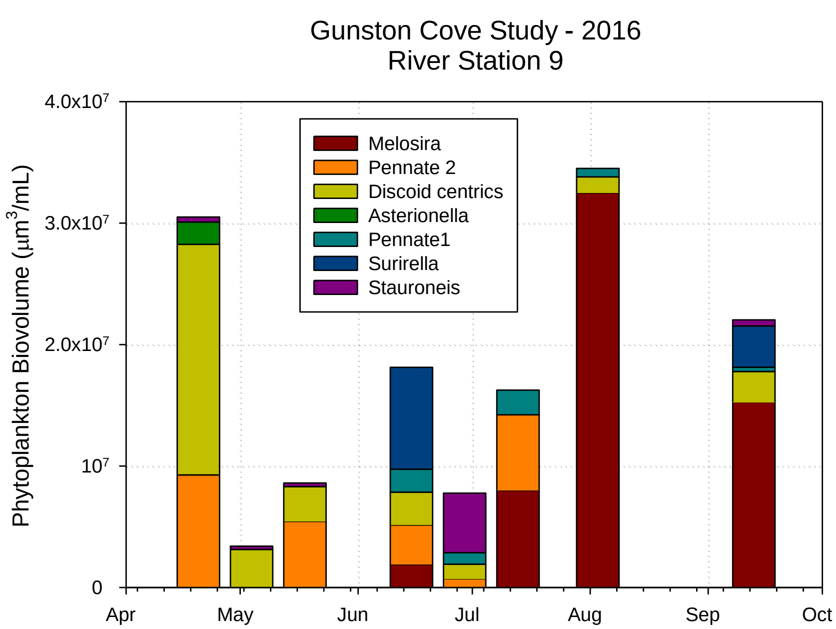
Figure 42. Phytoplankton Biovolume (um^3/mL) by Cyanobacterial Taxa. River.



The most numerous diatoms in the phytoplankton were:
Melosira – a filamentous centric diatom
Centrics – discoid centric diatoms
Stauroneis – larger pennate diatom
Pennate 2 – a small pinnate diatom
Pennate 1 – a small pinnate diatom
Surirella – a larger pennate diatom
Nitzschia – a small pennate diatom

Figure 43. Phytoplankton Biovolume (um^3/mL) by Diatom Taxa. Gunston Cove.

In the cove discoid centrics were dominant or made a significant showing on almost all dates (Figure 43). *Asterionella* was very abundant in late May. And in September, *Melosira*, usually the dominant all year, exhibited a strong peak in abundance. In the river discoid centrics and Pennate 2 were dominant in spring. *Melosira* came on in July and dominated in the river for the rest of the year (Figure 44).



Melosira

Surirella

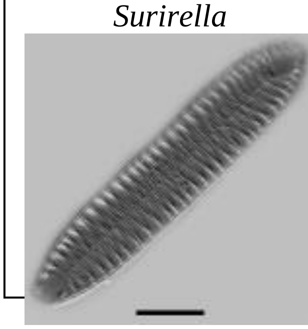
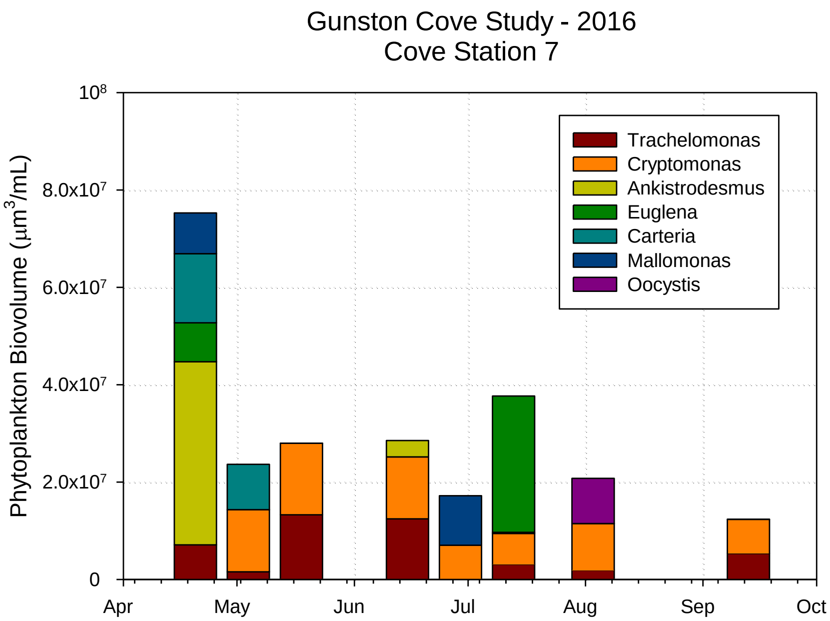


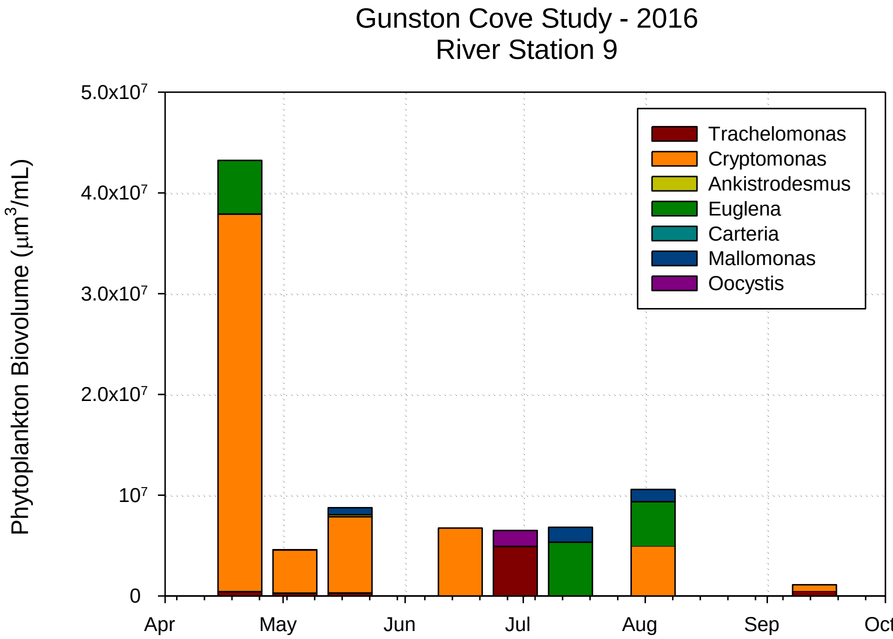
Figure 44. Phytoplankton Biovolume (um^3/mL) by Diatom Taxa. River.



The most numerous phyto-plankton among the cryptophytes, green algae and others were:
- *Euglena* – large euglenoid flagellate
- *Cryptomonas* – an ellipsoidal, flagellated unicell
- *Carteria* – flagellated green unicell
- *Mallomonas* – unicellular scaled flagellate
- *Trachelomonas* – spherical, armored euglenoid
- *Ankistrodesmus* – rod-like single celled green alga
- *Oocystis* –green unicells in small packets

Figure 45. Phytoplankton Biovolume (um^3/mL) by Dominant Other Taxa. Gunston Cove.

A number of other taxa were present in 2016 and some made strong contributions to biovolume (Figure 45). *Ankistrodesmus* was dominant in April and Euglena in July. For the remaining samples *Cryptomonas* and *Trachelomonas* were consistently dominant in the cove. In the river *Cryptomonas* was almost always strongly dominant with *Euglena* showing a marked presence as well on several dates (Figure 46).

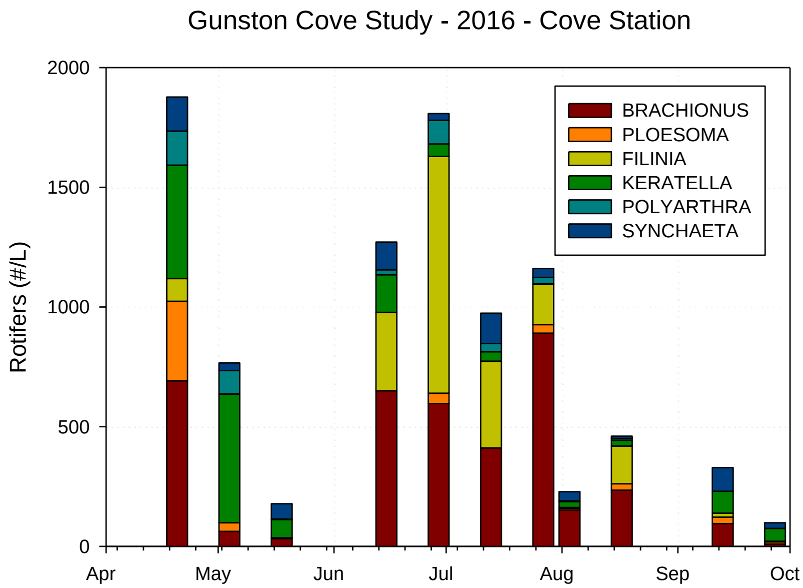



Cryptomonas

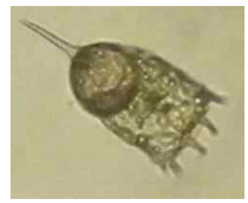
Euglena


Figure 46. Phytoplankton Biovolume (um^3/mL) by Dominant Other Taxa. River.

D. Zooplankton – 2016



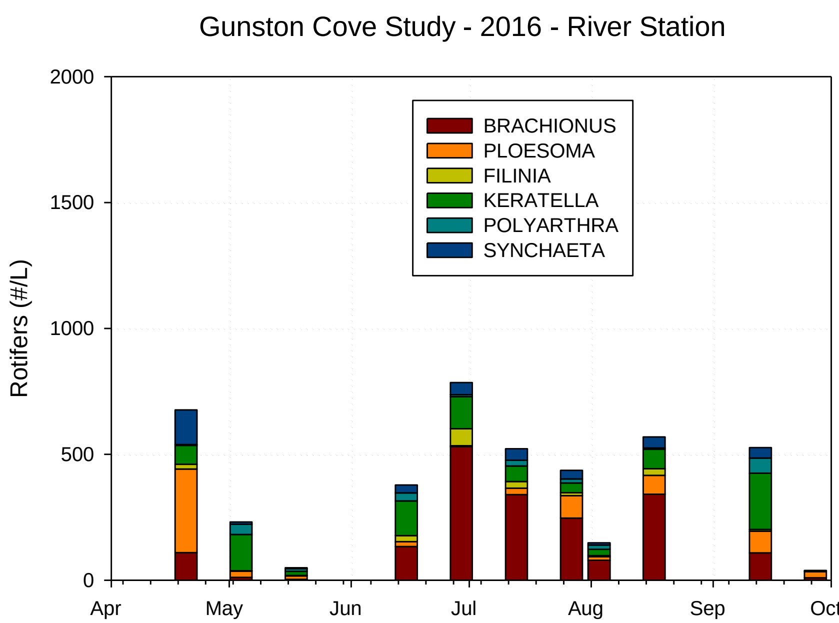
Brachionus (Sta 7, RCJ)



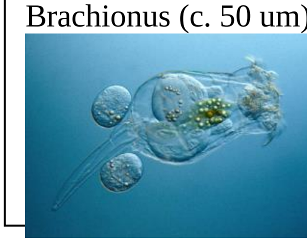
Keratella (Sta 7, RCJ)

Figure 47. Rotifer Density by Dominant Taxa (#/L). Cove.

In the cove, rotifers exhibited a strong presence in April, declined in May, and then increased again in June to an early summer peak (Figure 47). A gradual decline was observed for the late summer into the fall. *Brachionus* and Keratella were most prominent in spring while *Brachionus* and *Filinia* were most dominant for the remainder of the year. In the river rotifers exhibited a similar seasonal pattern at reduced levels (Figure 48). *Brachionus* and Conochilidae were most important in the spring and *Keratella* was co-dominant for the rest of the year.



Brachionus (c. 50 um)



Conochilidae

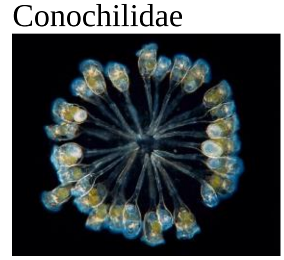
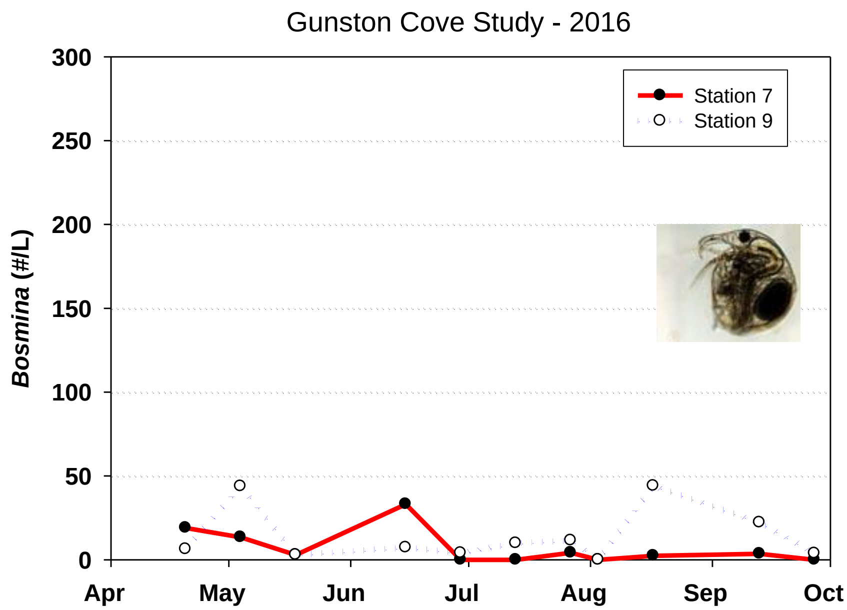


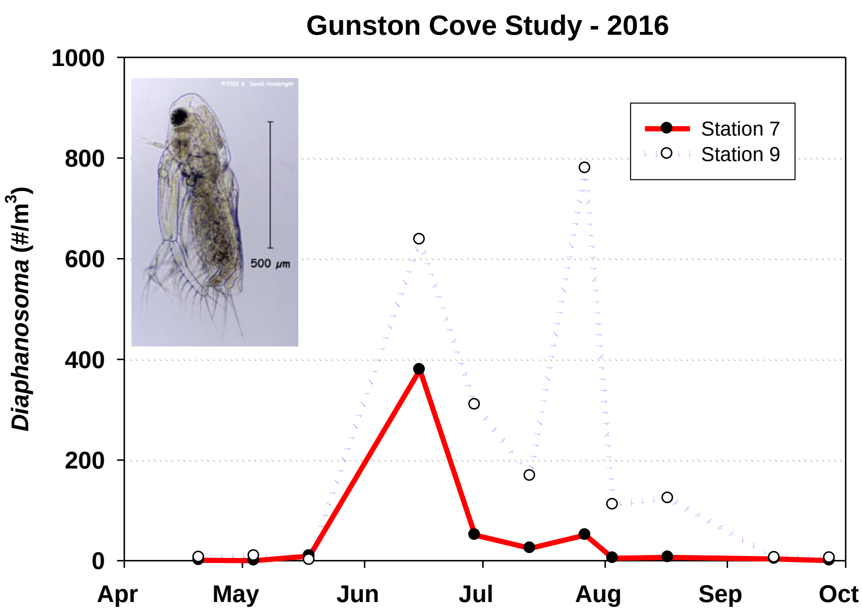
Figure 48. Rotifer Density by Dominant Taxa (#/L). River.



Bosmina is a small-bodied cladoceran, or "waterflea", which is common in lakes and freshwater tidal areas. It is typically the most abundant cladoceran with maximum numbers generally about 100-1000 animals per liter. Due to its small size and relatively high abundances, it is enumerated in the micro-zooplankton samples. *Bosmina* can graze on smaller phytoplankton cells, but can also utilize some cells from colonies by knocking them loose.

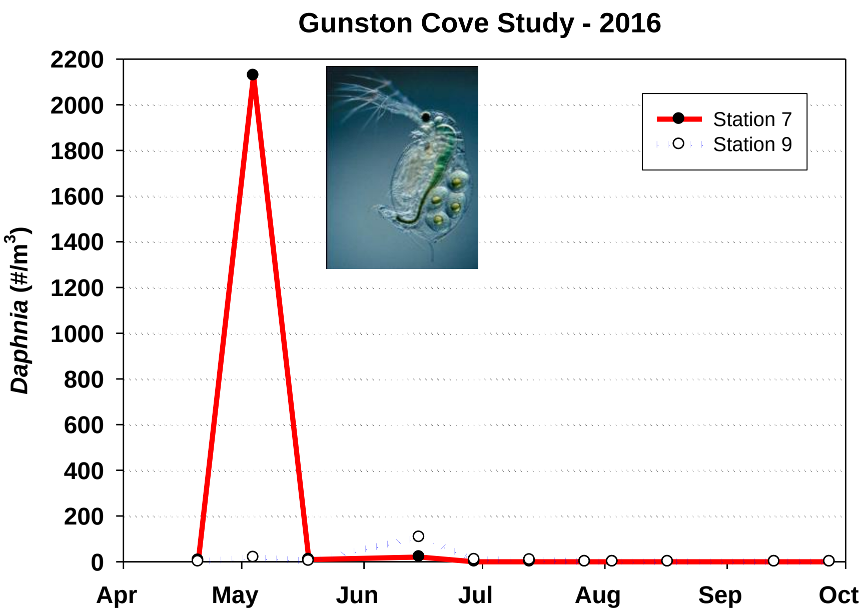
Figure 49. *Bosmina* Density by Station (#/L).

In 2016 the small cladoceran *Bosmina* was present in many samples, but at lower than normal levels at both sites (Figure 49). *Diaphanosoma*, typically the most abundant larger cladoceran in the study area, was present at appreciable levels in 2016, reaching a maximum of several hundred per m^3 in June at both stations (Figure 50). A second, somewhat higher peak of about 800/m^3 was observed in late June in the river.



Diaphanosoma is the most abundant larger cladoceran found in the tidal Potomac River. It generally reaches numbers of 1,000-10,000 per m^3 (which would be 1-10 per liter). Due to their larger size and lower abundances, *Diaphanosoma* and the other cladocera are enumerated in the macrozooplankton samples. *Diaphanosoma* prefers warmer temperatures than some cladocera and is often common in the summer.

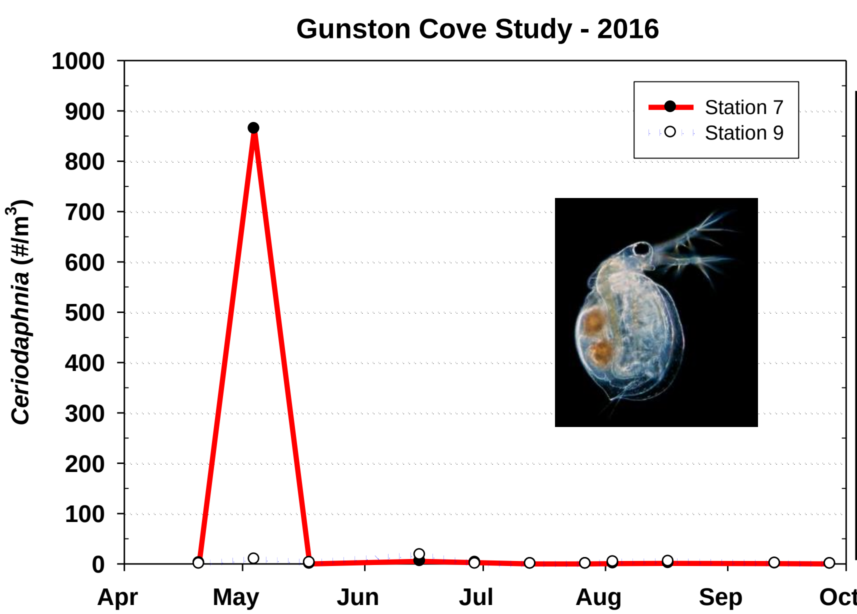
Figure 50. *Diaphanosoma* Density by Station (#/m^3).



Daphnia, the common waterflea, is one of the most efficient grazers of phytoplankton in freshwater ecosystems. In the tidal Potomac River it is present, but has not generally been as abundant as *Diaphanosoma*. It is typically most common in spring.

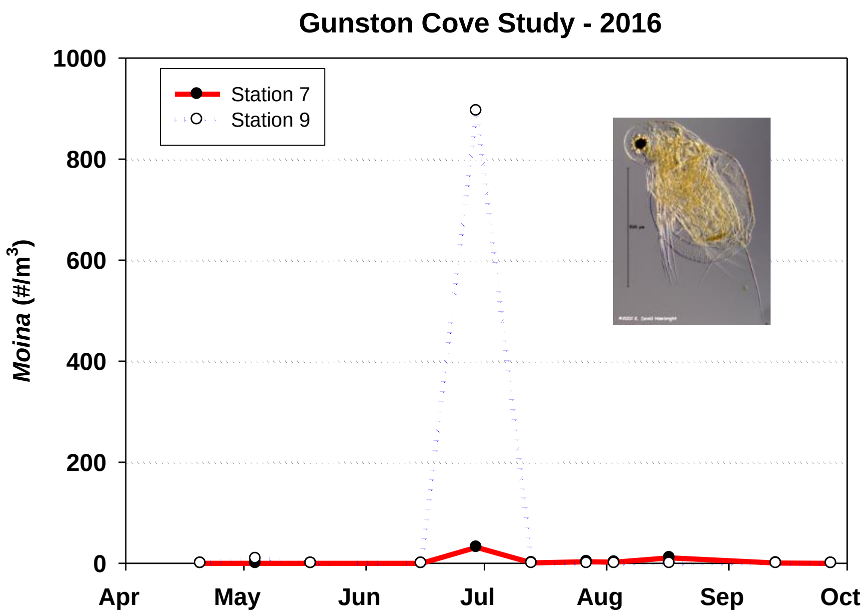
Figure 51. *Daphnia* Density by Station (#/m^3).

In 2016 *Daphnia* exhibited one high abundance sample – early May in the cove when it reached unusually high values of nearly 2000/m^3 (Figure 51). In the river, *Daphina* abundance was very limited. *Ceriodaphnia* exhibited a similar pattern at a lower level reaching a peak of about 850/m^3 in the cove and little presence in the river (Figure 52).



Ceriodaphnia, another common large-bodied cladoceran, is usually present in numbers similar to *Daphnia*. Like all waterfleas, the juveniles look like miniature adults and grow through a series of molts to a larger size and finally reach reproductive maturity. Most reproduction is asexual except during stressful environmental conditions.

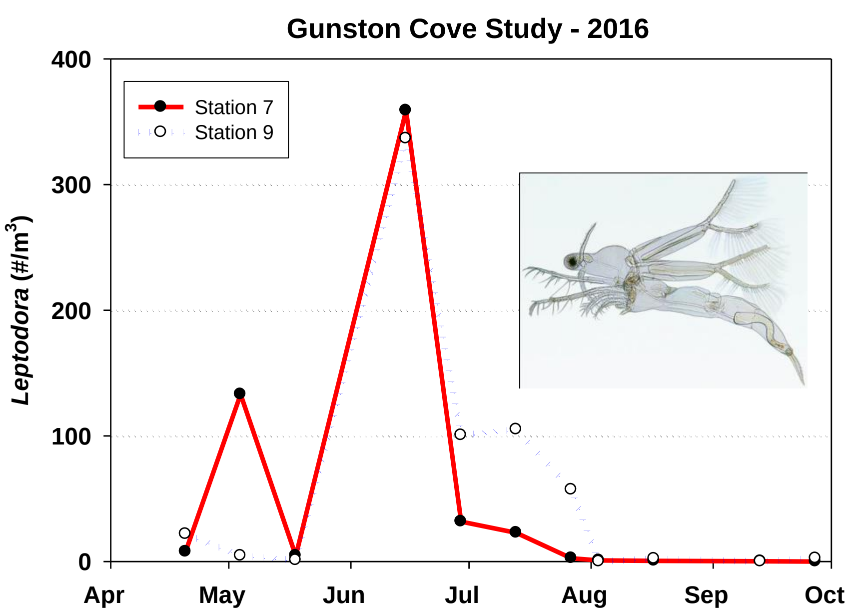
Figure 52. *Ceriodaphnia* Density by Station (#/m^3).



Moina is another waterflea that is often observed in the tidal Potomac River. Like the other cladocera mentioned so far, *Moina* grazes on phytoplankton to obtain its food supply.

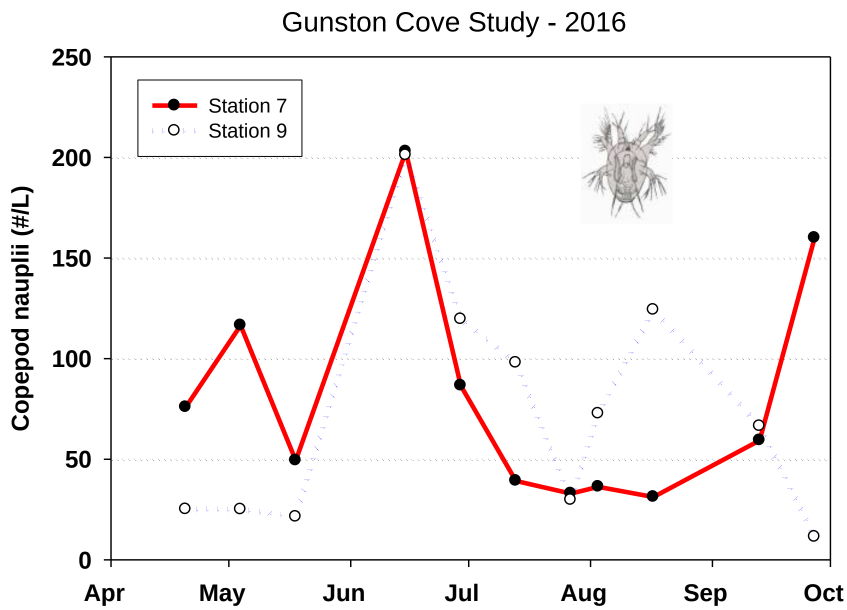
Figure 53. *Moina* Density by Station (#/m^3).

Moina, another medium-sized cladoceran showed different timing and location (Figure 53). Its peak was in the river and in late June. Again, it was a short-lived peak, but attained a substantial density of 900/m^3. *Leptodora*, the large cladoceran predator, was present in many samples at both stations (Figure 54). Both stations showed peak values of about 350/m^3 in June. In addition, the cove station exhibited a peak in early May.



Leptodora is substantially larger than the other cladocera mentioned. Also different is its mode of feeding – it is a predator on other zooplankton. It normally occurs for brief periods in the late spring or early summer.

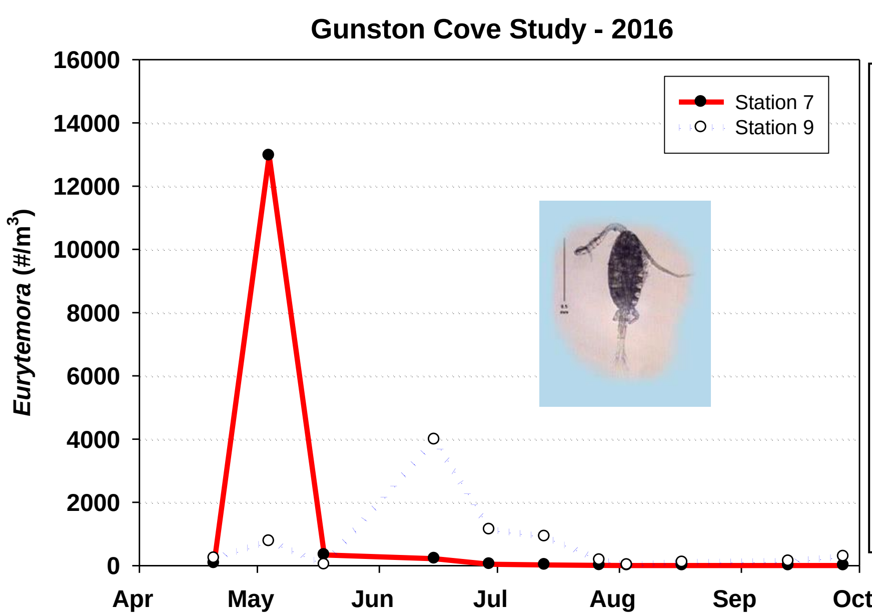
Figure 54. *Leptodora* Density by Station (#/m^3).



Copepod eggs hatch to form an immature stage called a nauplius. The nauplius is a larval stage that does not closely resemble the adult and the nauplii of different species of copepods are not easily distinguished so they are lumped in this study. Copepods go through 5 naupliar molts before reaching the copepodid stage which is morphologically very similar to the adult. Because of their small size and high abundance, copepod nauplii are enumerated in the micro-zooplankton samples.

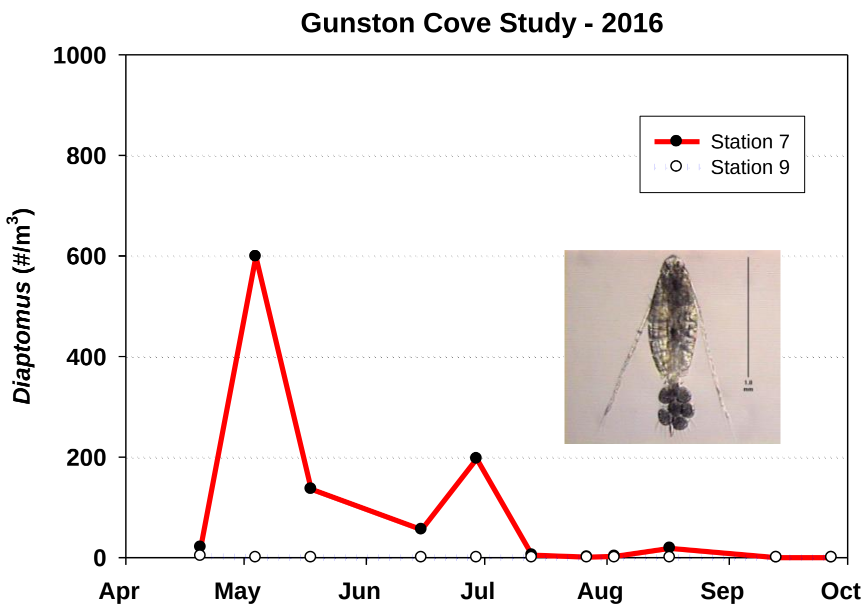
Figure 55. Copepod Nauplii Density by Station (#/L).

In the cove copepod nauplii peaked in early May, declined (Figure 55). Both stations exhibited a strong June peak. In the river, nauplii had a further maximum in August, while in the cover densities were low during July and August, but peaked again in late September. *Eurytemora* exhibited highest densities in the cove in early May attaining high densities of over 12,000/m^3 (Figure 56). Thereafter *Eurytemora* declined strongly in the cove. In the river *Eurytemora* peaked in June at about 4000/m^3. It declined slowly in the river over then ensuing months.



Eurytemora affinis is a large calanoid copepod characteristic of the freshwater and brackish areas of the Chesapeake Bay. *Eurytemora* is a cool water copepod which often reaches maximum abundance in the late winter or early spring. Included in this graph are adults and those copepodids that are recognizable as *Eurytemora*.

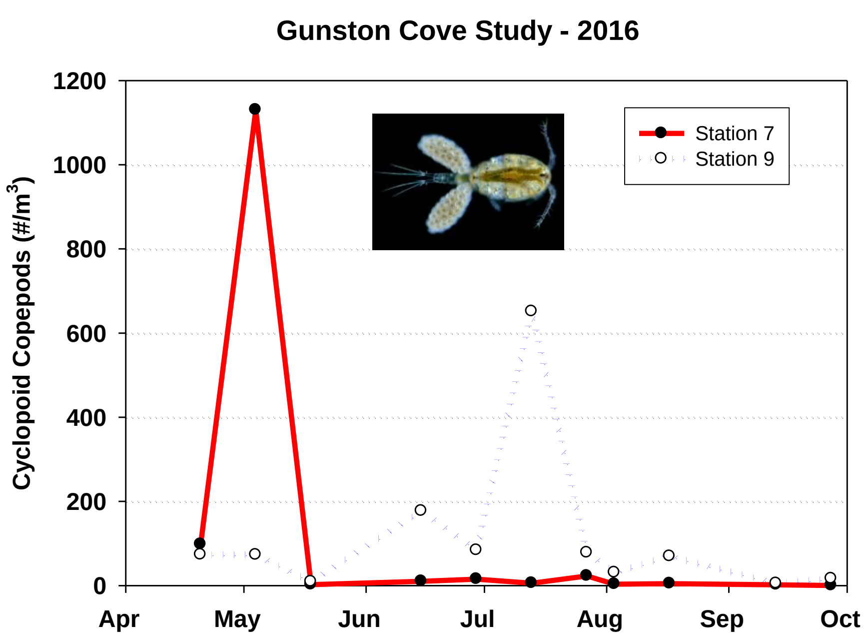
Figure 56. *Eurytemora* Density by Station (#/m^3).



Diaptomus pallidus is a calanoid copepod often found in moderate densities in the Gunston Cove area. *Diaptomus* is an efficient grazer of algae, bacteria, and detrital particles in freshwater ecosystems Included in this graph are adults and those copepodids that are recognizable as *Diaptomus*.

Figure 57. *Diaptomus* Density by Station (#/m^3).

Diaptomus was found in moderate densities in the cove in early May and declined thereafter. It was not observed in the river in 2016 (Figure 57). Cyclopoid copepods showed a peak in the cove in early spring at moderate levels (Figure 58). In the river they peaked at a somewhat lower level in July, but were present for most of the year.



Cyclopoids are the other major group of planktonic copepods. Cyclopoids feed on individual particles suspended in the water including small zooplankton as well as phytoplankton. In this study we have lumped all copepodid and adult cyclopoids together.

Figure 58. Cyclopoid Copepods by Station (#/m^3).

E. Ichthyoplankton – 2016

Larval fishes are transitional stages in the development of juvenile fishes. They range in development from newly hatched, embryonic fish to juvenile fish with morphological features similar to those of an adult. Many fishes such as clupeids (herring family), White Perch, Striped Bass, and Yellow Perch disperse their eggs and sperm into the open water. The larvae of these species are carried with the current and termed “ichthyoplankton”. Other fish species such as sunfishes and bass lay their eggs in “nests” on the bottom and their larvae are rare in the plankton.

After hatching from the egg, the larva draws nutrition from a yolk sack for a few days time. When the yolk sack diminishes to nothing, the fish begins a life of feeding on other organisms. This post yolk sack larva feeds on small planktonic organisms (mostly small zooplankton) for a period of several days. It continues to be a fragile, almost transparent, larva and suffers high mortality to predatory zooplankton and juvenile and adult fishes of many species, including its own. When it has fed enough, it changes into an opaque juvenile, with greatly enhanced swimming ability. It can no longer be caught with a slow-moving plankton net, but is soon susceptible to capture with the seine or trawl net.

In 2016, we collected 14 samples (7 at Station 7 and 7 at Station 9) during the months April through July and obtained a total of 1317 larvae (Table 4), which is similar to last year (1294). The fish larvae are sometimes too damaged to distinguish at the species level, thus some of the counts are only to the genus level. Much progress has been made in the identification of clupeid larvae (herring and shad), the dominant taxa is not Clupeidae anymore. The percent of the catch identified to the Family Clupeidae (but not further) was 5.54%. This is not because there were less herring and shad, but because we were able to identify them at the species level (as different species of *Alosa* and *Dorosoma*). Of the Clupeidae, Gizzard Shad was the dominant species with 22.78% of the catch. All clupeids together constituted 55.28% of the catch. Other abundant clupeids were Alewife at 15.72%, Blueback Herring at 7.52%, Hickory Shad at 1.59% and American Shad at 2.13%. The co-dominant species in the catch (together with Gizzard Shad) was White Perch at 22.78% of the catch. The total number of White Perch is likely slightly higher because 0.84% of the catch were *Morone sp.* that could only be identified to the genus level. It is likely that those were mostly White Perch. Striped bass (another *Morone sp.*) was present as well, and 3.49% of the catch was positively identified as Striped Bass. Another species somewhat abundant in the ichthyoplankton samples was Inland Silverside at 2.89%. A total of 14 species were identified; the species not mentioned yet (but included in Table 4) were present in low abundances.

Table 4. The larval fishes collected in Gunston Cove and the Potomac River in 2016

Taxon	Common Name	Station 7	Station 9	Total	% of Total
Alosa aestivalis	Blueback Herring	36	63	99	7.52
Alosa mediocris	Hickory Shad	14	7	21	1.59
Alosa pseudoharengus	Alewife	93	114	207	15.72
Alosa sapidissima	American Shad	26	2	28	2.13
Clupeidae	Herring or Shad	27	46	73	5.54
Dorosoma cepedianum	Gizzard Shad	169	131	300	22.78
Eggs	Unk. Fish eggs	76	92	168	12.76
Fundulus heteroclitus	Mummichog	1	0	1	0.075
Lepomis cyanellus	Green Sunfish	8	1	9	0.68
Lepomis gibbosus	Pumpkinseed	3	0	3	0.23
Lepomis macrochirus	Bluegill	1	0	1	0.075
Menidia beryllina	Inland Silverside	37	1	38	2.89
Morone americana	White Perch	26	274	300	22.78
Morone saxatilis	Striped Bass	1	45	46	3.49
Morone sp.	White Perch or Striped Bass	0	11	11	0.84
Perca flavescens	Yellow Perch	1	0	1	0.075
Strongylura marina	Atlantic Needlefish	0	1	1	0.075
unidentified	unidentified	9	1	10	0.76
	TOTAL	**528**	**789**	**1317**	100

The mean density of larvae, which takes the volume of water sampled into account over the time sampled, is shown in Figure 59 and 60. Clupeid larvae in Figure 59 include Blueback Herring, Hickory Shad, Alewife, American Shad, and Gizzard Shad. These have similar spawning patterns so they are lumped into one group for this analysis. Clupeids increased in the study areas in early spring attaining a maximum early to mid-May (Figure 59). This is similar to most earlier years of the study. The numbers dropped in June, but didn't come close to zero until July. The pattern still shows a distinct season of influx of larval Clupeids from May-June, which is right at the end of spawning seasons for most *Alosa*. The abundance of other larvae was generally lower, and had a distinct peak right at the start of sampling in April, and a smaller peak mid-May (Figure 60). The other larvae included all other taxa listed in Table 4.

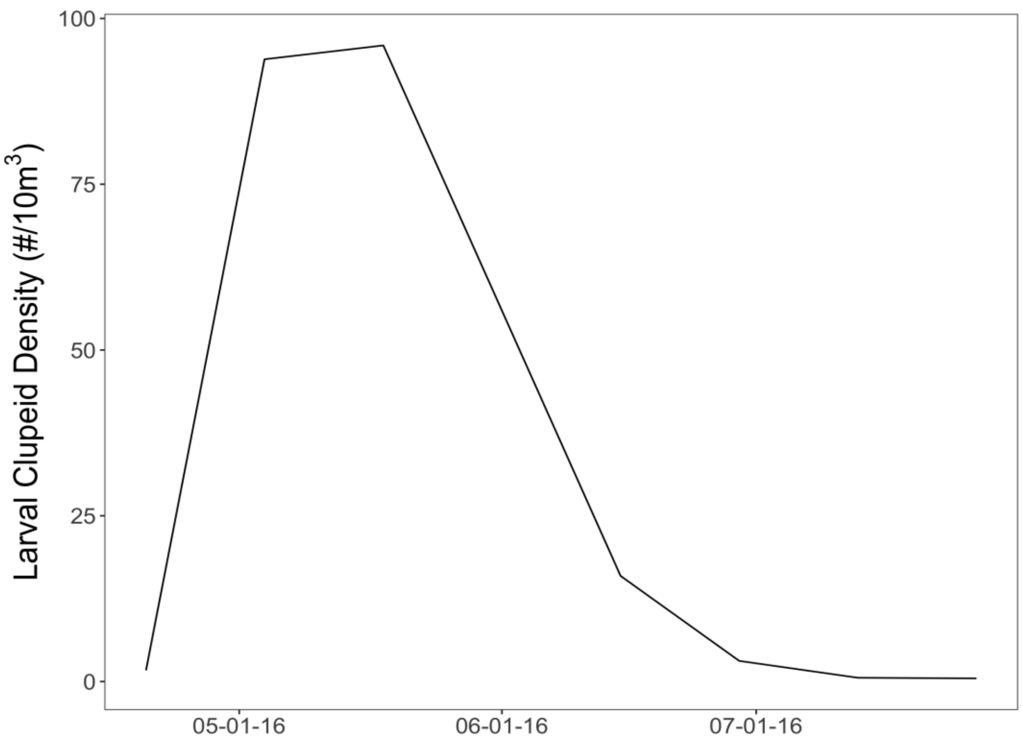


Figure 59. Clupeid larvae, mean density (abundance per $10m^3$).

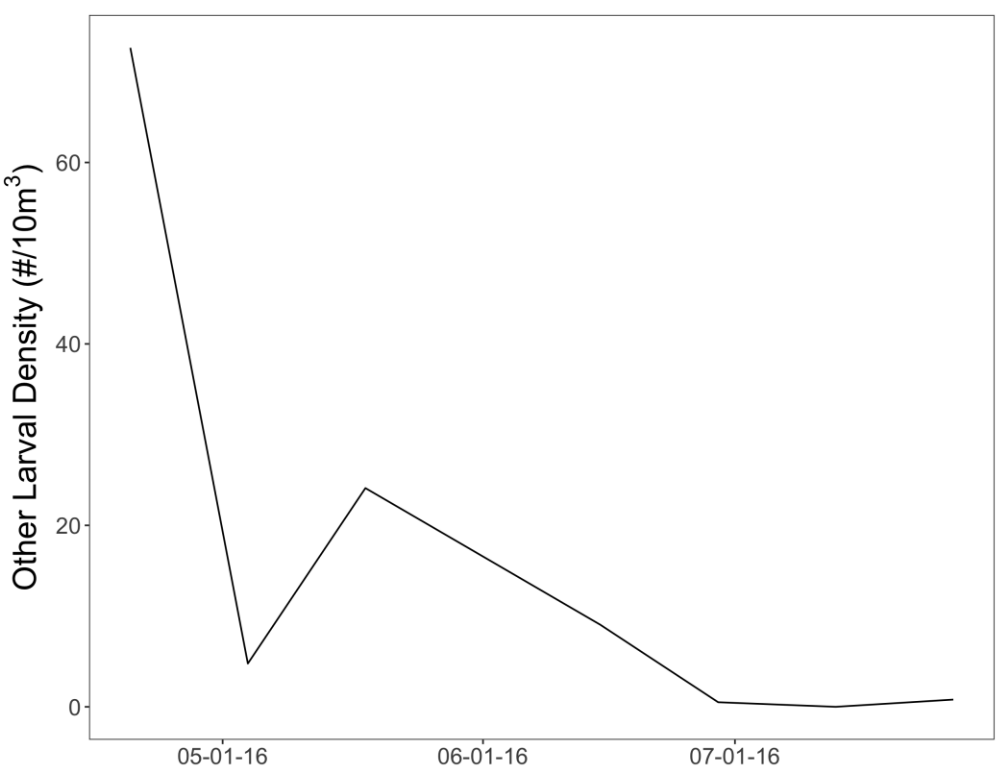


Figure 60. All other larvae, mean density (abundance per $10m^3$).

F. Adult and juvenile fishes – 2016

Trawls

Trawl sampling was conducted between April 21 and June 16 at station 10, and between April 21 and September 13 at station 7 and 9. These three fixed stations have been sampled continuously since the inception of the survey. Trawling at station 10 is obstructed by extensive submerged aquatic vegetation cover when we stop sampling. The site has been double sampled with a fyke net since 2012 which allows for comparison. The fyke net allows us to continue sampling that area when trawling at station 10 becomes impossible. A total of 2484 fishes comprising 24 species were collected in all trawl samples combined (Table 5). The dominant species of the fish collected in the trawls was White Perch (69.4%, numerically). Dominance of White Perch in the trawls is higher than last year, which indicates a decreased evenness (measure of diversity) of the fish community as sampled by the trawl. Gear selectivity plays a role here too, which is why we sample with multiple types of sampling gear. Other abundant taxa included herring or shad (7.9%), Spottail Shiner (7.7%), Sunfishes (2.2%), and Bay Anchovy (2.1%). Other species were observed sporadically and at low abundances (Tables 5 and 6).

The dominant migratory species, White Perch, was ubiquitous, occurring at all stations on every sampling date (Tables 6 and 7). In the spring, adult White Perch were primarily caught in the nets while later in the summer juveniles dominated. A clear peak in abundance for White Perch was end of June to early July (Table 6).

White Perch (*Morone americana*), the most common fish in the open waters of Gunston Cove, continues to be an important commercial and popular game fish. Adults grow to over 30 cm long. Sexual maturity begins the second year at lengths greater than 9 cm. As juveniles, they feed on zooplankton and macrobenthos, but as they get larger they consume fish as well.

Spottail Shiner (*Notropis hudsonius*), a member of the minnow family, is moderately abundant in the open water and along the shore. Spawning occurs throughout the warmer months. It reaches sexual maturity at about 5.5 cm and may attain a length of 10 cm. They feed primarily on benthic invertebrates and occasionally on algae and plants.

Trawling collects fish that are located in the open water near the bottom. Due to the shallowness of Gunston Cove, the volume collected is a substantial part of the water column. However, in the river channel, the near bottom habitat through which the trawl moves is only a small portion of the water column. Fishes tend to concentrate near the bottom or along shorelines rather than in the upper portion of the open water.

Table 5. Adult and Juvenile Fish Collected by Trawling. Gunston Cove Study - 2016

Scientific Name	Common Name	Abundance
Morone americana	White Perch	1724
Alosa sp.	herring or shad	196
Notropis hudsonius	Spottail Shiner	192
Lepomis sp.	Sunfishes	54
Anchoa mitchilli	Bay Anchovy	53
Brevoortia tyrannus	Atlantic Menhaden	49
Lepomis microlophus	Redear Sunfish	46
Lepomis macrochirus	Bluegill	45
Etheostoma olmstedi	Tessellated Darter	33
Lepomis gibbosus	Pumpkinseed	17
Fundulus diaphanus	Banded Killifish	15
Ictalurus furcatus	Blue Catfish	9
Lepomis auritus	Redbreast Sunfish	9
Perca flavescens	Yellow Perch	8
Menidia beryllina	Inland Silverside	6
Morone saxatilis	Striped Bass	6
Carassius auratus	Goldfish	5
Ameiurus nebulosus	Brown Bullhead	4
Ameiurus catus	White Bullhead	3
Pomoxis nigromaculatus	Black Crappie	3
Hybognathus regius	Eastern Silvery Minnow	2
Alosa pseudoharengus	Alewife	1
Alosa sapidissima	American Shad	1
Dorosoma cepedianum	Gizzard Shad	1
Enneacanthus gloriosus	Bluespotted Sunfish	1
Lepomis cyanellus	Green Sunfish	1
	Total	**2484**

Table 6. Adult and Juvenile Fish Collected by Trawling. Gunston Cove Study - 2016

Scientific Name	Common Name	21-Apr	5-May	19-May	16-Jun	29-Jun	13-Jul	27-Jul	3-Aug	17-Aug	13-Sep
Alosa pseudoharengus	Alewife	0	0	0	1	0	0	0	0	0	0
Alosa sapidissima	American Shad	1	0	0	0	0	0	0	0	0	0
Alosa sp.	Herring or shad	2	0	0	1	26	13	134	13	6	1
Ameiurus catus	White Bullhead	0	0	1	0	2	0	0	0	0	0
Ameiurus nebulosus	Brown Bullhead	0	1	1	1	1	0	0	0	0	0
Anchoa mitchilli	Bay anchovy	3	0	0	0	0	0	0	0	50	0
Brevoortia tyrannus	Atlantic Menhaden	0	0	0	0	0	19	23	7	0	0
Carassius auratus	Goldfish	2	1	1	0	0	0	0	0	1	0
Dorosoma cepedianum	Gizzard Shad	0	1	0	0	0	0	0	0	0	0
Enneacanthus gloriosus	Bluespotted Sunfish	0	0	1	0	0	0	0	0	0	0
Etheostoma olmstedi	Tessellated Darter	6	0	14	0	10	2	0	1	0	0
Fundulus diaphanus	Banded Killifish	5	1	0	0	0	0	0	0	9	0
Hybognathus regius	Eastern Silvery Minnow	0	1	0	0	0	1	0	0	0	0
Ictalurus furcatus	Blue Catfish	0	0	0	0	0	0	0	0	9	0
Lepomis auritus	Redbreast Sunfish	0	0	1	0	0	0	0	0	0	8
Lepomis cyanellus	Green Sunfish	1	0	0	0	0	0	0	0	0	0
Lepomis gibbosus	Pumpkinseed	3	3	3	3	1	3	0	0	0	1
Lepomis macrochirus	Bluegill	9	1	6	0	0	1	1	0	13	14
Lepomis microlophus	Redear Sunfish	5	7	5	6	18	0	2	0	0	3
Lepomis sp.	Sunfishes	0	0	0	0	0	0	2	20	11	21
Menidia beryllina	Inland Silverside	5	0	1	0	0	0	0	0	0	0
Morone americana	White Perch	14	3	2	18	1070	526	15	20	53	3
Morone saxatilis	Striped Bass	0	1	0	0	0	2	0	3	0	0
Notropis hudsonius	Spottail Shiner	49	5	7	48	51	12	6	11	3	0
Perca flavescens	Yellow Perch	3	1	2	0	0	0	1	1	0	0
Pomoxis nigromaculatus	Black Crappie	2	0	0	0	0	0	0	1	0	0
	Total	**110**	**26**	**45**	**78**	**1179**	**579**	**184**	**77**	**155**	**51**

In total numbers and species richness of fish, station 7 dominated the other stations by far with 2243 individuals from 22 species (Table 7, Figure 61a). Stations 9 and 10 had 98 individuals from 4 species and 143 individuals from 17 species, respectively (Table 7). Station 9 samples the open water of the mainstem Potomac and thereby doesn't sample preferred habitat such as the littoral zone or the bottom. The few species collected are indeed pelagic (open water) species; herring or shad, Bay Anchovy, and White Perch. A notable other species collected only in station 9 is Blue Catfish, which is an invasive piscivorous species. The total abundance and number of species in station 9 have been declining over time. Whether this is related to the introduced catfish is yet unknown. A high number of White Perch were collected in the Cove (station 7) in mid-summer (Table 6), which constitutes the bulk of the total sample. Other taxa collected in high abundance in station 7 were herring or shad (174 specimens) and Spottail Shiner (139 specimens).

Table 7. Adult and Juvenile Fish Collected by Trawling. Gunston Cove Study – 2016

Scientific Name	**Common Name**	**7**	**9**	**10**
Alosa pseudoharengus	Alewife	1	0	0
Alosa sapidissima	American Shad	0	0	1
Alosa sp.	Herring or shad	174	20	2
Ameiurus catus	White Bullhead	3	0	0
Ameiurus nebulosus	Brown Bullhead	1	0	3
Anchoa mitchilli	Bay Anchovy	1	49	3
Brevoortia tyrannus	Atlantic Menhaden	49	0	0
Carassius auratus	Goldfish	2	0	3
Dorosoma cepedianum	Gizzard Shad	1	0	0
Enneacanthus gloriosus	Bluespotted Sunfish	1	0	0
Etheostoma olmstedi	Tessellated Darter	15	0	18
Fundulus diaphanus	Banded Killifish	9	0	6
Hybognathus regius	Eastern Silvery Minnow	1	0	1
Ictalurus furcatus	Blue Catfish	0	9	0
Lepomis auritus	Redbreast Sunfish	9	0	0
Lepomis cyanellus	Green Sunfish	0	0	1
Lepomis gibbosus	Pumpkinseed	10	0	7
Lepomis macrochirus	Bluegill	32	0	13
Lepomis microlophus	Redear Sunfish	34	0	12
Lepomis sp.	Sunfishes	54	0	0
Menidia beryllina	Inland Silverside	0	0	6
Morone americana	White Perch	1694	20	10
Morone saxatilis	Striped Bass	6	0	0
Notropis hudsonius	Spottail Shiner	139	0	53
Perca flavescens	Yellow Perch	6	0	2
Pomoxis nigromaculatus	Black Crappie	1	0	2
	Total	**2243**	**98**	**143**

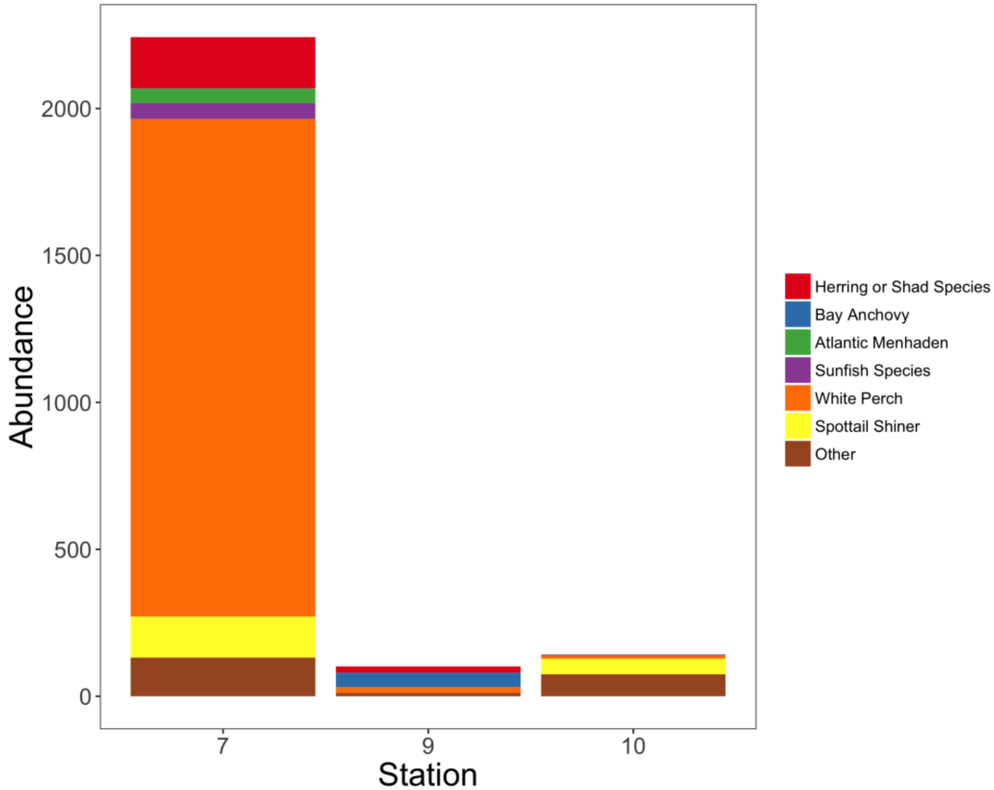


Figure 61a. Adult and Juvenile Fishes Collected by Trawling in 2016. Dominant Species by Station.

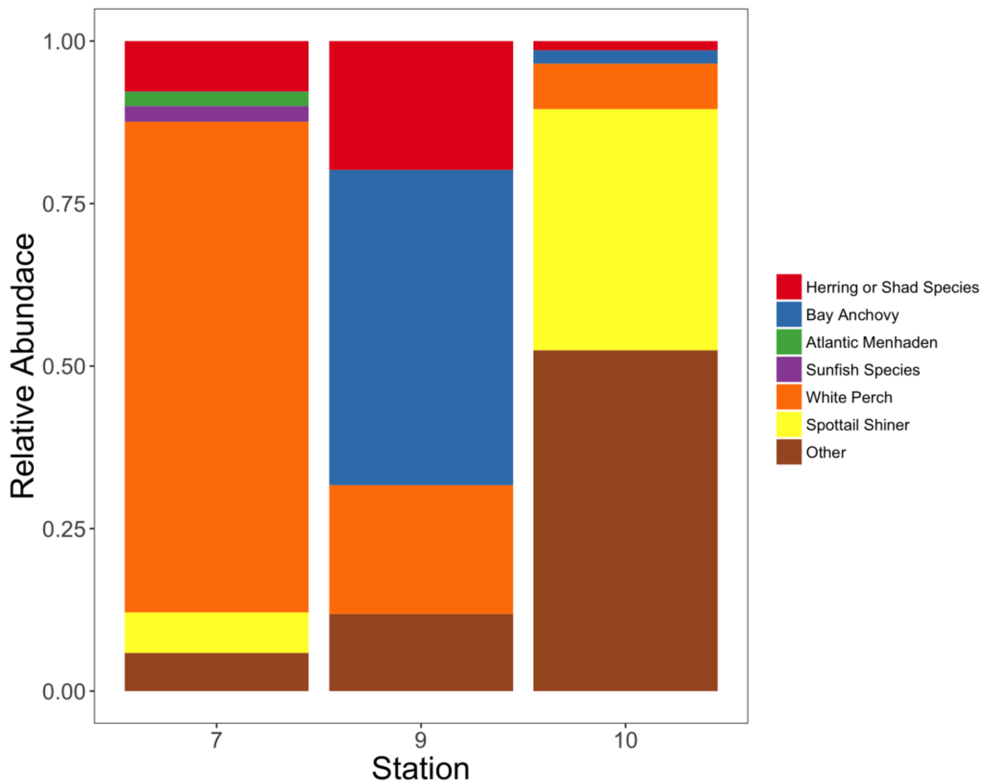


Figure 61b. Relative abundance of Adult and Juvenile Fishes Collected by Trawling in 2016.

The six most abundant species varied in representation across stations (Figure 62b). At all stations, White Perch made up a significant proportion of the total catch. Total catch of White Perch was significantly higher in Station 7 than Station 9 and 10, and is the main reason for the high total catch of station 7 (Figure 61a). We were able to identify a few (6) juvenile Striped Bass among the representatives of the *Morone* genus (the rest were White Perch); which shows that the juveniles of Striped Bass can be found in the fresh upper reaches of the Potomac River. Station 10 showed a high proportion of Spottail Shiner, which was caught in lower abundance at station 7 as well. Alosines (herring or shad) were a dominant group at station 7 and 9, with representation from at least Alewife and American Shad (most were only identified to genus). Blue Catfish (not shown in figure) are primarily a mainstem species and have not been featured prominently at stations within the cove (9 collected at station 9 this year, while 0 at station 7 or 10). All species were present in their highest abundance at Station 7, except for bay Anchovy. Bay Anchovy, which is a migratory species that is usually found in higher salinities was mostly found in the mainstem (station 9). Station 7 was overall the most productive site, with a total abundance an order of magnitude higher than the other two stations.

When looking at the seasonal trend in the same data it is clear that White Perch was the most common species, with a distinct peak in abundance in mid-summer (Figure 62a and b). The relative abundance of Spottail Shiner was highest early in the season, while sunfishes were mostly collected at the end of the season. Herring or shad species were most abundant in July, but had some presence throughout the season. These all constitute juveniles that were spawned in spring (March-May) and remain in Gunston Cove, which serves as a nursery to these species. Just as in previous years, the most productive month was June, which was dominated by a large cohort of juvenile White Perch. Atlantic menhaden is an interesting find in station 7, which like Bay Anchovy is usually found in higher salinities.

Blueback Herring (*Alosa aestivalis*) and Alewife (*Alosa pseudoharengus*) were formerly major commercial species, but are now collapsed stocks. Adults grow to over 30 cm and are found in the coastal ocean. They are anadromous and return to freshwater creeks to spawn in March, April and May. They feed on zooplankton and may eat fish larvae.

Bay Anchovy (*Anchoa mitchilli)* is commonly found in shallow tidal areas but usually in higher salinities. Due to its euryhaline nature, it can occur in freshwater. Feeds mostly on zooplankton, but also on small fishes, gastropods and isopods. They are an important forage fish.

Blue Catfish (*Ictalurus furcatus*) is an introduced species from the Mississippi River basin. They have been intentionally stocked in the James and Rappahannock rivers for food and sport. They have expanding their range and seem to replace white catfish and perhaps also Channel Catfish and bullheads. As larvae, they feed on zooplankton; juveniles and adults mostly on fishes, and on benthos, and detritus.

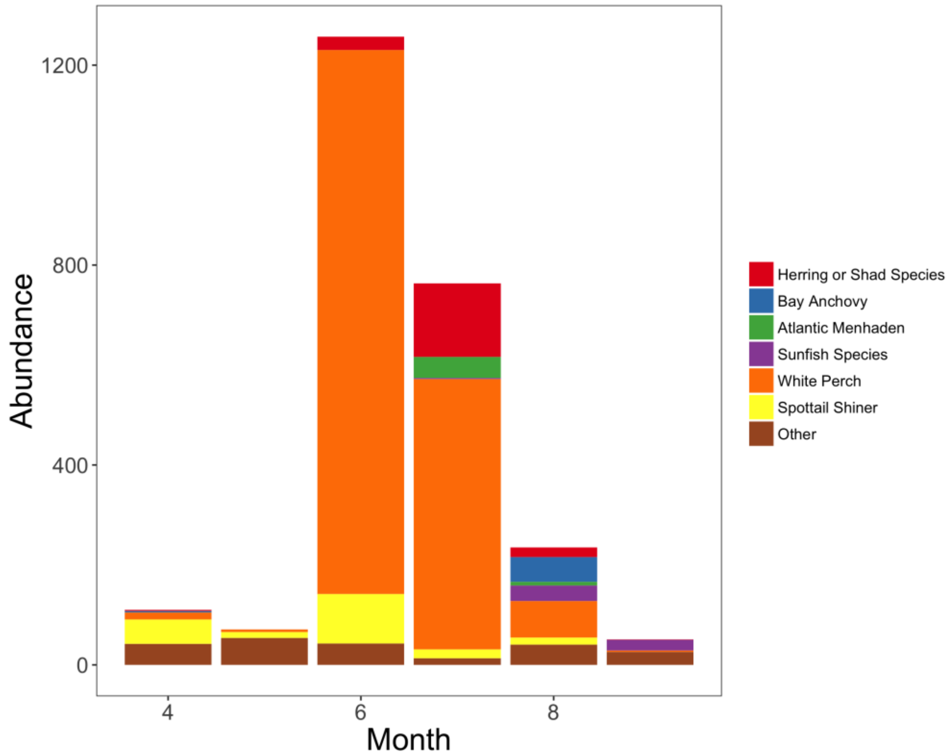


Figure 62a. Adult and Juvenile Fishes Collected by Trawling in 2016. Dominant Species by Month.

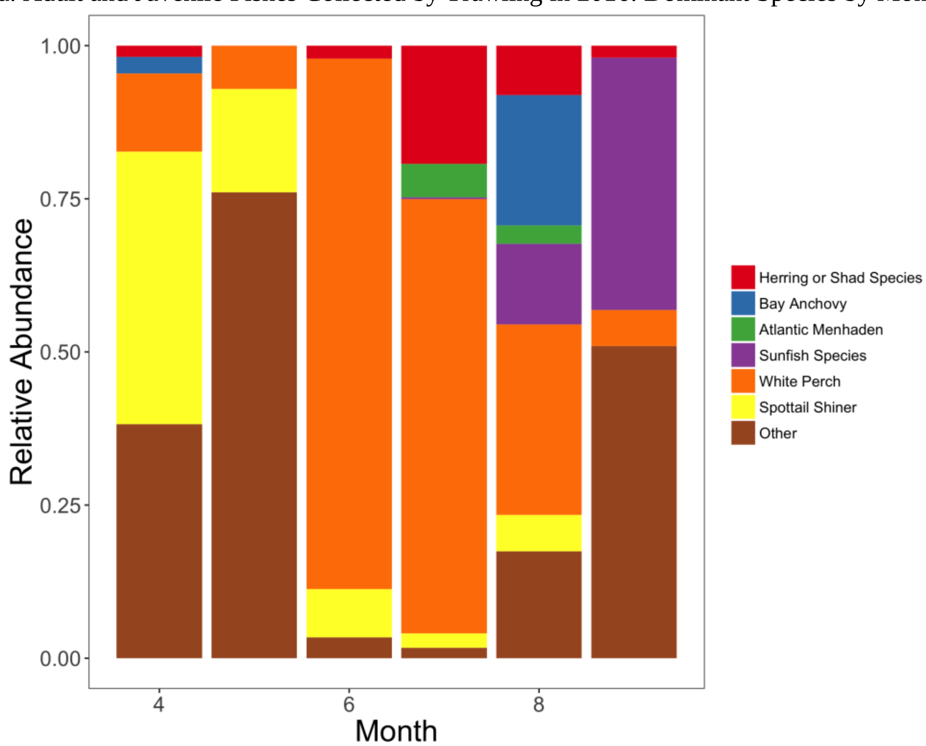


Figure 62b. Relative Abundance for Adult and Juvenile Fishes Collected by Trawling in 2016.

Seines

Seine sampling was conducted approximately semi-monthly at 4 stations between April 14 and September 9. As planned, only one sampling trip per month was performed in April and September. We stopped seining at station 4 on June 16 (last seine sample was on June 16) due to dense SAV growth.

Stations 4, 6, and 11 have been sampled continuously since 1985. Station 4B was added in 2007 to have a continuous seine record when dense SAV impedes seining in 4. Station 4B is a routine station now, also when seining at 4 is possible. This allows for comparison between 4 and 4B.

A total of 35 seine samples were conducted, comprising 3885 fishes of 26 species (Table 8). This is a little lower than the number of individuals and species collected last year. Similar to last year, the most dominant species in seine catches was Banded Killifish, with a relative contribution to the catch of 56.4%. Other dominant species (with >5% of relative abundance) were White Perch (10.2%) followed by Inland Silverside (7.1%), eastern Silvery Minnow (6.2%), and *Alosa* sp. (5.2%). Other taxa that contributed at least 1% to total abundance include *Lepomis* sp. (3.4%), Mummichog (1.5%), Quillback (1.3%), pumpkinseed (1.2%), Tessellated Darter (1%) and Spottail shiner (1%). Other species occurred at low abundances (Table 8). The extensive SAV cover, which now is an established presence in the cove, is responsible for the high abundance of Banded Killifish in the seine catches.

Banded Killifish was abundant and present at all sampling dates in seines, with higher abundances in early summer than late summer (Table 9, Figure 63). While the highest abundance of Banded Killifish occurred in June, the highest total abundance was in May due to high numbers of *Alosa sp*. and inland Silverside in May. White Perch was mostly collected later in the season with highest numbers in August (Table 9, Figure 63)

The highest abundance of Banded Killifish was found in station 6 this year, which was also the site of the highest total abundance (Table 10, Figure 64). Banded Killifish was most dominant in all sites except Station 11, where White Perch, eastern Silvery Minnow, and Inland Silverside were more abundant. Station 11 is a beach closest to the mainstem and is the station least associated with SAV. Banded Killifish can find more preferred habitat in the other stations. Abundance varied from 1364 fish at station 6 to 631 fish at station 4 (Table 10). Species richness varied from 14 species at station 4 to 22 species at station 4B.

Table 8. Adult and Juvenile Fish Collected by Seining. Gunston Cove Study - 2016

Scientific Name	Common Name	Abundance
Fundulus diaphanus	Banded Killifish	2192
Morone americana	White Perch	395
Menidia beryllina	Inland Silverside	277
Hybognathus regius	Eastern Silvery Minnow	242
Alosa sp.	Herring or shad	203
Lepomis sp.	Sunfishes	133
Fundulus heteroclitus	Mummichog	60
Carpiodes cyprinus	Quillback	49
Lepomis gibbosus	Pumpkinseed	47
Etheostoma olmstedi	Tessellated Darter	41
Notropis hudsonius	Spottail Shiner	41
Notemigonus crysoleucas	Golden Shiner	40
Alosa sapidissima	American Shad	30
Lepomis microlophus	Redear Sunfish	29
Lepomis macrochirus	Bluegill	23
Micropterus salmoides	Large-mouth Bass	21
Perca flavescens	Yellow Perch	16
Carassius auratus	Goldfish	11
Morone saxatilis	Striped Bass	10
Enneacanthus gloriosus	Bluespotted Sunfish	7
Gambusia holbrooki	Mosquitofish	5
Brevoortia tyrannus	Atlantic Menhaden	3
Anchoa mitchilli	Bay anchovy	2
Lepomis auritus	Redbreast Sunfish	2
Lepisosteus osseus	Longnose Gar	2
Strongylura marina	Atlantic Needlefish	2
Alosa aestivalis	Blueback Herring	1
Ameiurus nebulosus	Brown Bullhead	1
	TOTAL	**3885**

Banded Killifish (*Fundulus diaphanus*) is a small fish, but the most abundant species in shoreline areas of the cove. Individuals become sexually mature at about 5 cm in length and may grow to over 8 cm long. Spawning occurs throughout the warmer months over vegetation and shells. They feed on benthic invertebrates, vegetation, and very small fishes.

White Perch (*Morone americana*), which was discussed earlier in the trawl section, is also a common shoreline fish as juveniles collected in seines. Abundances of White Perch in the seine collections are decreasing as the Banded Killifish catches increase, which indicates a change in community structure in the littoral zone.

Seining is conducted in shallow water adjacent to the shoreline. Some fish minimize predation by congregating along the shoreline rather than disperse through the open water. While seines and trawls tend to collect about the same number of individuals per effort, seines sample a smaller volume of water emphasizing the higher densities of fish along the shoreline.

Table 9. Adult and Juvenile Fish Collected by Seining. Gunston Cove Study - 2016

Scientific Name	**Common Name**	**21-Apr**	**5-May**	**19-May**	**16-Jun**	**29-Jun**	**13-Jul**	**27-Jul**	**3-Aug**	**17-Aug**	**13-Sep**
Alosa aestivalis	Blueback Herring	0	1	0	0	0	0	0	0	0	0
Alosa sapidissima	American Shad	0	26	4	0	0	0	0	0	0	0
Alosa sp.	Herring or shad	0	181	1	0	8	0	1	10	0	2
Ameiurus nebulosus	Brown Bullhead	0	0	0	0	0	1	0	0	0	0
Anchoa mitchilli	Bay anchovy	1	1	0	0	0	0	0	0	0	0
Brevoortia tyrannus	Atlantic Menhaden	0	0	0	0	0	0	2	1	0	0
Carassius auratus	Goldfish	1	1	0	1	0	0	2	3	1	2
Carpiodes cyprinus	Quillback	0	0	0	5	3	2	27	7	5	0
Enneacanthus gloriosus	Bluespotted Sunfish	0	0	2	1	0	0	0	0	0	4
Etheostoma olmstedi	Tessellated Darter	8	1	11	9	8	2	0	0	2	0
Fundulus diaphanus	Banded Killifish	370	159	414	533	262	103	33	172	108	38
Fundulus heteroclitus	Mummichog	10	8	4	4	0	13	3	6	5	7
Gambusia holbrooki	Mosquitofish	0	0	0	0	1	0	1	0	3	0
Hybognathus regius	Eastern Silvery Minnow	2	24	3	0	2	14	0	109	45	43
Lepisosteus osseus	Longnose Gar	0	0	0	0	1	1	0	0	0	0
Lepomis auritus	Redbreast Sunfish	0	0	0	0	0	0	0	0	1	1
Lepomis gibbosus	Pumpkinseed	0	0	1	39	5	0	0	0	0	2
Lepomis macrochirus	Bluegill	0	0	3	5	0	4	6	1	2	2
Lepomis microlophus	Redear Sunfish	4	8	1	6	4	0	0	0	1	5
Lepomis sp.	Sunfishes	0	0	0	0	0	1	18	14	39	61
Menidia beryllina	Inland Silverside	33	144	50	15	1	4	3	1	13	13
Micropterus salmoides	Large-mouth Bass	0	0	1	0	15	2	2	0	0	1
Morone americana	White Perch	1	2	0	2	59	67	72	31	122	39
Morone saxatilis	Striped Bass	0	1	0	5	0	0	2	2	0	0
Notemigonus crysoleucas	Golden Shiner	1	23	0	2	0	1	0	0	4	9
Notropis hudsonius	Spottail Shiner	7	5	4	7	9	0	1	1	5	2
Perca flavescens	Yellow Perch	4	3	3	2	3	0	0	1	0	0
Strongylura marina	Atlantic Needlefish	0	0	0	1	1	0	0	0	0	0
	TOTAL	**442**	**588**	**502**	**637**	**382**	**215**	**173**	**359**	**356**	**231**

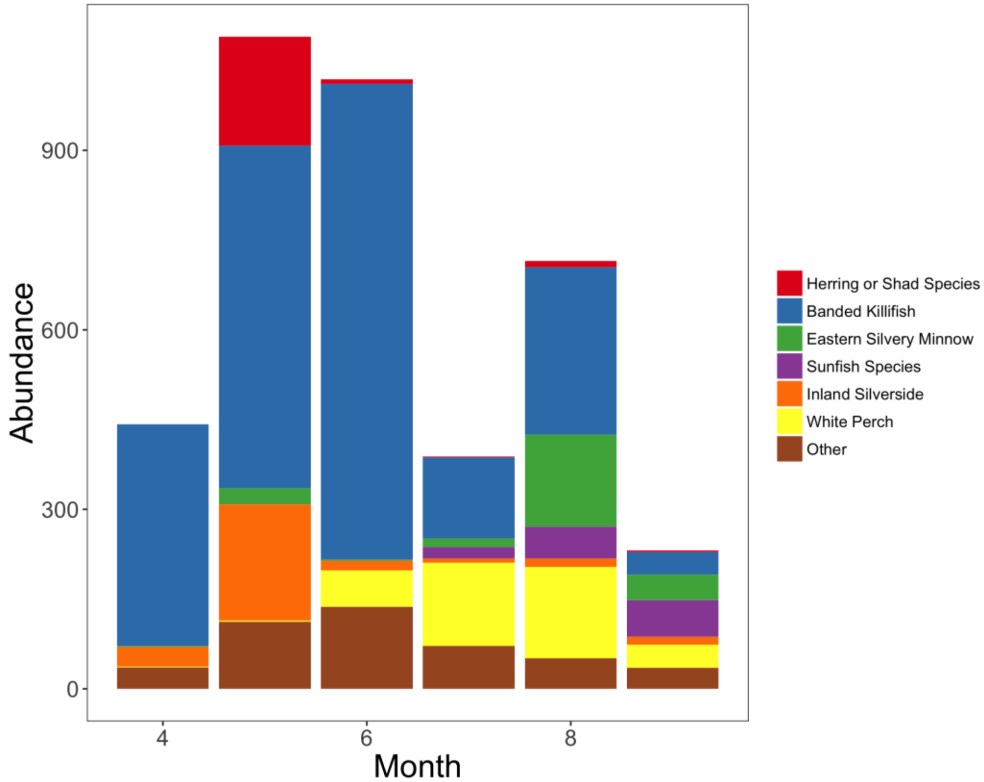


Figure 63. Adult and Juvenile Fish Collected by Seining in 2016. Dominant Species by Month.

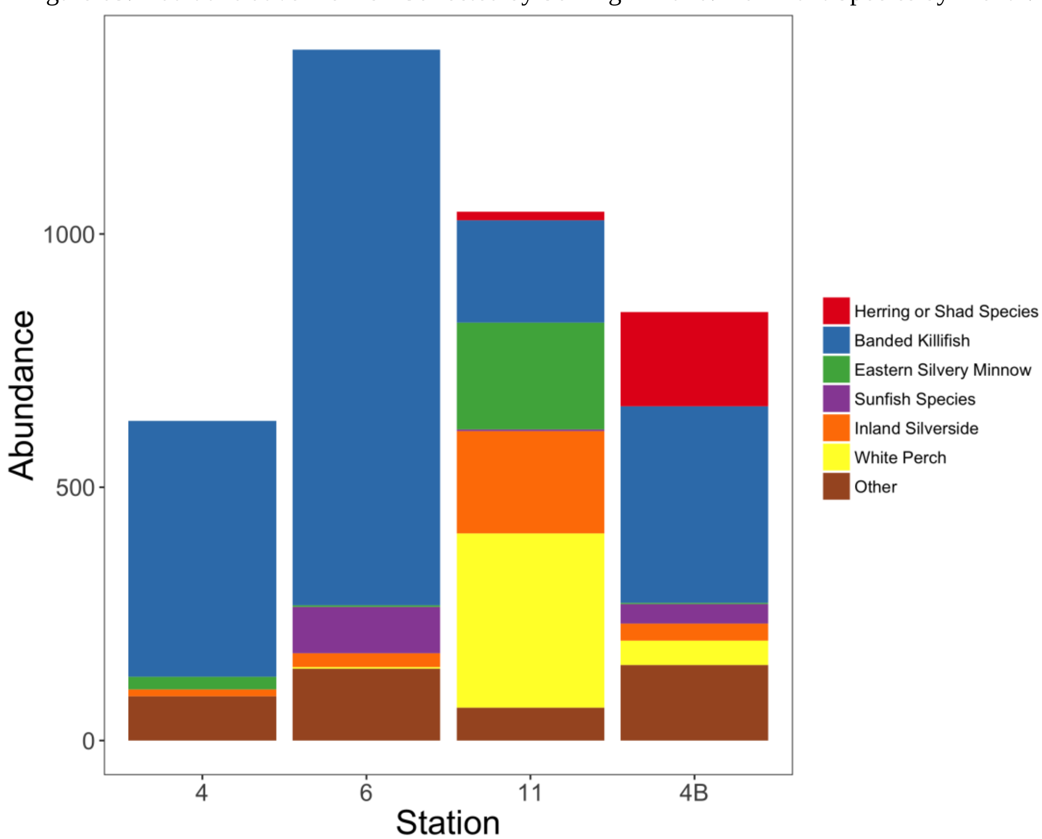


Figure 64. Adult and Juvenile Fishes Collected by Seining in 2016. Dominant Species by Station.

Table 10. Adult and Juvenile Fish Collected by Seining in 2016 per station in Gunston Cove.

Scientific Name	**Common Name**	**4**	**6**	**11**	**4B**
Alosa aestivalis	Blueback Herring	0	0	0	1
Alosa sapidissima	American Shad	5	0	0	25
Alosa sp.	Herring or shad	0	0	17	186
Ameiurus nebulosus	Brown Bullhead	0	0	1	0
Anchoa mitchilli	Bay anchovy	0	0	2	0
Brevoortia tyrannus	Atlantic Menhaden	0	0	3	0
Carassius auratus	Goldfish	1	8	0	2
Carpiodes cyprinus	Quillback	0	0	32	17
Enneacanthus gloriosus	Bluespotted Sunfish	2	5	0	0
Etheostoma olmstedi	Tessellated Darter	3	9	0	29
Fundulus diaphanus	Banded Killifish	505	1097	202	388
Fundulus heteroclitus	Mummichog	12	43	2	3
Gambusia holbrooki	Mosquitofish	0	5	0	0
Hybognathus regius	Eastern Silvery Minnow	25	3	211	3
Lepisosteus osseus	Longnose Gar	0	1	0	1
Lepomis auritus	Redbreast Sunfish	0	0	0	2
Lepomis gibbosus	Pumpkinseed	38	1	0	8
Lepomis macrochirus	Bluegill	5	13	0	5
Lepomis microlophus	Redear Sunfish	12	9	2	6
Lepomis sp.	Sunfishes	0	92	3	38
Menidia beryllina	Inland Silverside	14	27	202	34
Micropterus salmoides	Large-mouth Bass	0	6	1	14
Morone americana	White Perch	0	3	344	48
Morone saxatilis	Striped Bass	0	0	7	3
Notemigonus crysoleucas	Golden Shiner	5	33	1	1
Notropis hudsonius	Spottail Shiner	2	2	13	24
Perca flavescens	Yellow Perch	2	6	0	8
Strongylura marina	Atlantic Needlefish	0	1	1	0
	TOTAL	**631**	**1364**	**1044**	**846**

Fyke nets

We added fyke nets to the sampling regime in 2012 to better represent the fish community present within SAV beds. This year we collected a total number of 456 specimens of 15 species in the two fyke nets (Station Fyke 1 and Station Fyke 2; Figure 1b; Table 11), which is a little bit less than last year. While Banded Killifish is abundant here as well (23% of the catch), which is not surprising seen as this gear specifically samples SAV habitat, the fyke nets show a high contribution of sunfishes relative to the other gear types (64% of the catch). Taxa other than Banded Killifish contributing to more than 1% of the catch include Sunfishes (not further identified than the genus level) at 39.7%, Pumpkinseed at 13.8%, Bluegill at 6.6%, Inland Silverside at 4.4%, White Perch 4.2%, Redear Sunfish at 3.9%, and Largemouth Bass at 1.9%. This emphasizes the value of sampling with different gear types when striving to represent the community present in Gunston Cove. We collected one brown Bullhead, which is a native catfish, in the fyke nets this year. Relative high catches in the fyke nets of native catfishes in previous years may be an indication of a spatial shift of native bullheads and catfishes to shallow vegetated habitat, now that Blue Catfish is caught in higher numbers in the open water trawls (in the Potomac mainstem).

Table 11. Adult and Juvenile Fish Collected by Fyke Nets. Gunston Cove Study - 2016

Scientific Name	**Common Name**	**Abundance**
Lepomis sp.	Sunfishes	181
Fundulus diaphanus	Banded Killifish	105
Lepomis gibbosus	Pumpkinseed	63
Lepomis macrochirus	Bluegill	30
Menidia beryllina	Inland Silverside	20
Morone americana	White Perch	19
Lepomis microlophus	Redear Sunfish	18
Micropterus salmoides	Largemouth Bass	9
Perca flavescens	Yellow Perch	3
Pomoxis nigromaculatus	Black Crappie	2
Ameiurus nebulosus	Brown Bullhead	1
Carassius auratus	Goldfish	1
Enneacanthus gloriosus	Bluespotted Sunfish	1
Hybognathus regius	Eastern Silvery Minnow	1
Lepomis auritus	Redbreast Sunfish	1
Notropis hudsonius	Spottail Shiner	1
	TOTAL	456

Table 12. Adult and Juvenile Fish Collected by Fyke Nets. Gunston Cove Study - 2016

Scientific Name	Common Name	21-Apr	5-May	19-May	16-Jun	29-Jun	13-Jul	27-Jul	3-Aug	17-Aug	13-Sep
Ameiurus nebulosus	Brown Bullhead	1	0	0	0	0	0	0	0	0	0
Carassius auratus	Goldfish	0	0	0	0	0	1	0	0	0	0
Enneacanthus gloriosus	Bluespotted Sunfish	0	0	0	0	1	0	0	0	0	0
Fundulus diaphanus	Banded Killifish	1	0	2	7	37	1	2	26	25	4
Hybognathus regius	Eastern Silvery Minnow	0	0	1	0	0	0	0	0	0	0
Lepomis auritus	Redbreast Sunfish	0	0	0	0	0	0	0	1	0	0
Lepomis gibbosus	Pumpkinseed	0	0	1	8	6	22	11	4	1	10
Lepomis macrochirus	Bluegill	0	2	1	0	1	4	8	0	7	7
Lepomis microlophus	Redear Sunfish	0	0	1	0	11	0	5	0	1	0
Lepomis sp.	Sunfishes	0	0	0	0	0	6	7	70	76	22
Menidia beryllina	Inland Silverside	1	0	0	0	0	1	0	15	3	0
Micropterus salmoides	Large-mouth Bass	0	0	0	0	3	4	2	0	0	0
Morone americana	White Perch	0	3	0	0	7	4	4	0	1	0
Notropis hudsonius	Spottail Shiner	0	0	0	0	0	0	0	1	0	0
Perca flavescens	Yellow Perch	0	0	0	0	1	0	0	2	0	0
Pomoxis nigromaculatus	Black Crappie	0	2	0	0	0	0	0	0	0	0
	TOTAL	**3**	**7**	**6**	**15**	**67**	**43**	**39**	**119**	**114**	**43**

Highest abundances were collected in August this year which was attributable to a high abundance of sunfishes that month (Table 12, Figure 65). Other species, namely banded Killifish and Inland Silverside, were present at their highest abundance that month as well. August is the month during which SAV is most extensive, which is used as habitat by these species.

Fyke 1 had a higher total catch (268 fishes; Table 13), The community structure collected with the two fyke nets is very similar; similar community composition with a similar relative contribution to the catch (Table 13, Figure 66). Abundance in Fyke 1 was higher than Fyke 2 just like last year, due to the higher abundance of Sunfishes and Inland Silverside collected in Fyke 1 (Figure 66).

Table 13. Adult and Juvenile Fish Collected by Fyke Nets. Gunston Cove Study - 2016

Scientific Name	**Common Name**	**Fyke 1**	**Fyke 2**
Ameiurus nebulosus	Brown Bullhead	1	0
Carassius auratus	Goldfish	0	1
Enneacanthus gloriosus	Bluespotted Sunfish	0	1
Fundulus diaphanus	Banded Killifish	57	48
Hybognathus regius	Eastern Silvery Minnow	0	1
Lepomis auritus	Redbreast Sunfish	1	0
Lepomis gibbosus	Pumpkinseed	37	26
Lepomis macrochirus	Bluegill	16	14
Lepomis microlophus	Redear Sunfish	8	10
Lepomis sp.	Sunfishes	114	67
Menidia beryllina	Inland Silverside	16	4
Micropterus salmoides	Largemouth Bass	8	1
Morone americana	White Perch	9	10
Notropis hudsonius	Spottail Shiner	0	1
Perca flavescens	Yellow Perch	1	2
Pomoxis nigromaculatus	Black Crappie	0	2
	TOTAL	268	188

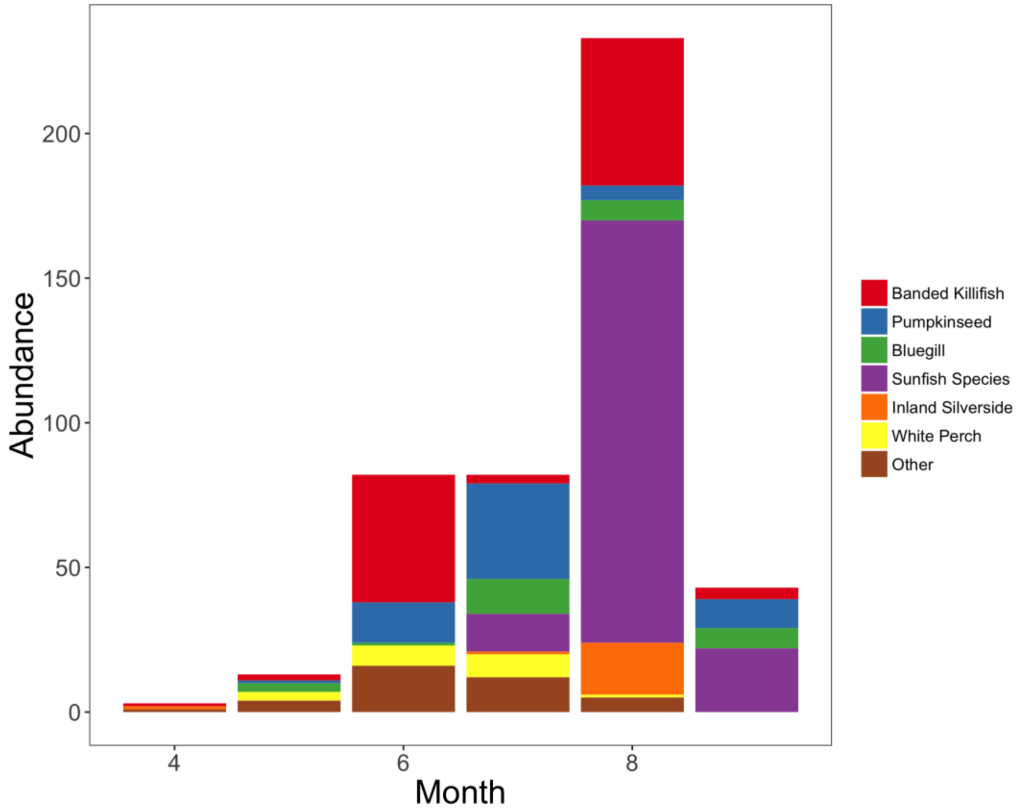


Figure 65. Adult and Juvenile Fish Collected by Fyke Nets. Dominant Species by Month. 2016.

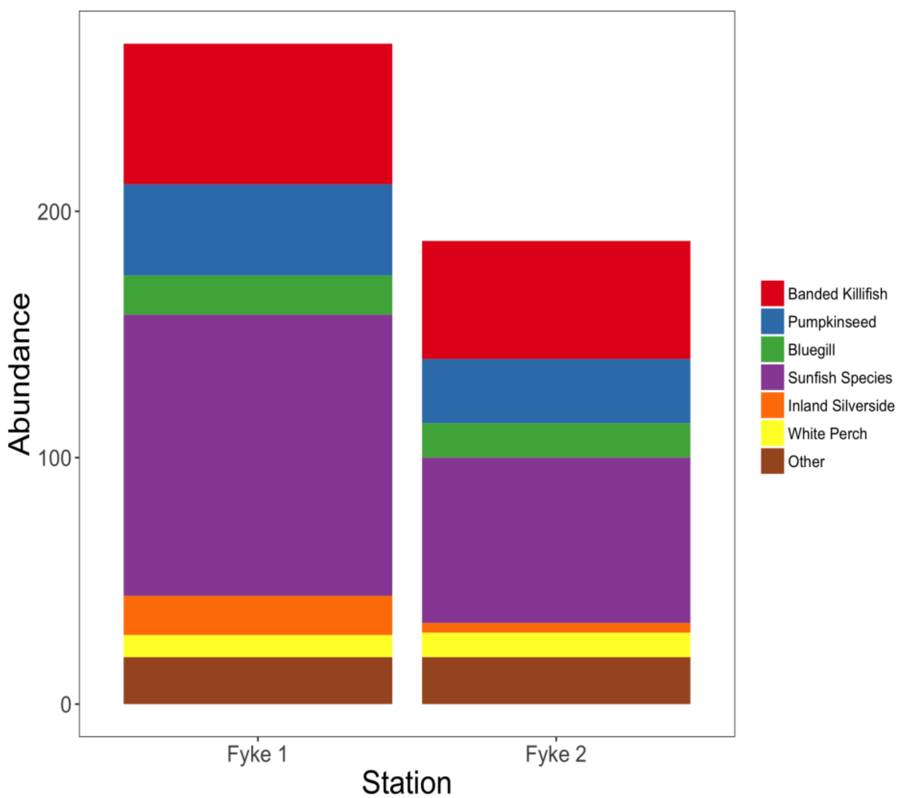


Figure 66. Adult and Juvenile Fishes Collected by Fyke Nets. Dominant Species by Station. 2016.

G. Submersed Aquatic Vegetation – 2016

The map below (Figure 67) depicts the area covered by SAV as determined by the Virginia Institute of Marine Science utilizing aerial imagery for 2016. This map indicates that SAV coverage in 2016 was similar to 2015 and was more extensive than in 2013 and 2014. Again, covering almost all of the inner cove up to about Station 7 which was just outside the SAV area.

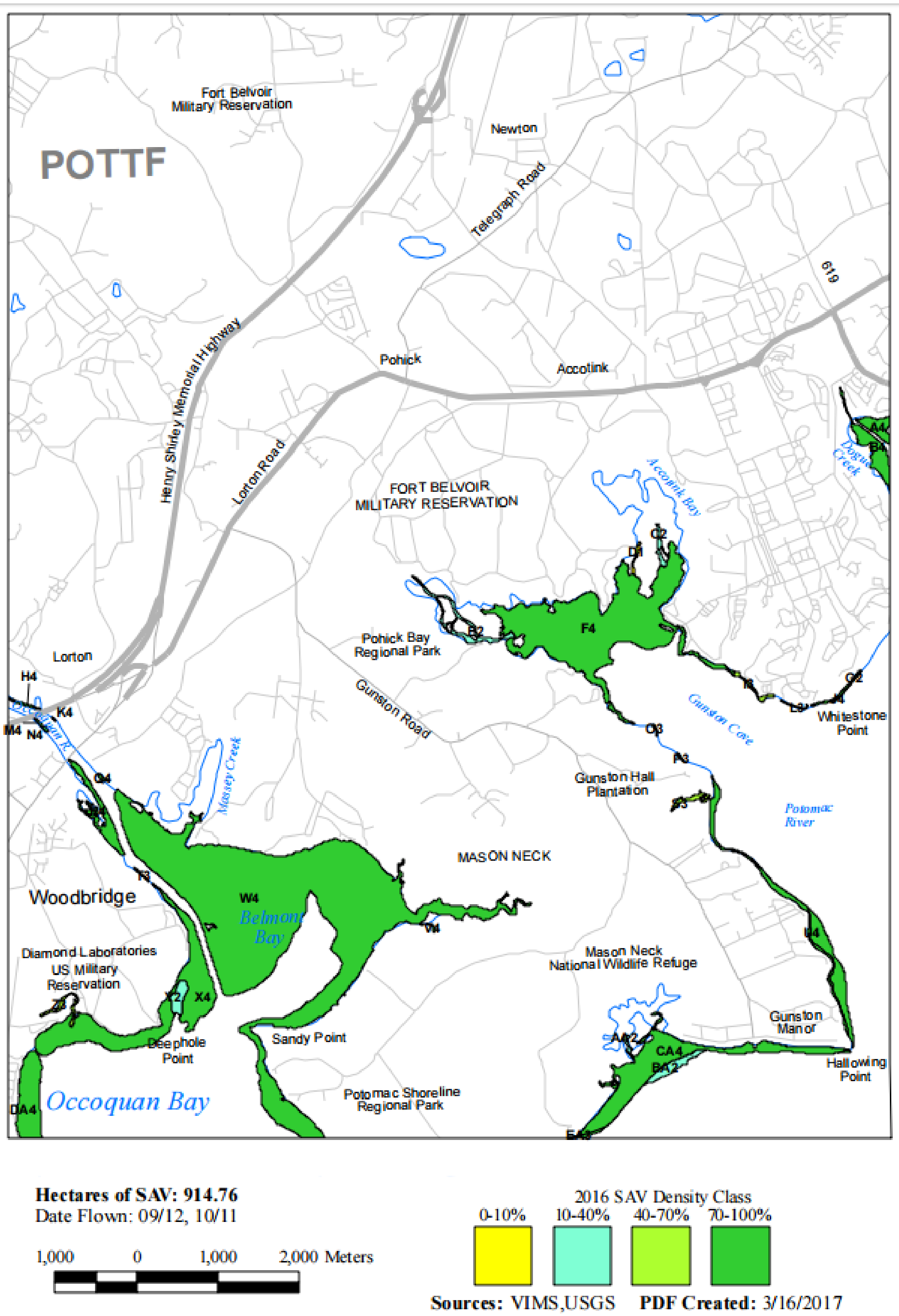


Figure 67. Distribution and density of Submersed Aquatic Vegetation (SAV) in the Gunston Cove area in 2016. VIMS (http://www.vims.edu/bio/sav/index.html).

The distribution of dominant SAV taxa was determined during data mapping cruises (Figure 68). *Hydrilla* was found at most sites and had high coverage at many. *Ceratophyllum* was also found at most sites with generally lower coverage. *Vallisneria* and *Heteranthera* were more restricted in their occurance.

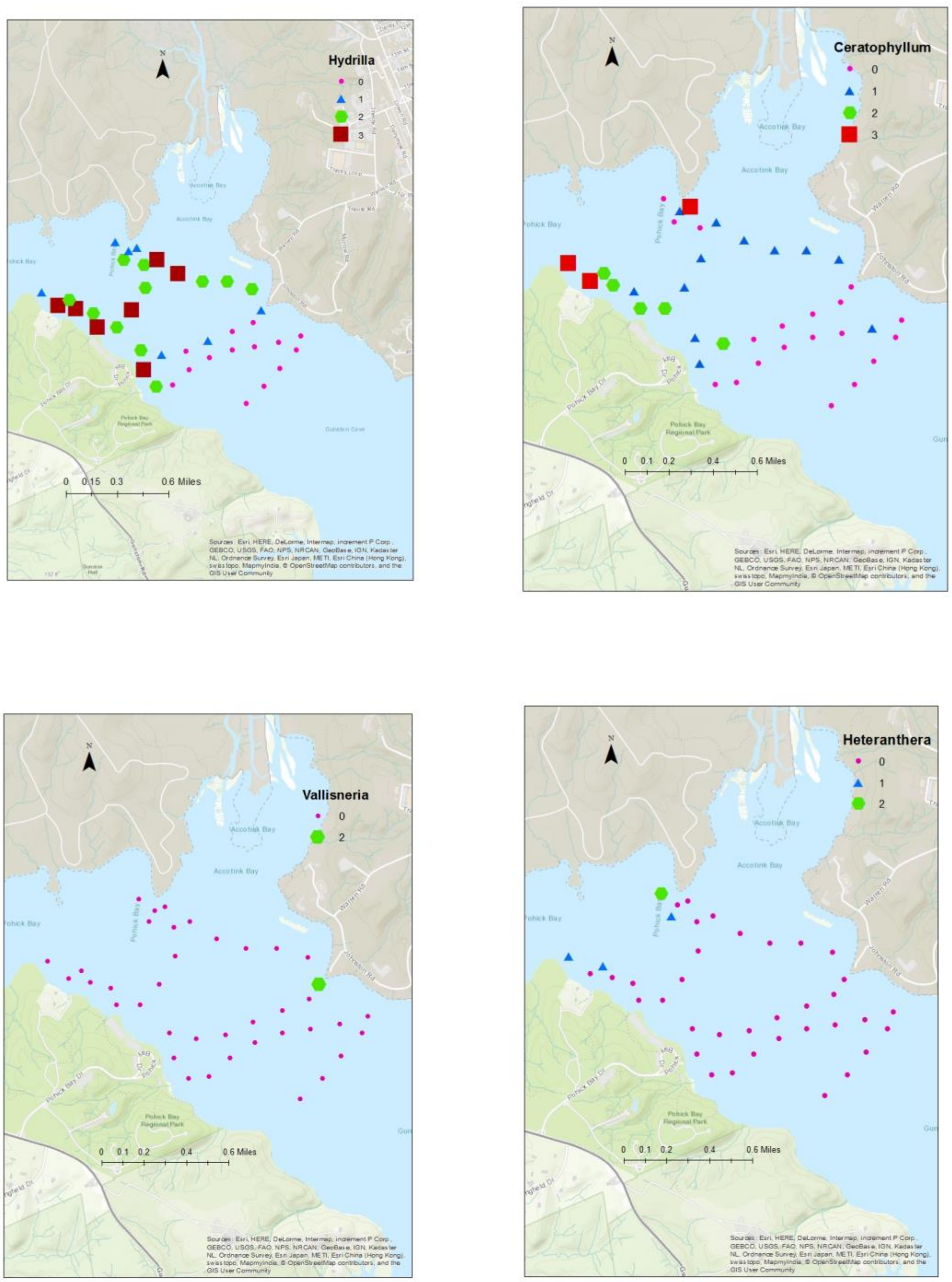


Figure 68. Relative abundance of dominant SAV species determined during data mapping cruises. All points sampled have some kind of symbol. The smallest symbol is for points at which a given species was not found. The largest symbol represents 75-100% coverage.

H. Benthic Macroinvertebrates - 2016

Triplicate petite ponar samples were collected at the cove (Station 7) and river (Station 9) sites on monthly from May to September. As in most previous years, oligochaetes were the most common invertebrates collected in these samples (Figure 69). Chironomids were the second most abundant in the cove, but were found at much lower levels in the river. In 2016 amphipods were the second most abundant taxon at Station 9 with isopods also very common. Gastropods and bivalves were found in roughly even numbers at both sites. Most of the bivalves were the exotic species *Corbicula*, but two native river mussels were also sampled. Other taxa found chiefly in the river were Turbellaria (flatworms), isopods, and Hirundinea (leeches).

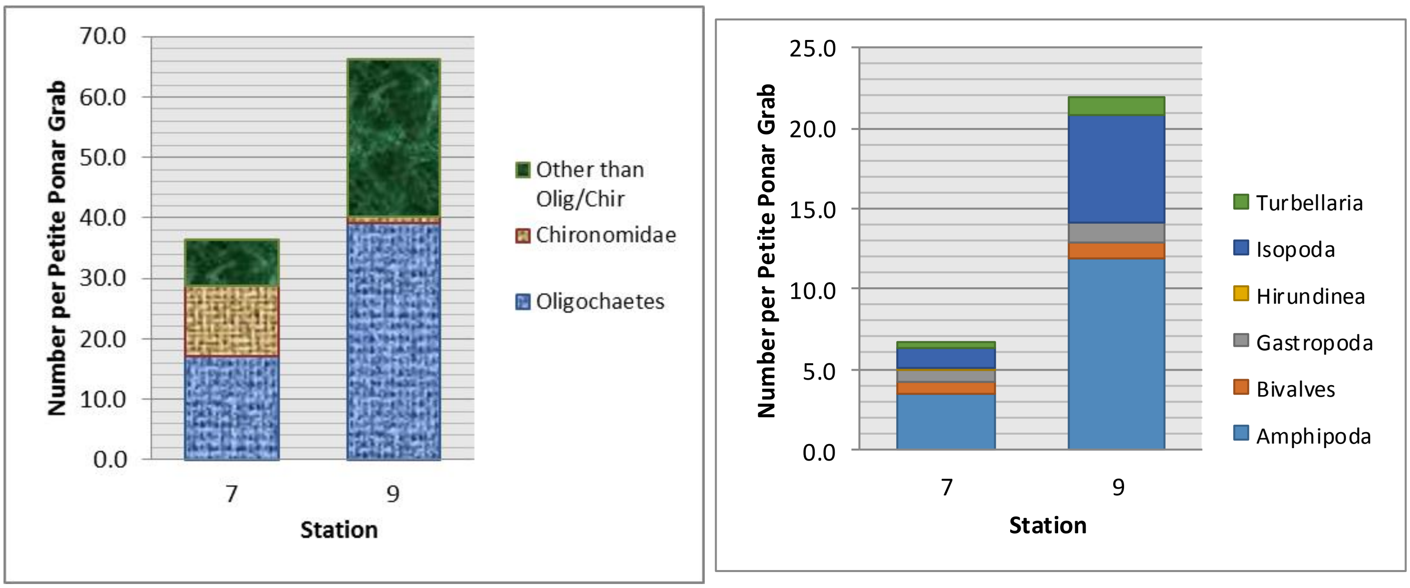


Figure 69. Average abundance of various benthic macroinvertebrate taxa in petite ponar samples collected on four dates in 2016. (a) dominant taxa. (b) "other" group from (a) broken out by taxa.

DISCUSSION

A. 2016 Data

In 2016 air temperature was above average for most of the year including all but one of the months when sampling occurred. July and August were the warmest months, 2°C. Precipitation was well above normal during May, but close to normal in the other months when sampling occurred. The largest daily rainfall total was on May 2 with over 3 cm on top of May 1 with 1.2 cm. This was significant since the early May sampling occurred within two days following this event. Potomac River discharge during 2016 was below normal in March and April, but generally above normal in May and June. July through August had typical decreasing flow interrupted by a couple of short term increases. Accotink Creek flows followed a similar pattern with most sampling months near normal. Throughout the year there were large, short lived flow peaks due to individual storms.

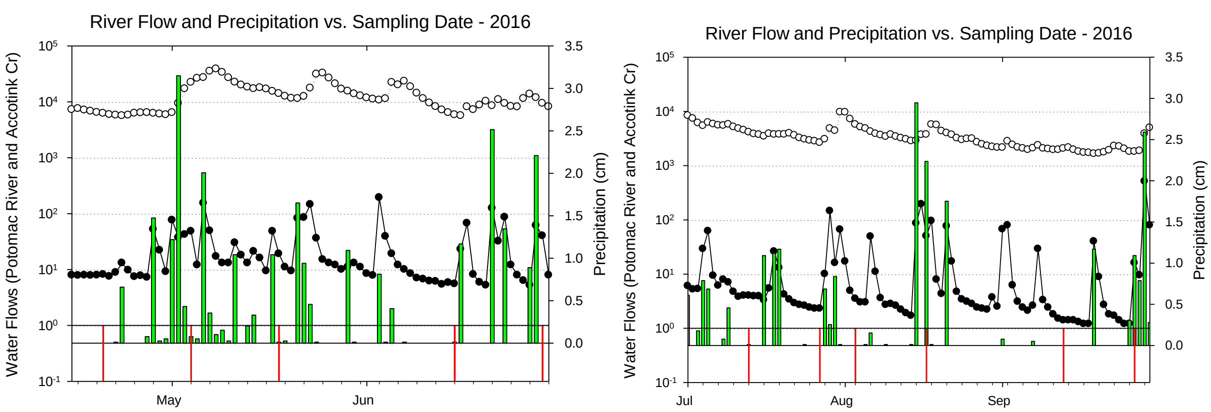


Figure 70. Precipitation (green bars), Accotink Creek flows (solid circles), Potomac River flows (open circles) and water quality/plankton sampling events (red lines at bottom).

Mean water temperature was similar at the two stations reaching a maximum over 30° in late July. Specific conductance declined substantially at both stations in the wake of the early May flow events, then gradually increased through the remainder of the year. Chloride showed a similar pattern, but was consistently somewhat higher in the cove. Dissolved oxygen saturation (DO) was normally substantially higher in the cove than in the river due to photosynthetic activity of phytoplankton and SAV. An exception to this occurred in the wake of the early May flow event when both areas showed a depression in DO which was very marked in the cove. A second, lesser decline in late June was in the wake of a second flow event. Field pH patterns mirrored those in DO: higher values in the cove than the river and strong response to the early May flow event. Total alkalinity was generally higher in the river than in the cove and was fairly constant seasonally. Secchi disk transparency was generally lower in the cove in spring and showed a depression in the early May sampling as well as the late June sampling. By late summer Secchi disk transparency in the cove increased above river values and approached 1.8 m by late September. Light attenuation coefficient and turbidity followed a similar pattern.

Ammonia nitrogen was consistently low in the study area during 2016. All but one value was below the limits of detection which makes analyzing any temporal or spatial trends impossible. Un-ionized ammonia remained below values that would cause toxicity issues, but exact values were not possible due to the high incidence of non-detects on total ammonia. Nitrate values declined seasonally at both sites due to algal and plant uptake and possibly denitrification. By late July nitrate nitrogen in the cove was below detection limits where it remained through the remainder of the year. River nitrate nitrogen levels reached a low of about 0.2 mg/L. Organic nitrogen exhibited substantial variability with a decline in values in the cove through the course of the year. Total phosphorus was similar at both sites and showed little seasonal change. Soluble reactive phosphorus was very low and consistently below detection limits in the cove and higher in the river. N to P ratio declined strongly at both stations reaching a minimum of about 12 in September which is still indicative of P limitation of phytoplankton and SAV. BOD was generally higher in the cove than in the river. TSS was fair constant throughout the year. Peak value in the river as observed in late June; interestingly, the early May flow event did not seem to affect TSS. VSS was also fairly constant between sites and seasonally.

In the cove algal populations as measured by chlorophyll *a* declined strongly in the wake of the early May flow event. A strong rebound was observed in late May followed by a gradual decline and another peak in early August. In the river, the early May decline was observed, but levels recovered only gradually reaching a late May peak. Both cell density and biovolume indicated the flow-induced decline in phytoplankton in early May in the cove and in the river. Values in the cove also showed a decline in late June, the time of the second flow event. The early August peak in the cove in chlorophyll was not seen in the cell count data. Due to a cutback in phytoplankton count frequency, cell counts were not done on the late July sample from the river. Cell density data from the cove was dominated by cyanobacteria, the principal species being *Oscillatoria*. In the river, diatoms dominated cell density data for most of the year, first Pennate 2, then Pennate 1, and finally *Melosira*. In late summer other groups were important with *Anabaena* numerous in August and *Dictyospherium* important in September. Cell biovolume was more evenly distributed among various taxa in the cove than was cell density. In the river cell biovolume was dominated by diatoms with discoid centrics most important in the first half of the year and *Melosira* in the second half.

Rotifers continued to be the most numerous zooplankton in 2016. Rotifer densities were unusually high in April in both areas, but declined dramatically in early May in response to the flow event. Another peak was observed in late June. *Brachionus*, *Filinia*, and *Keratells* shared dominance in the cove; *Filinia* was not common in the river, but *Brachionus* and *Keratella* were. *Bosmina*, a small cladoceran that was often common was only present at low densities in 2016. *Diaphanosoma*, a larger cladoceran was found in both area at moderate densities. Both peaked in late June and then declined after in the wake of the flow event. A subsequent higher peak in *Diaphanosoma* in the river was not found in the cove. Surprisingly, *Daphnia* and *Ceriodaphnia* exhibited their one strong peak in the cove in early May. *Moina* was only found in substantial number is late June in the river. *Leptodora* also seemed to respond positively to the early May flow event in the cove and reached even higher levels in early June in both areas. Copepod nauplii densities reached a peak in both study areas in early June and then declined. A second peak was found in the river in late August. The calanoid copepod *Eurytemora* was very abundant in the cove in early May whereas the river maximum was found in early June. A second calanoid

Diaptomus was restricted to the river at lower levels. Cyclopoid copepods had a strong maximum in early May in the cove and a mid-July maximum in the river.

In 2016 ichthyoplankton was dominated by clupeids, most of which were Gizzard Shad and Alewife, and to a lesser extent, Blueback Herring, American Shad, and Hickory Shad. White Perch was a dominant species as well, with the same relative contribution to the total ichthyoplankton community as Gizzard shad. Striped Bass and Inland Silverside was found in relatively high densities as well. *Morone* species (White Perch and Striped Bass) were mostly found in the Potomac mainstem, confirming their affinity for open water. Other taxa were found in very low densities similar to the previous year. The highest density of fish larvae occurred in mid-May, which was driven by a high density of Clupeid larvae.

A total of 2484 fishes comprising 24 species were collected in all trawl samples combined (Table 5). The dominant species of the fish collected in the trawls was White Perch (69.4%, numerically). In the spring, adult White Perch were primarily caught in the nets while later in the summer juveniles dominated. Other abundant taxa included herring or shad (7.9%), Spottail Shiner (7.7%), Sunfishes (2.2%), and Bay Anchovy (2.1%). Other species were observed sporadically and at low abundances. A total of 35 seine samples were conducted, comprising 3885 fishes of 26 species. This is a little lower than the number of individuals and species collected last year. Similar to last year, the most dominant species in seine catches was Banded Killifish, with a relative contribution to the catch of 56.4%. Other dominant species (with >5% of relative abundance) were White Perch (10.2%) followed by Inland Silverside (7.1%), eastern Silvery Minnow (6.2%), and Alosa sp. (5.2%). In 2016 we collected a total number of 456 specimens of 15 species in the two fyke nets, which is a little bit less than last year. While Banded Killifish is abundant here as well (23% of the catch), the fyke nets show a high contribution of sunfishes too.

The coverage of submersed aquatic vegetation (SAV) in 2016 was similar to recent years. For the first year, species distribution was mapped. The exotic plant *Hydrilla* was the most dense and widespread species, but the native species *Ceratophyllum* (coontail) was also widespread. As in most previous years, oligochaetes were the most common invertebrates collected in ponar samples in 2016. Chironomids were the second most abundant in the cove, but were found at much lower levels in the river. Amphipods were the second most abundant taxon at Station 9 with isopods also very common.

B. Water Quality Trends: 1983-2016

To assess long-term trends in water quality, data from 1983 to 2016 were pooled into two data files: one for Mason data and one for Noman Cole laboratory data. Then, subgroups were selected based on season and station. For water quality parameters, we focused on summer (June-September) data as this period is the most stable and often presents the greatest water quality challenges and the highest biological activity and abundances. We examined the cove and river separately with the cove represented by Station 7 and the river by Station 9. We tried several methods for tracking long-term trends, settling on a scatterplot with LOWESS trend line. Each observation in a particular year is plotted as an open circle on the scatterplot. The LOWESS (locally weighted sum of squares) line is drawn by a series of linear regressions moving through the years. We also calculated the Pearson correlation coefficient and performed linear regressions to test for statistical significance of a linear relationship over the entire period of record (Tables 14 and 15). This was similar to the analysis performed in previous reports.

Table 14
Correlation and Linear Regression Coefficients
Water Quality Parameter vs. Year for 1984-2016
GMU Water Quality Data
June-September

	Station 7			Station 9		
Parameter	Corr. Coeff.	Reg. Coeff.	Signif.	Corr. Coeff.	Reg. Coeff.	Signif.
Temperature	0.206	0.061	0.001	0.130	0.034	0.037
Conductivity, standardized to 25°C	0.191	2.38	0.001	0.044	-----	NS
Dissolved oxygen, mg/L	0.046	-----	NS	0.200	0.026	0.001
Dissolved oxygen, percent saturation	0.026	-----	NS	0.232	0.379	<0.001
Secchi disk depth	0.719	1.87	<0.001	0.379	0.619	<0.001
Light attenuation coefficient	0.681	0.096	<0.001	0.204	0.021	0.004
pH, Field	0.163	-0.011	0.011	0.203	0.010	0.003
Chlorophyll, depth-integrated	-0.600	-3.88	<0.001	0.254	-0.696	<0.001
Chlorophyll, surface	0.616	-3.96	<0,001	0.242	-0.776	<0.001

For Station 7, n=289-304 except pH, Field where n=242 and Light attenuation coefficient where n=226
For Station 9, n=247-261 except pH, Field where n=209 and Light attenuation coefficient where n=196.

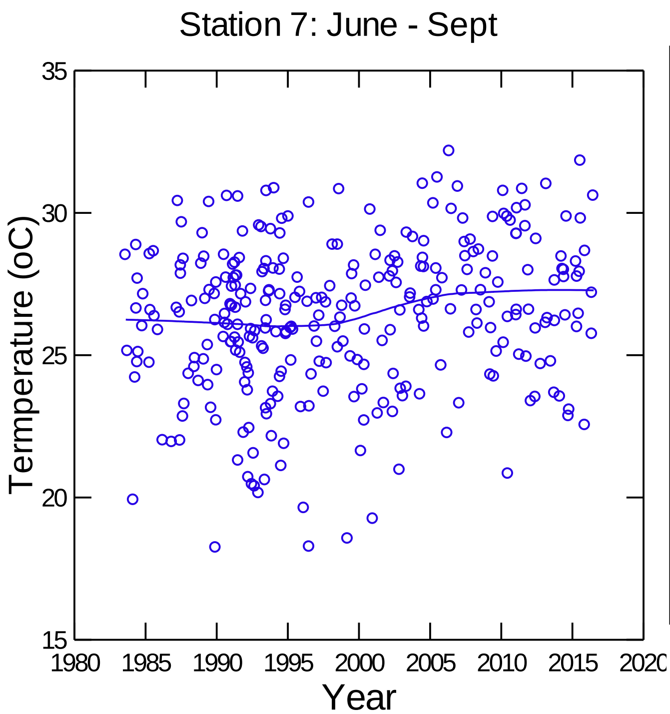
Significance column indicates the probability that a correlation coefficient this large could be due to chance alone. If this probability is greater than 0.05, then NS (not significant) is indicated.

Table 15
Correlation and Linear Regression Coefficients
Water Quality Parameter vs. Year for 1983-2016
Fairfax County Environmental Laboratory Data
June-September

Parameter	Station 7 Corr. Coeff.	Station 7 Reg. Coeff.	Station 7 Signif.	Station 9 Corr. Coeff.	Station 9 Reg. Coeff.	Station 9 Signif.
Chloride	0.029	-----	NS	0.012	-----	NS
Lab pH	0.482	-0.033	<0.001	0.264	-0.013	<0.001
Alkalinity	0.092	0.113	0.048	0.308	0.412	<0.001
BOD	0.644	-0.168	<0.001	0.420	-0.046	<0.001
Total Suspended Solids	0.342	-0.919	<0.001	0.158	-0.172	<0.001
Volatile Suspended Solids	0.401	-0.624	<0.001	0.367	-0.132	<0.001
Total Phosphorus	0.552	-0.004	<0.001	0.278	-0.0009	<0.001
Soluble Reactive Phosphorus	0.109	-0.0001	0.020	0.055	-----	NS
Ammonia Nitrogen	0.305	-0.017	<0.001	0.309	-0.003	<0.001
Un-ionized Ammonia Nitrogen	0.331	-0.004	<0.001	0.324	-0.0003	<0.001
Nitrite Nitrogen	0.420	-0.003	<0.001	0.153	-0.001	0.002
Nitrate Nitrogen	0.585	-0.035	<0.001	0.674	-0.040	<0.001
Organic Nitrogen	0.563	-0.047	<0.001	0.326	-0.011	<0.001
N to P Ratio	0.312	-0.354	<0.001	0.556	-0.604	<0.001

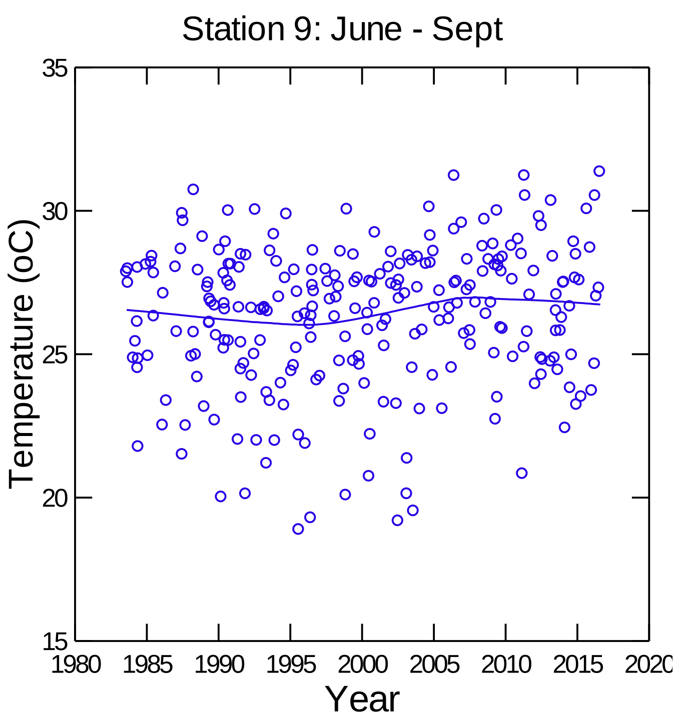
For Station 7, n=431-474 except Nitrite Nitrogen where n=396
For Station 9, n=432-482 except Nitrite Nitrogen where n = 396.

Significance column indicates the probability that a correlation coefficient this large could be due to chance alone. If this probability is greater than 0.05, then NS (not significant) is indicated.



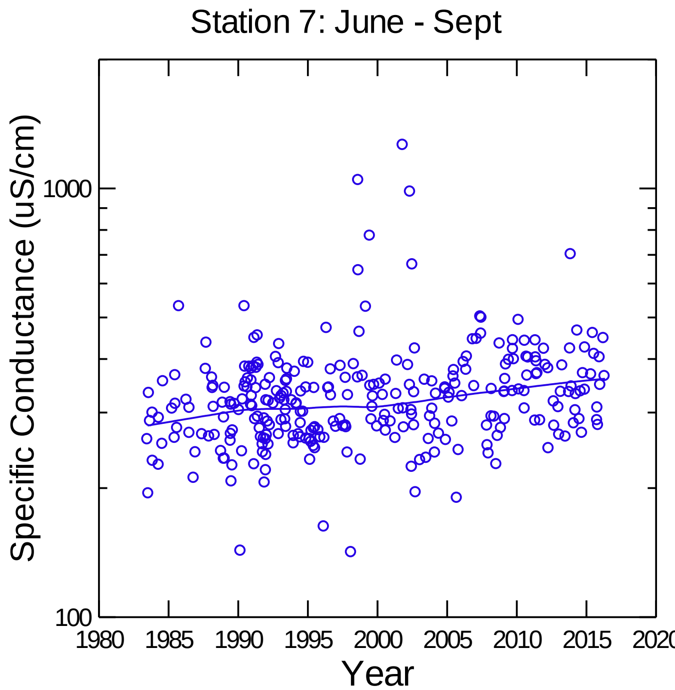
Water temperatures during the summer months generally varied between 20ºC and 30ºC over the study period (Figure 71). The LOWESS curve indicated an average of about 26°C during the period 1984-2000 with a slight upward trend in the last few years to about 27ºC. Linear regression analysis indicated a significant linear trend in water temperature in the cove when the entire period of record is considered (Table 14). The slope of this relationship is 0.06°C/year.

Figure 71. Long term trend in Water Temperature (GMU Field Data). Station 7. Gunston Cove.



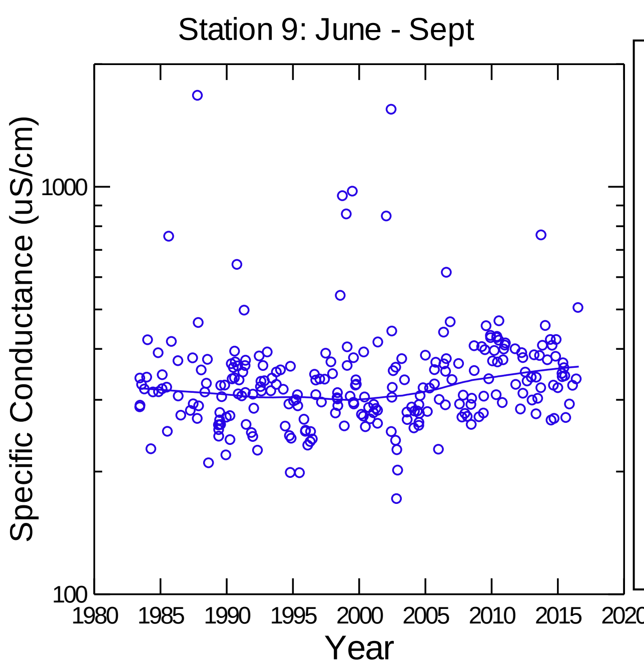
In the river summer temperatures have been similar to those in the cove with fewer readings above 30°C in the river (Figure 72). For the first time in 2016, the long term trend exhibited a significant increase, but at a lower rate (0.03°C/year) (Table 14).

Figure 72. Long term trend in Water Temperature (GMU Field Data). Station 9. Gunston Cove.



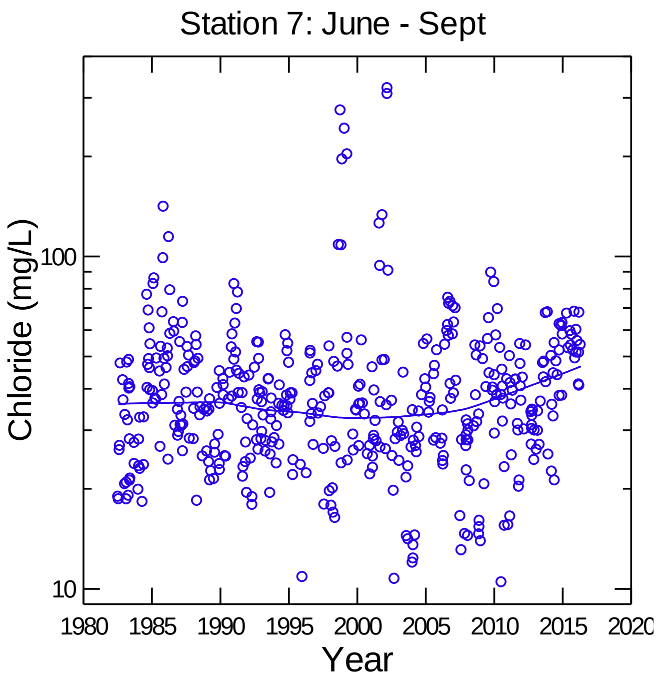
Specific conductance was generally in the range 200-500 μS/cm over the study period (Figure 73). Some significantly higher readings have been observed sporadically. A slight increase in specific conductance was suggested by the LOWESS line over the study period. This was confirmed by linear regression analysis which found a significant linear increase of 2.4 μS/cm per year over the long term study period (Table 14). This would yield a total increase of 77 uS/cm over the entire study period.

Figure 73. Long term trend in Specific Conductance (GMU Field Data). Station 7. Gunston Cove.



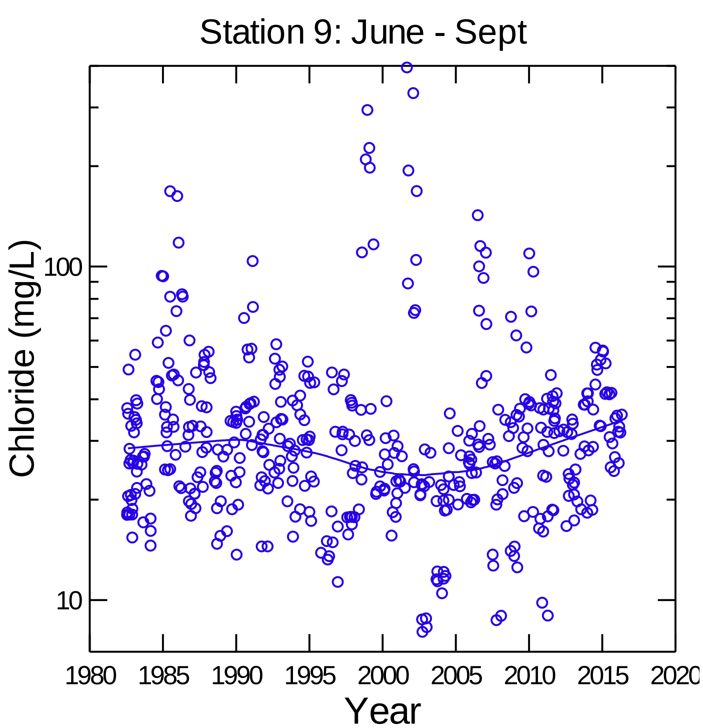
Conductivity values in the river were in the same general range as in the cove (Figure 74). Most values were between 200 and 500 μS/cm with a few much higher values. These higher values are probably attributable to intrusions of brackish water from downstream during years of low river flow. Linear regression did not reveal a significant trend in river conductivity (Table 14). However, the trend line has moved up in recent years from about 300 μS/cm to nearly 400 μS/cm.

Figure 74. Long term trend in Specific Conductance (GMU Field Data). Station 9. River mainstem.



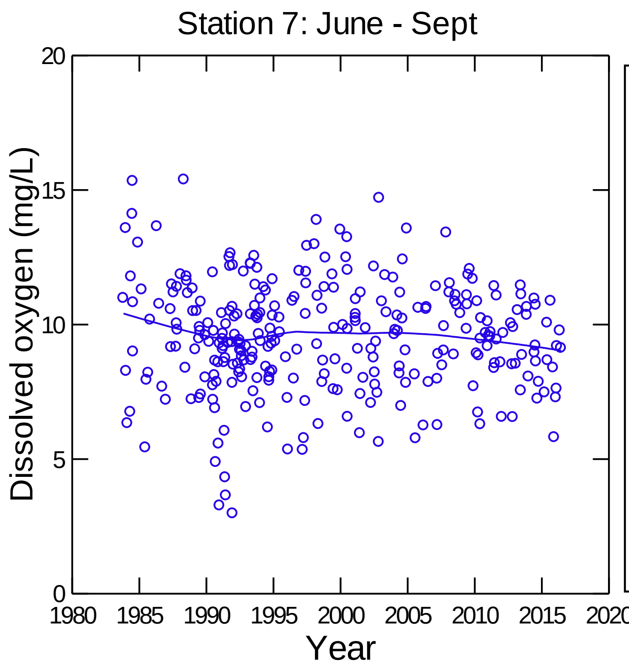
Chloride levels were clustered in a relatively narrow range of 20-60 mg/L for the entire study period (Figure 75). Higher values observed in some years were probably due to the estuarine water intrusions that occur in dry years. The trend line is nearly flat and a linear regression was not statistically significant (Table 15).

Figure 75. Long term trend in Chloride (Fairfax County Lab Data). Station 7. Gunston Cove.



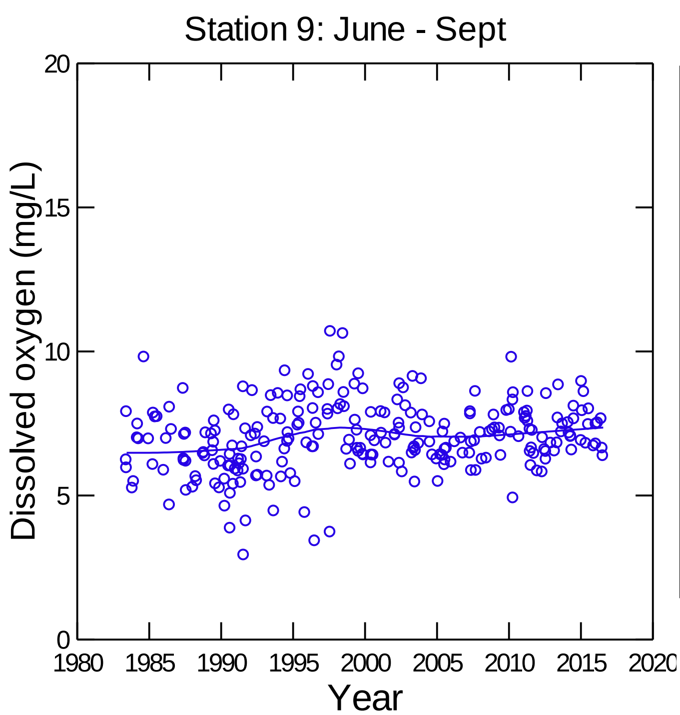
Chloride in the river has been slightly more variable than that in the cove, but in the same general range (Figure 76). The higher readings are again due to brackish water intrusions in dry years. A slight trend of increasing values in the 1980's followed by decreases in the 1990's and increases since 2005 was suggested by the LOWESS trend line. However, temporal linear regression analysis was not statistically significant (Table 12).

Figure 76. Long term trend in Chloride (Fairfax County Lab Data). Station 9. River mainstem.



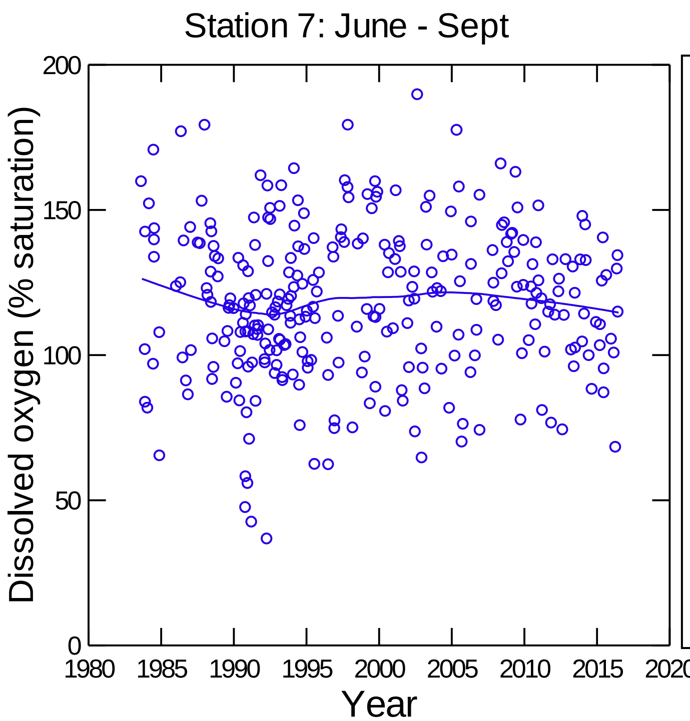
Dissolved oxygen in the cove has generally been in the range 7-13 mg/L during the summer months (Figure 77). A slight downward trend was observed through 1990, but since then the trend line has flattened, suggesting little consistent change and a mean of about 10 mg/L. In the cove dissolved oxygen (mg/L) did not exhibit a significant linear trend over the long term study period (Table 14).

Figure 77. Long term trend in Dissolved Oxygen, mg/L (GMU Data). Station 7. Gunston Cove.



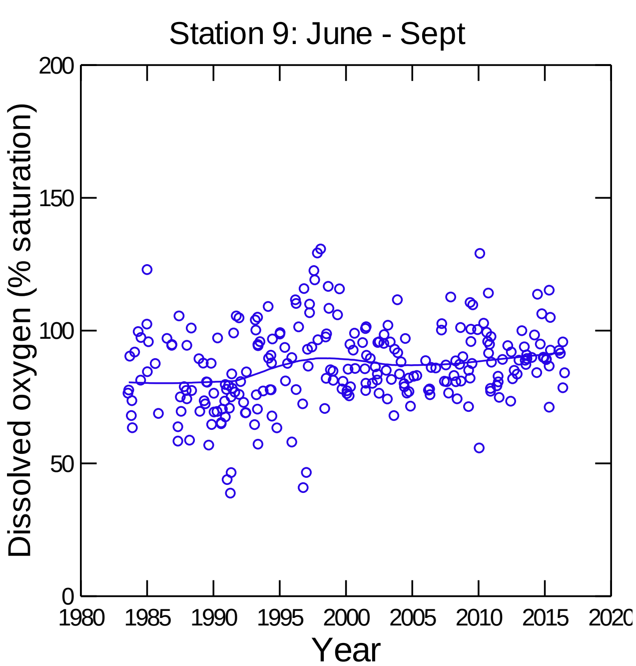
In the river dissolved oxygen values generally were in the range 5-9 mg/L over the long term study period (Figure 78). The LOWESS trend line some subtle changes from year to year, but little consistent pattern. The linear regression analysis over the entire period indicated a significant positive trend with slope of 0.026 mg/L per year or 0.8 mg/L over the period of record (Table 14).

Figure 78. Long term trend in Dissolved Oxygen, mg/L (GMU Data). Station 9. River mainstem.



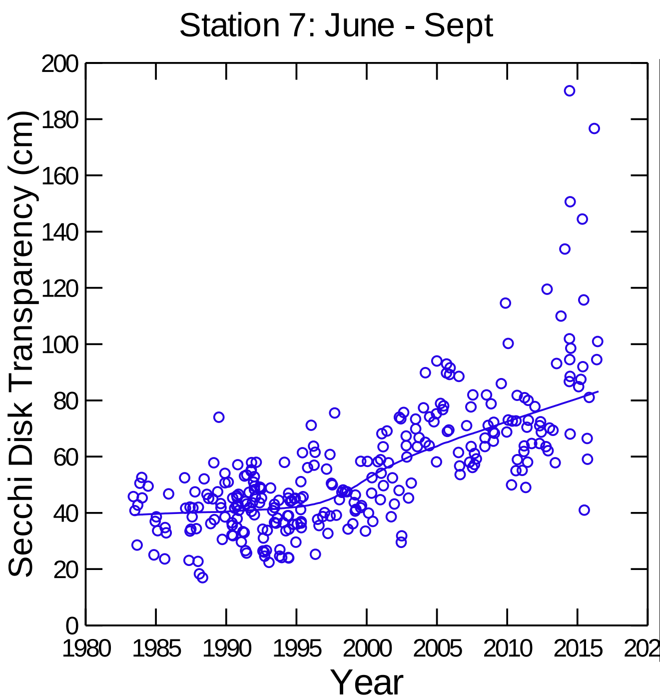
Dissolved oxygen was generally in the range 100-150% saturation in the cove over the long term study period indicating the importance of photosynthesis in the cove (Figure 79). A decline was indicated by the trend line through 1990 followed by a slight recovery in subsequent years. Percent saturation DO did not exhibit a significant linear trend over the long term study period (Table 14). 2016 values fell around the trend line.

Figure 79. Long term trend in Dissolved Oxygen, % saturation (GMU Data). Station 7. Gunston Cove.



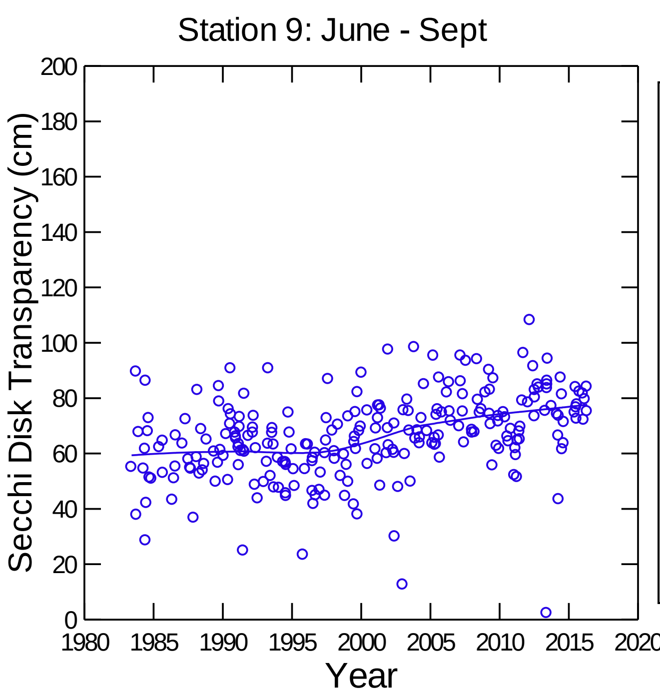
In the river dissolved oxygen was generally less than 100% indicating that photosynthesis was much less important in the river than in the cove and that respiration dominated (Figure 80). The trend line showed a very gradual increase which was statistically significant as indicated by regression analysis with a slope of 0.3% per year or about 12% over the course of the study (Table 14). 2016 readings were near the long term trend line. Despite this increase river DO was still below cove DO in general.

Figure 80. Long term trend in Dissolved Oxygen, % saturation (GMU Data). Station 9. Gunston Cove.



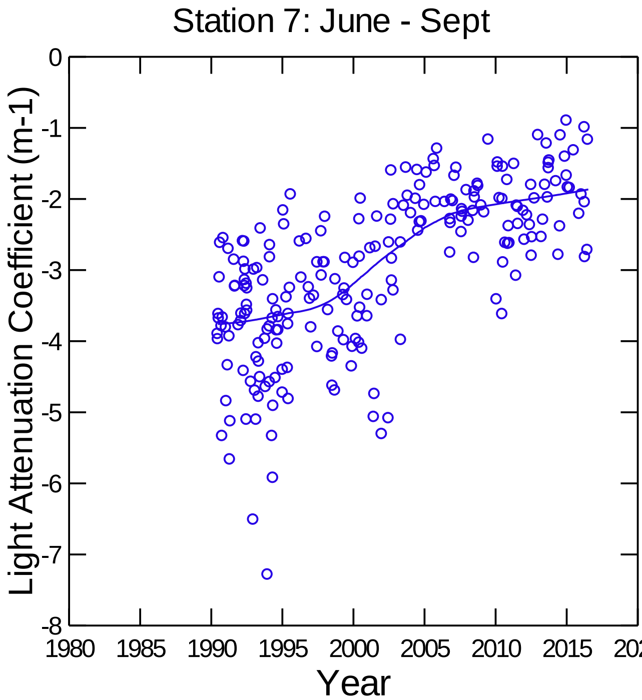
Secchi disk transparency is a measure of water clarity. Secchi disk was fairly constant from 1984 through 1995 with the trend line at about 40 cm (Figure 81). Since 1995 there has been a steady increase in the trend line from 40 cm to 85 cm in 2016. Linear regression was highly significant with a predicted increase of 1.87 cm per year or a total of nearly 60 cm over the study period (Table 14). In 2016 most readings were above the trend line as in 2014 and 2015 which is forcing the trend line up.

Figure 81. Long term trend in Secchi Disk Transparency (GMU Data). Station 7. Gunston Cove.



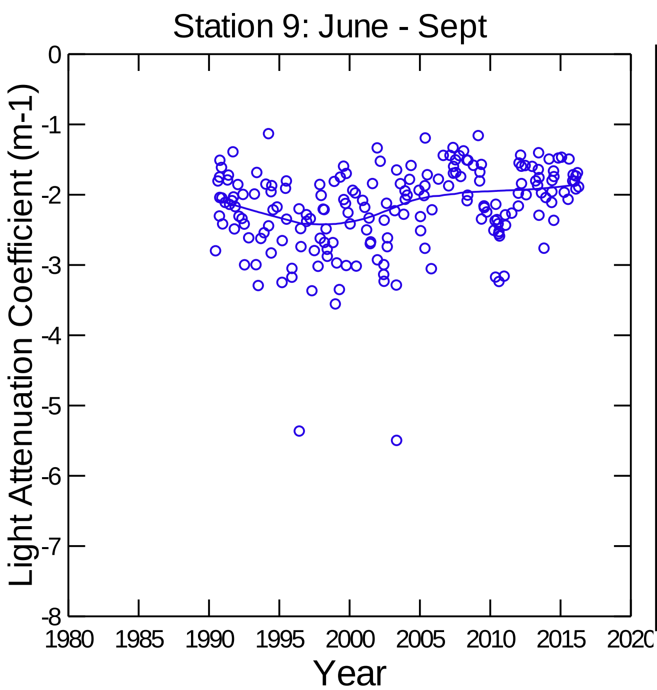
In the river Secchi depth was somewhat greater than in the cove in the 1980's (Figure 82). The trend line was fairly constant at about 60 cm until about 2000. A rise to about 75 cm was observed by 2005 where it has remained. Linear regression revealed a significant increase of 0.62 cm per year with total increase of 20 cm predicted over of the study period (Table 14).

Figure 82. Long term trend in Secchi Disk Transparency (GMU Data). Station 9. River mainstem.



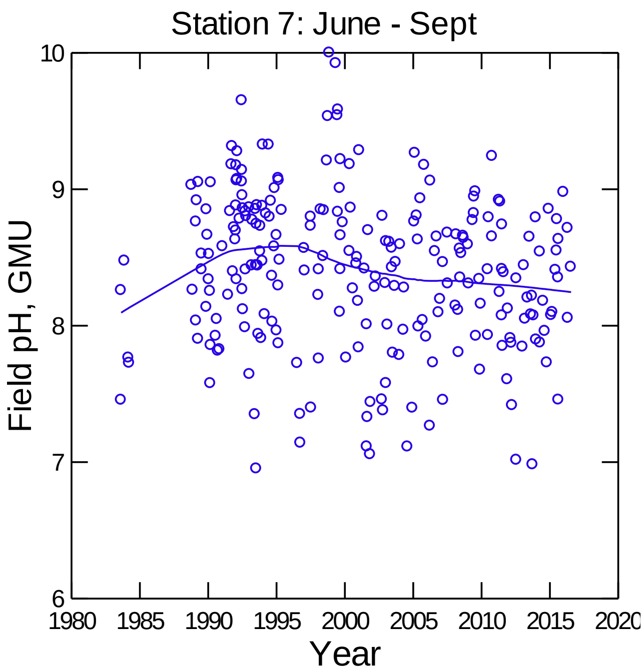
Light attenuation coefficient, another measure of water clarity, reinforces the conclusion that water clarity has been improving in the cove since 1995 (Figure 83). Trend line for the coefficient rose from about -4 to -2 m^{-1} during this time. And values for 2015 and 2016 were generally above -2. Consistent with this was the regression analysis which revealed a significant linear increase in light attenuation coefficient over the period 1991-2016 with a slope of 0.1 per year yielding a prediction that light attenuation improved by about 2.3 units over this period (Table 14).

Figure 83. Long term trend in Light Attenuation Coefficient (GMU Data). Station 7. Gunston Cove.



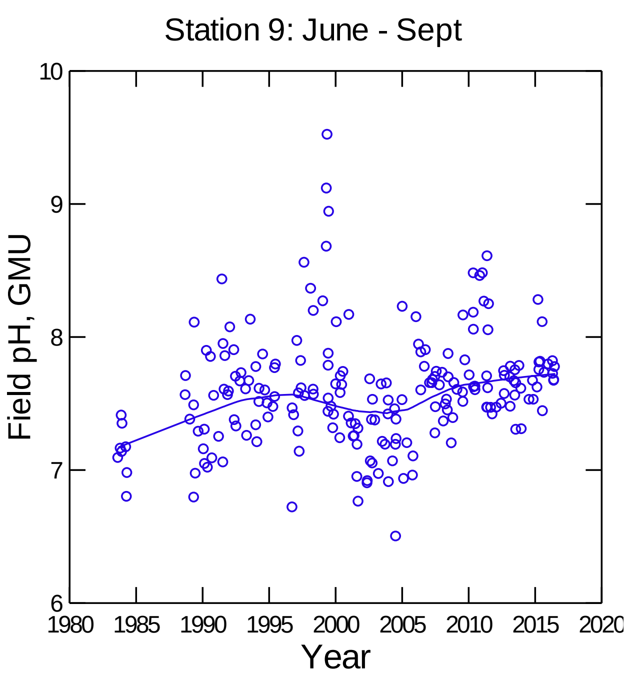
In the river light attenuation coefficient suggested a decline in light transparency between 1991 and 1997 followed by an increase through about 2008 (Figure 84). Between 2008 and 2016 the trend line indicates that light transparency has held fairly constant. Regression indicated a weak, but significant linear trend over the entire period (Table 14). The regression coefficient was 0.02 yielding a change of 0.6 units over the period, much less than found at Station 7.

Figure 84. Long term trend in Light Attenuation Coefficient (GMU Data). Station 9. River mainstem.



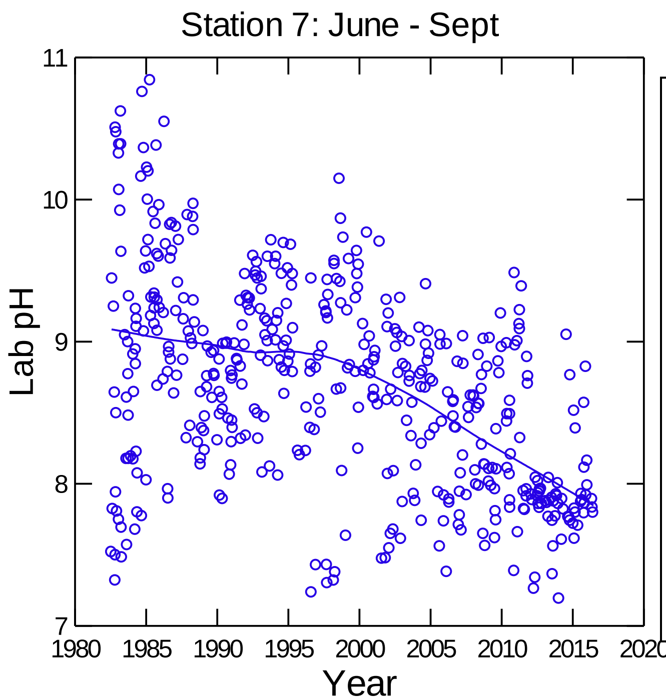
Field pH has not been measured as consistently over the entire study period as other parameters. Cove values have generally been in the 8-9 range. There is a clear trend of decreasing values since 1995 (Figure 85). Linear regression analysis now gives some evidence of a declining linear trend with a slope of -0.011 units per year when the entire study period was considered (Table 14).

Figure 85. Long term trend in Field pH (GMU Data). Station 7. Gunston Cove.



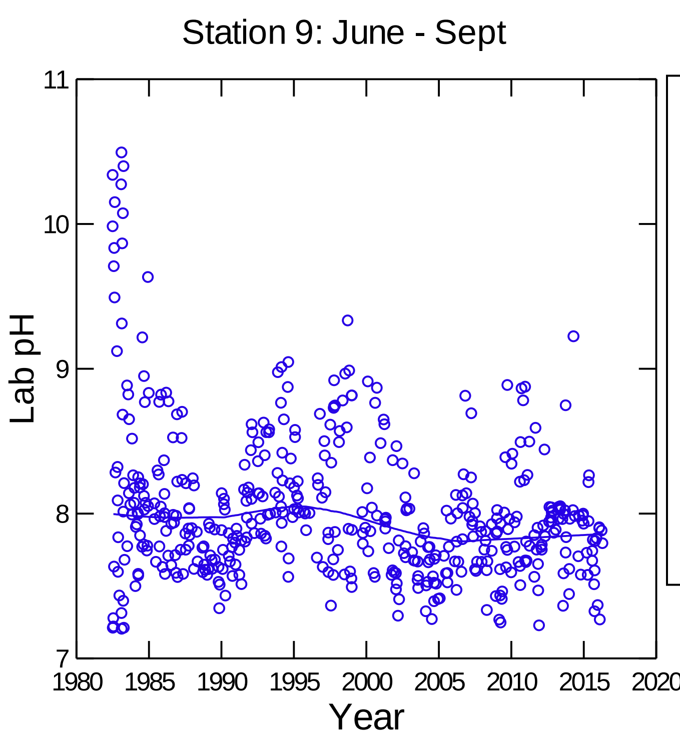
In the river a different pattern has been observed over this period (Figure 86). pH in the river has been consistently lower by about 1 pH unit than in the cove. If anything the trend line has shown a tendency to increase. When all years were considered, field pH in the river shows a significant increase at a rate of 0.01 units per year (Table 14).

Figure 86. Long term trend in Field pH (GMU Data). Station 9. River mainstem.



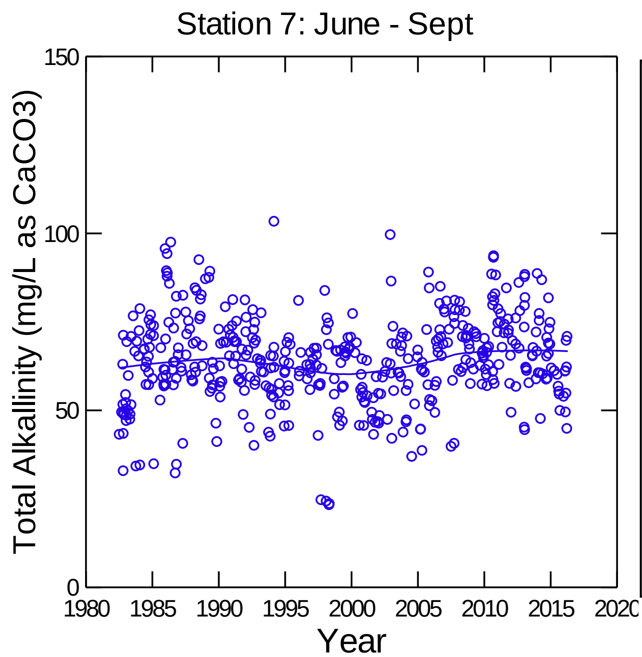
Lab pH as measured by Fairfax County personnel was generally in the range 8 to 10 over most of the long term study period (Figure 87). Since about 1997 a decline is very evident with the trend line decreasing from about 9.1 to about 7.8. Linear regression indicates a significant decline in lab pH over the study period at a rate of about 0.03 pH units per year or a total of 1.0 units over the study period (Table 15). 2016 data were above the trend line.

Figure 87. Long term trend in Lab pH (Fairfax County Lab Data). Station 7. Gunston Cove.



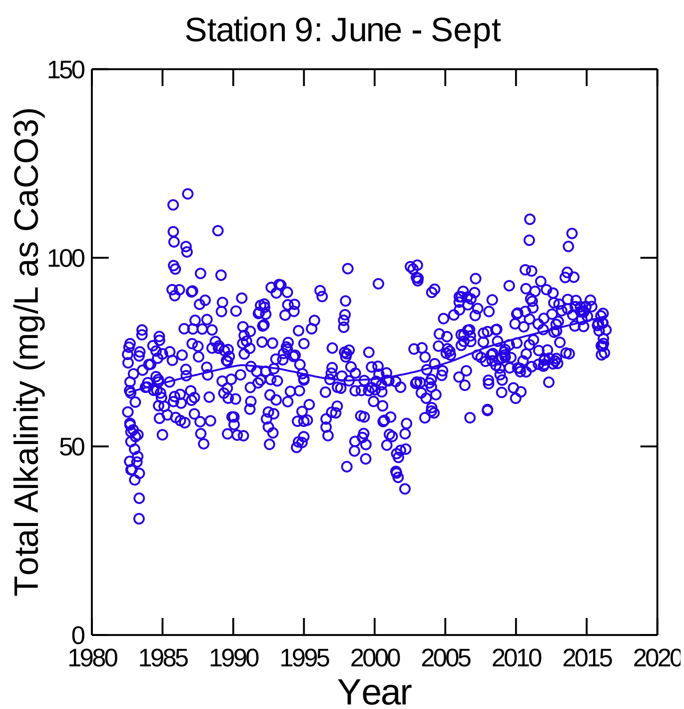
In the river, long term pH trends as measured by Fairfax County lab personnel indicate that most values fell between 7 and 8.5 (Figure 88). The trend line has increased and decreased slightly over the years. pH in the river showed a significant linear decline with a rate of 0.013 per year yielding a total decline of 0.43 units over the long term study period (Table 15).

Figure 88. Long term trend in Lab pH (Fairfax County Lab Data). Station 9. Potomac mainstem.



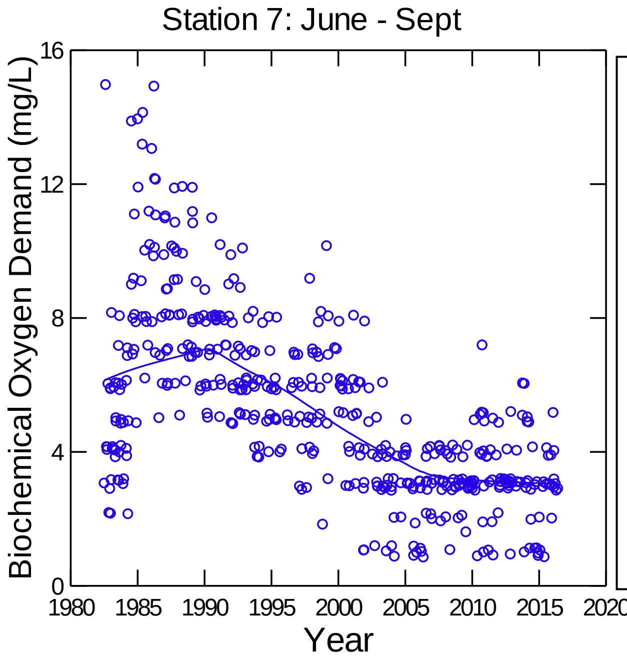
Total alkalinity as measured by Fairfax County personnel exhibited little variation early on and a slow increase since 2000 (Figure 89). The trend line at 2016 was slightly higher than it was in 1983. Overall, a very weak statistically significant linear trend has developed in total alkalinity in the cove over this period with a slope of 0.11 mg/L per year yielding a projected increase of about 3.5 mg/L over the entire study period (Table 15).

Figure 89. Long term trend in Total Alkalinity (Fairfax County Lab Data). Station 7. Gunston Cove.



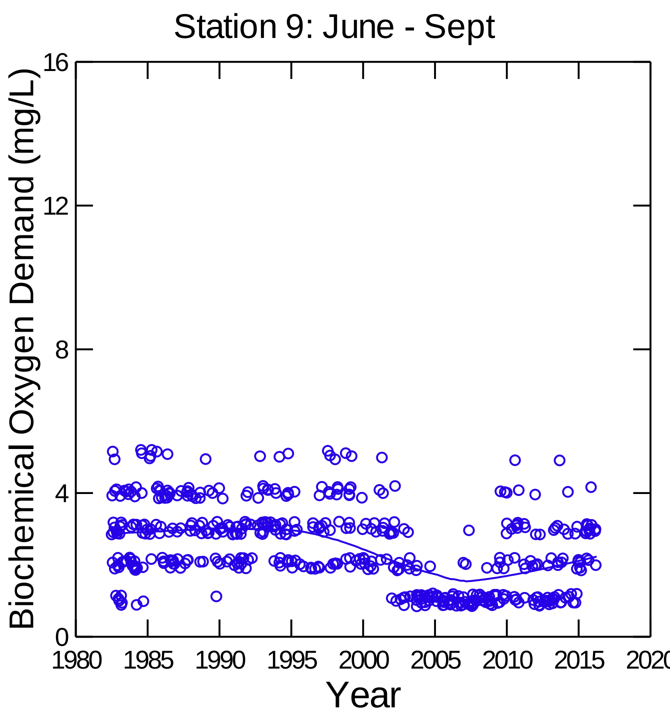
In the river a similar pattern has been observed over the three decades with an even clearer recent increase (Figure 90). There is a significant linear trend over the period with a slope of 0.41 mg/L suggesting a modest increase of about 13 mg/L over the entire study period (Table 15).

Figure 90. Long term trend in Total Alkalinity (Fairfax County Lab Data). Station 9. Potomac mainstem.



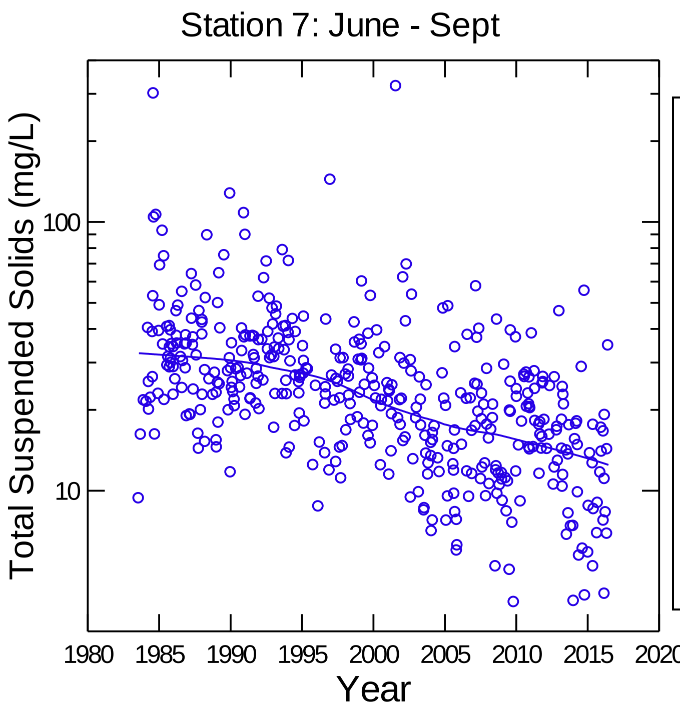
Biochemical oxygen demand has shown a distinct pattern over the long term study period in Gunston Cove (Figure 91). In the 1980's the trend line rose from about 6 mg/L to 7 mg/L by 1989. Since then there has been a steady decline such that the trend line has dropped back to about 3 mg/L. BOD has shown a significant linear decline over the entire study period at a rate of 0.17 mg/L per year yielding a net decline of about 5.4 mg/L over the entire period of record (Table 15).

Figure 91. Long term trend in Biochemical Oxygen Demand (Fairfax County Lab Data). Station 7. Gunston Cove.



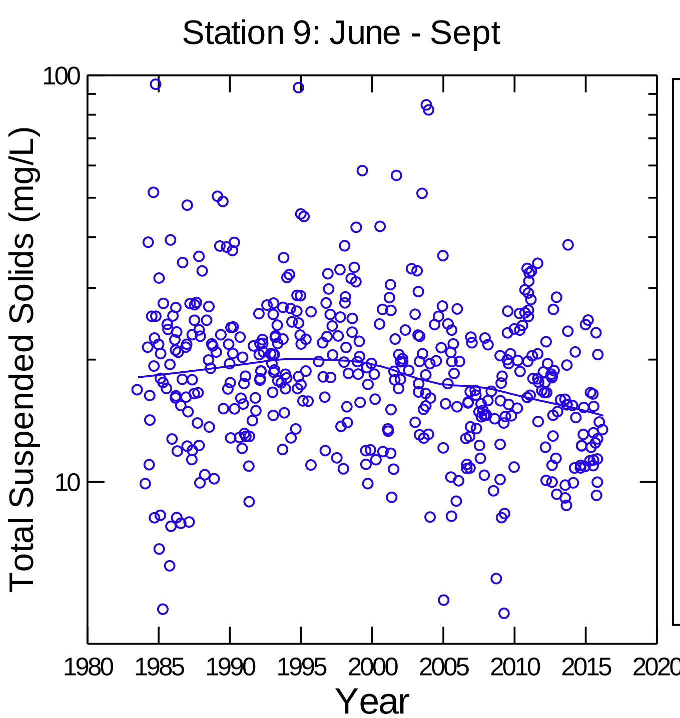
In the river biochemical oxygen demand exhibited a less distinct pattern through the mid 1990's (Figure 92). However, since that time it has decreased steadily to a trend line value of about 1.5 mg/L. BOD in the river has exhibited a significant linear decrease at a rate of 0.046 units when the entire period of record was considered (Table 15). This would project to an overall decrease of 1.5 units.

Figure 92. Long term trend in Biochemical Oxygen Demand (Fairfax County Lab Data). Station 9. Potomac mainstem.



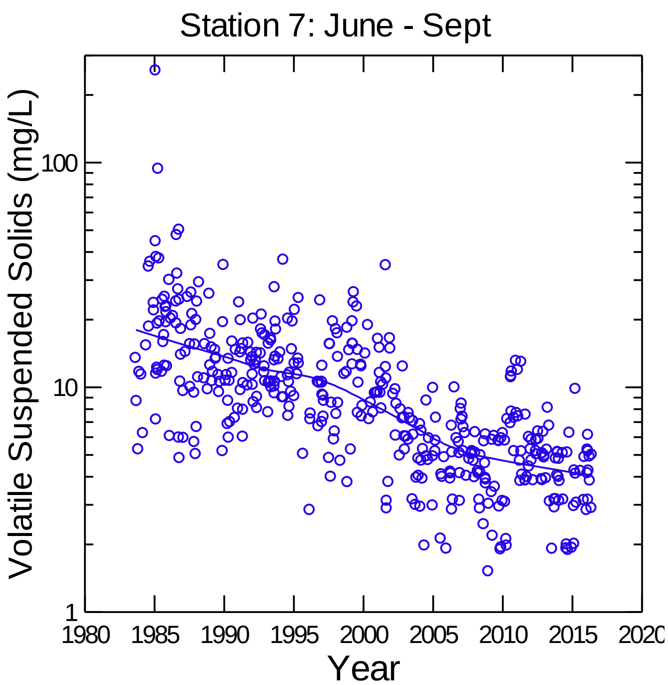
Total suspended solids (TSS) has shown a great deal of variability over the long term study period. Nonetheless, a decreasing trend in TSS is clear in the cove with the trend line decreasing from about 32 mg/L in 1983 to about 12 mg/L in 2016 (Figure 93). Linear regression was significant indicating a decline of 0.92 mg/L per year yielding a total decline of 29.5 mg/L since 1984 (Table 15).

Figure 93. Long term trend in Total Suspended Solids (Fairfax County Lab Data). Station 7. Gunston Cove.



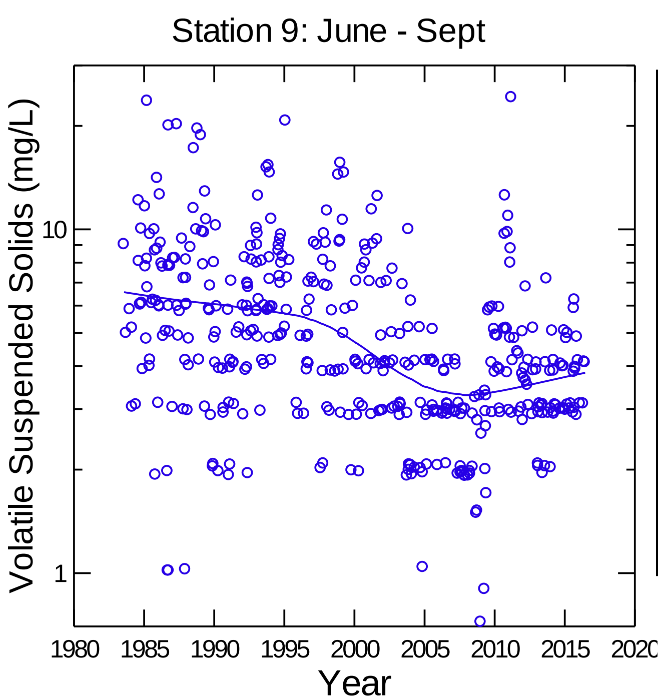
In the river TSS trends have not been as apparent (Figure 94). While much higher values have been observed sporadically, the LOWESS line remained steady at about 18-20 mg/L through most of the period with a slight decrease to about 15 mg/L suggested recently. In the river TSS exhibited a significant linear decline over the period of record at a rate of about 0.17 units per year yielding a total decline of about 5.4 mg/L over the entire study period (Table 15).

Figure 94. Long term trend in Total Suspended Solids (Fairfax County Lab Data). Station 9. Potomac mainstem.



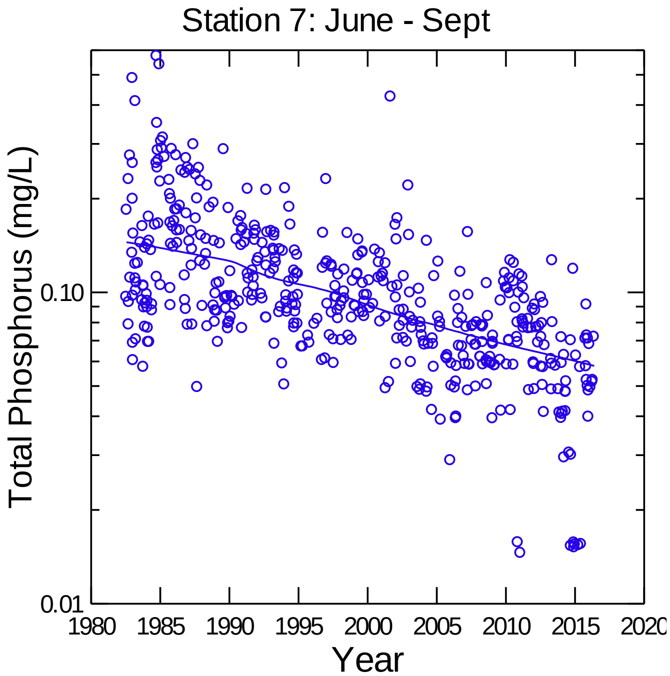
Volatile suspended solids have consistently declined over the study period in the cove (Figure 95). The LOWESS trend line has declined from 20 mg/L in 1984 to about 3 mg/L in 2016. VSS has demonstrated a significant linear decline at a rate of 0.63 mg/L per year or a total of 20 mg/L over the study period (Table 15).

Figure 95. Long term trend in Volatile Suspended Solids (Fairfax County Lab Data). Station 7. Gunston Cove.



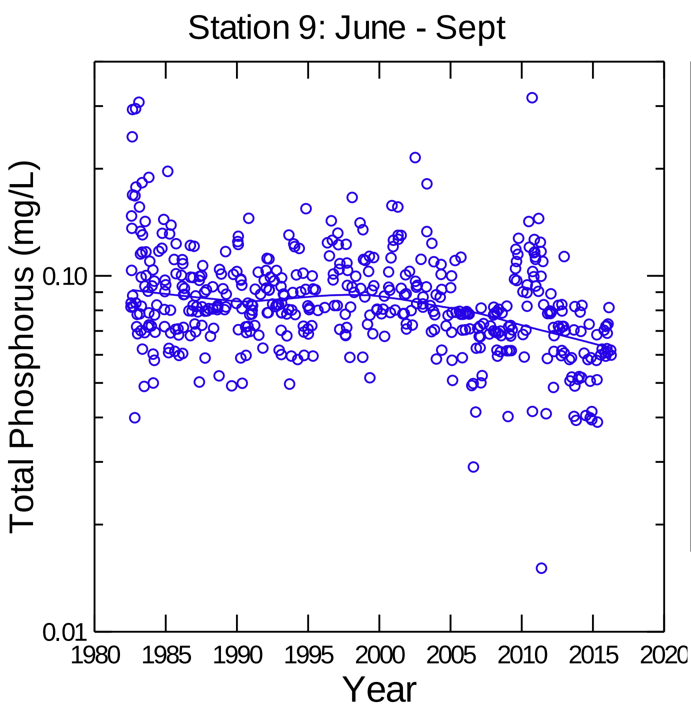
In the river the trend line for volatile suspended solids (VSS) was steady from 1984 through the mid 1990's, but decreased from 1995 to 2005. Trend line values of about 7 mg/L in 1984 dropped to about 4 mg/L by 2016 (Figure 96). VSS in the river demonstrated a significant linear decline at a rate of 0.13 mg/L per year or 4.2 mg/L since 1984 (Table 15).

Figure 96. Long term trend in Volatile Suspended Solids (Fairfax County Lab Data). Station 9. Potomac mainstem.



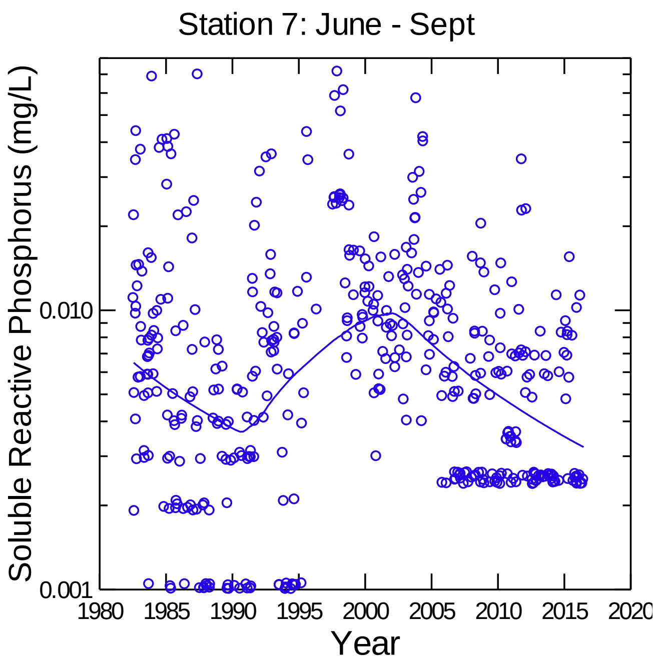
In the cove, total phosphorus (TP) has undergone a consistent steady decline since the late 1980's (Figure 97). By 2016 the trend line had dropped to 0.06 mg/L, more than half of the starting level. Linear regression over the entire period of record indicated a significant linear decline of -0.004 mg/L per year or 0.13 mg/L over the entire study period (Table 15).

Figure 97. Long term trend in Total Phosphorus (Fairfax County Lab Data). Station 7. Gunston Cove.



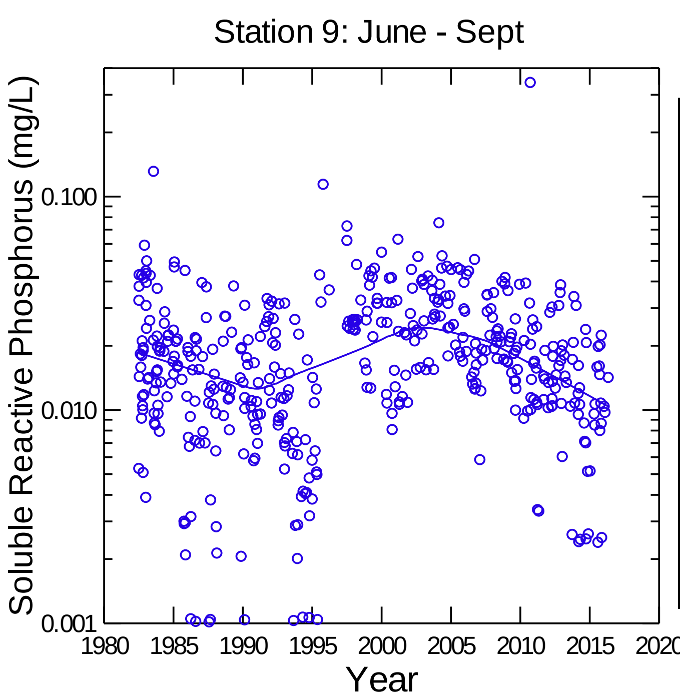
Total phosphorus (TP) values in the river have shown less of a trend over time (Figure 98). Values were steady through about 2000, then declined somewhat. TP exhibited a slight, but significant linear decrease in the river over the long term study period with a very modest slope of -0.0009 mg/L per year (Table 15).

Figure 98. Long term trend in Total Phosphorus (Fairfax County Lab Data). Station 9. Potomac mainstem.



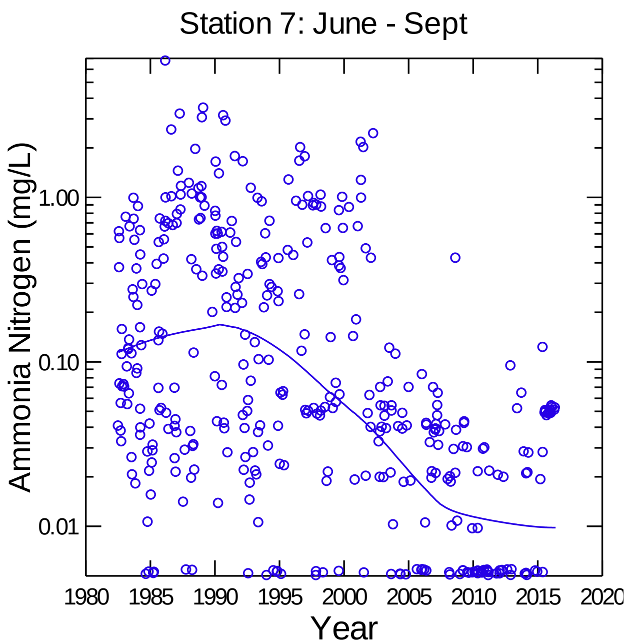
Soluble reactive phosphorus (SRP) declined in the cove during the first few years of the long term data set, but demonstrated an increase to near its initial level by 2000 (Figure 99). Since then a decline has ensued. (Table 15). One possibility is that less SRP is entering the cove water; another is that increased SAV is taking more up. Note also that the detection limit has changed and that many readings are at the detection limit making trend analysis difficult and uncertain.

Figure 99. Long term trend in Soluble Reactive Phosphorus (Fairfax County Lab Data). Station 7. Gunston Cove.



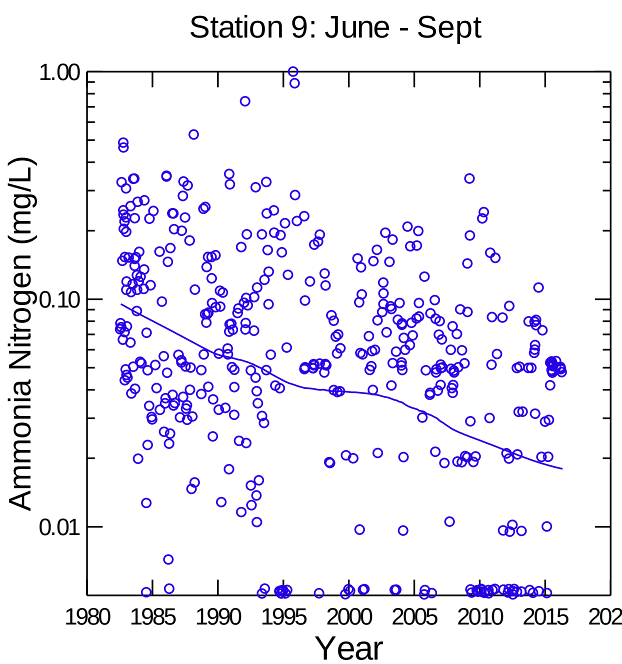
Soluble reactive phosphorus (SRP) in the river has generally been present at higher levels than in the cove, but has undergone a similar decline-resurgence-decline (Figure 100). Linear regression was not significant (Table 15). There were a significant number of non-detect values, but fewer than in the cove.

Figure 100. Long term trend in Soluble Reactive Phosphorus (Fairfax County Lab Data). Station 9. Potomac mainstem.



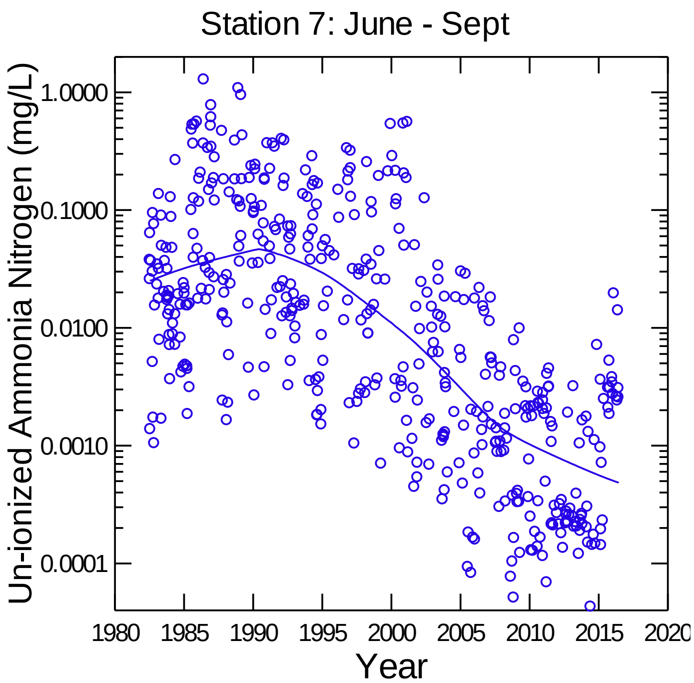
Ammonia nitrogen levels were very variable over the long term study period in the cove, but a trend of decreasing values is evident from the LOWESS trend line (Figure 101). Since 1989 the trend line has decreased from about 0.2 mg/L to less 0.01 mg/L. Linear regression has revealed a significant decline over the entire period of record with a rate of 0.017 mg/L per year yielding a total decline of 0.58 mg/L (Table 15). Note the increase in values below the detection limit over time (clustered at bottom of graph). This is making the detection of trends increasingly uncertain.

Figure 101. Long term trend in Ammonia Nitrogen (Fairfax County Lab Data). Station 7. Gunston Cove.



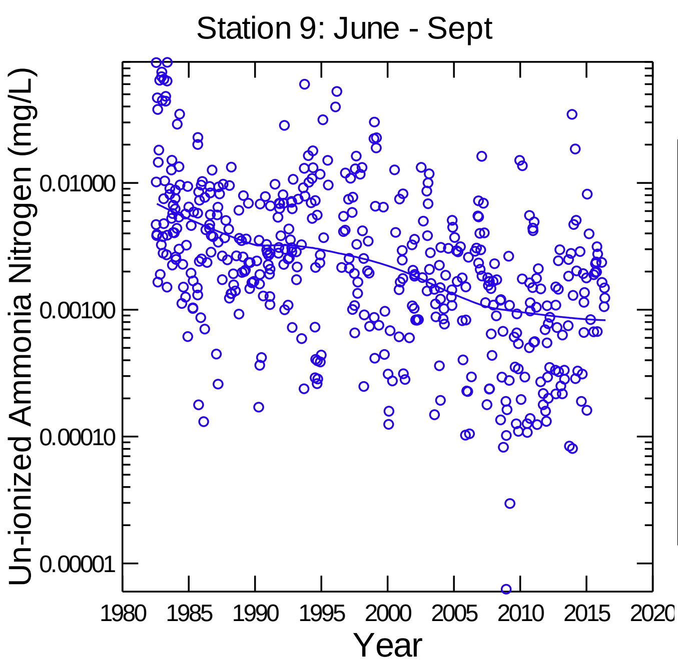
In the river a decreasing trend in ammonia nitrogen has also been observed over most of the study period (Figure 102). Between 1983 and 1999 the trend line dropped from 0.1 mg/L to 0.04 mg/L. Since 1999 it has continued to decline and is now at about 0.02 mg/L. Overall, in the river ammonia nitrogen has demonstrated a significant decline over the study period at a rate of 0.003 mg/L per year or a total of 0.09 over the study period (Table 15). Again, the number of non-detects is increasing.

Figure 102. Long term trend in Ammonia Nitrogen (Fairfax County Lab Data). Station 9. Potomac mainstem.



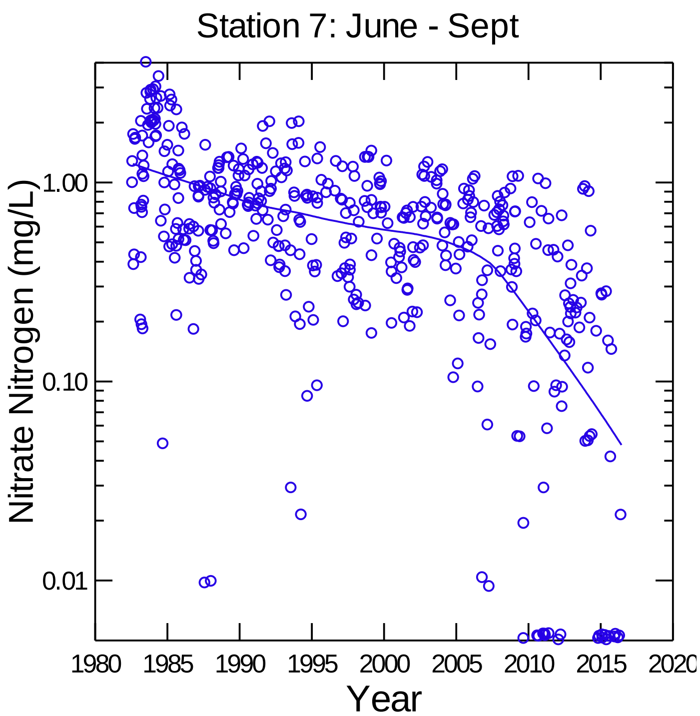
Un-ionized ammonia nitrogen in the cove demonstrated a clear increase in the 1980's with a continuous decline since that time (Figure 103). The LOWESS trend peaked at about 0.05 mg/L and is now about 0.0005 mg/L. When considered over the entire time period, there was a significant decline at a rate of 0.004 mg/L per year or a total of 0.15 mg/L over the 33 years (Table 15). Note that these values are dependent on ammonia nitrogen which has been showing increasing incidence of non-detects.

Figure 103. Long term trend in Un-ionized Ammonia Nitrogen (Fairfax County Lab Data). Station 7. Gunston Cove.



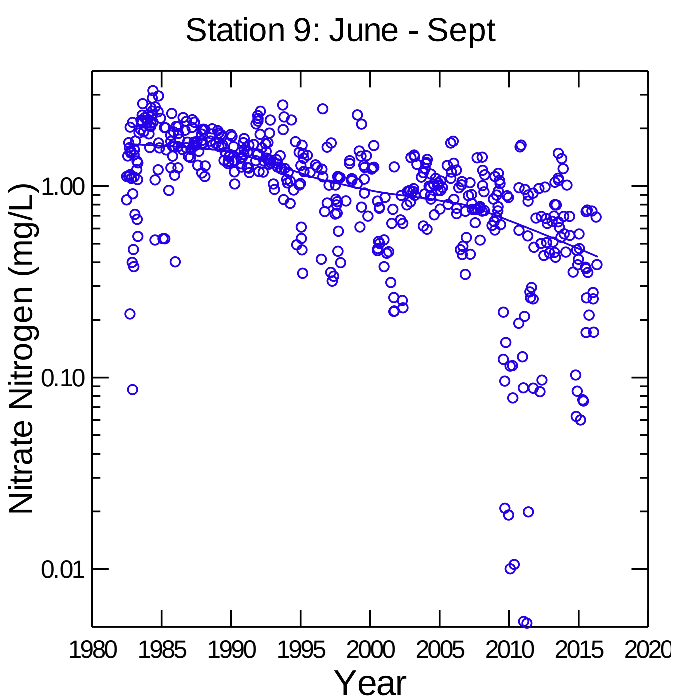
Un-ionized ammonia nitrogen in the river declined fairly consistently over the entire study period (Figure 104). LOWESS values have dropped from about 0.007 mg/L to about 0.0009 mg/L. Linear regression analysis over the entire period of record suggested a significant decline at a rate of 0.0003 units per year (Table 15).

Figure 104. Long term trend in Un-ionized Ammonia Nitrogen (Fairfax County Lab Data). Station 9. Potomac mainstem.



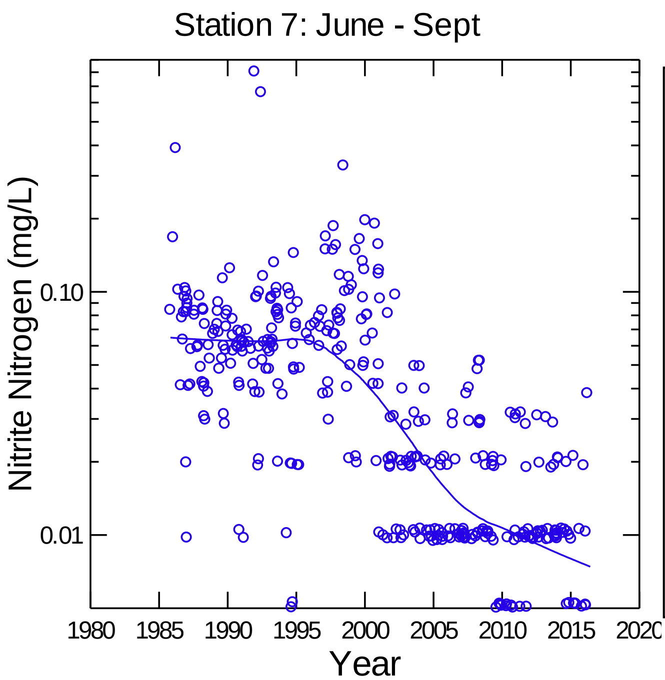
Nitrate nitrogen has demonstrated a steady decline in the cove over the entire period of record (Figure 105). The trend line was at about 1 mg/L in 1983 and by 2016 was below 0.05 mg/L. Linear regression suggested a decline rate of 0.035 mg/L per year yielding a total decline of 1.1 mg/L over the long term study period (Table 15). Note the large number of non-detect values in recent years.

Figure 105. Long term trend in Nitrate Nitrogen (Fairfax County Lab Data). Station 7. Gunston Cove.



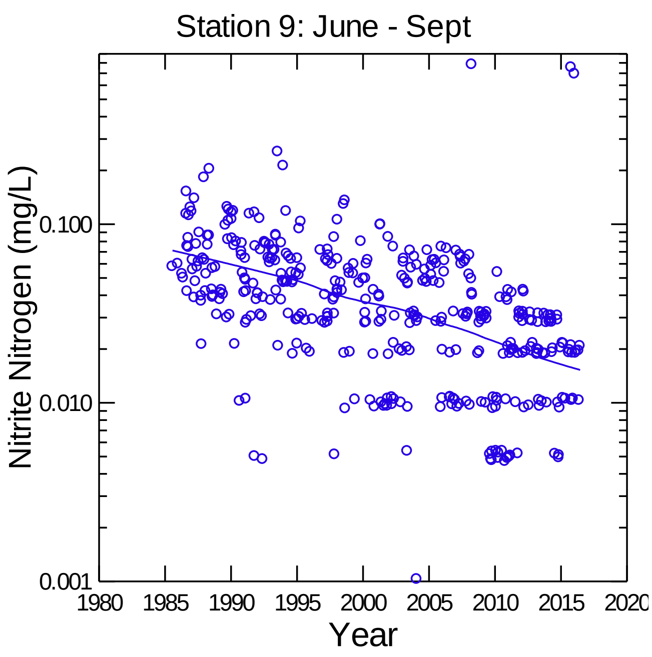
In the river nitrate nitrogen has declined steadily since about 1985 (Figure 106). The trend line dropped from 1.5 mg/L in the mid 1980's to 0.4 mg/L in 2016. Linear regression indicated a rate of decline of -0.04 mg/L per yr which would have yielded a 1.3 mg/L decrease in nitrate nitrogen over the study period (Table 15).

Figure 106. Long term trend in Nitrate Nitrogen (Fairfax County Lab Data). Station 9. River mainstem.



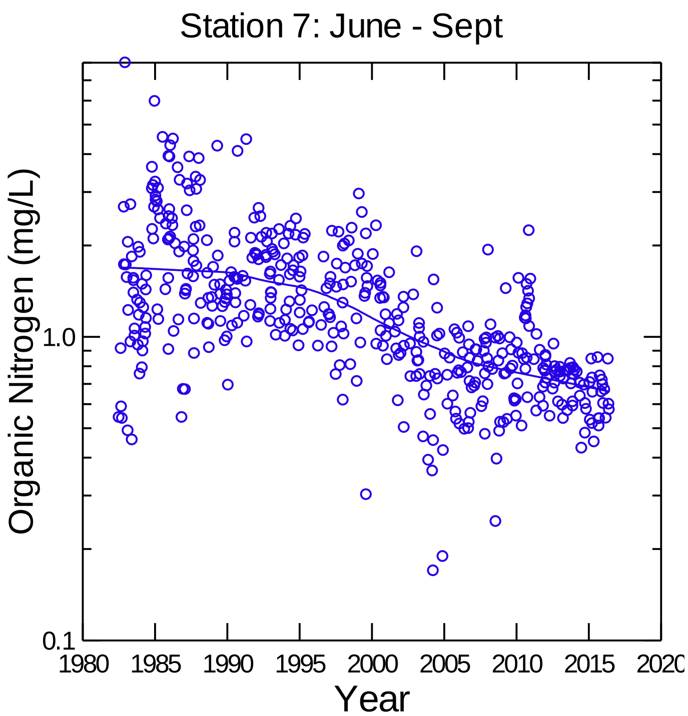
The trend line for nitrite nitrogen indicated steady values at about 0.06-0.07 mg/L through 1999 (Figure 107). Since then there is clear evidence for a decline with the LOWESS line dropping below 0.01 in 2016. Linear regression revealed a significant decline with a slope of 0.003 mg/L per year when the entire period of record was considered (Table 15). Most values in recent years have been at or below the detection limits.

Figure 107. Long term trend in Nitrite Nitrogen (Fairfax County Lab Data). Station 7. Gunston Cove.



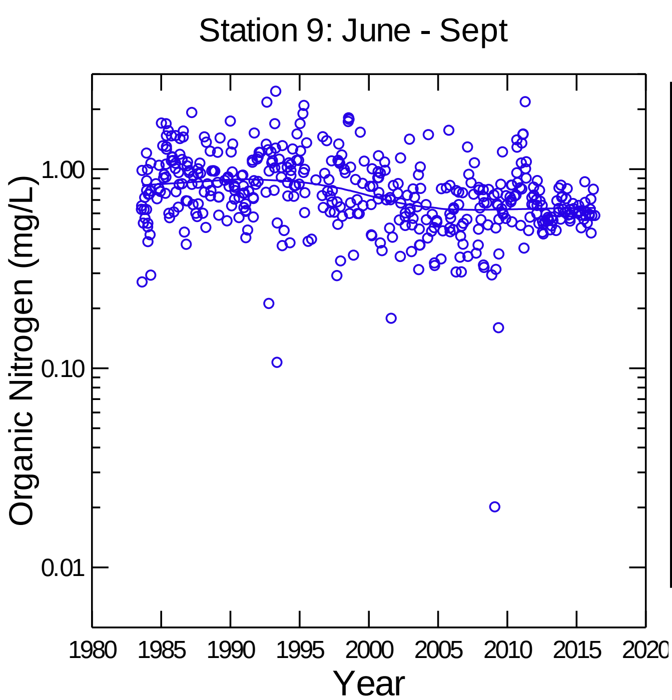
Nitrite nitrogen in the river demonstrated a pattern of decrease during the long term study period (Figure 108). The LOWESS line dropped from 0.07 mg/L in 1986 to less than 0.02 mg/L in 2016. Linear regression indicated a significant linear decline at a rate of 0.001 mg/L per year or 0.03 mg/L over the study period (Table 15).

Figure 108. Long term trend in Nitrite Nitrogen (Fairfax County Lab Data). Station 9. Potomac mainstem.



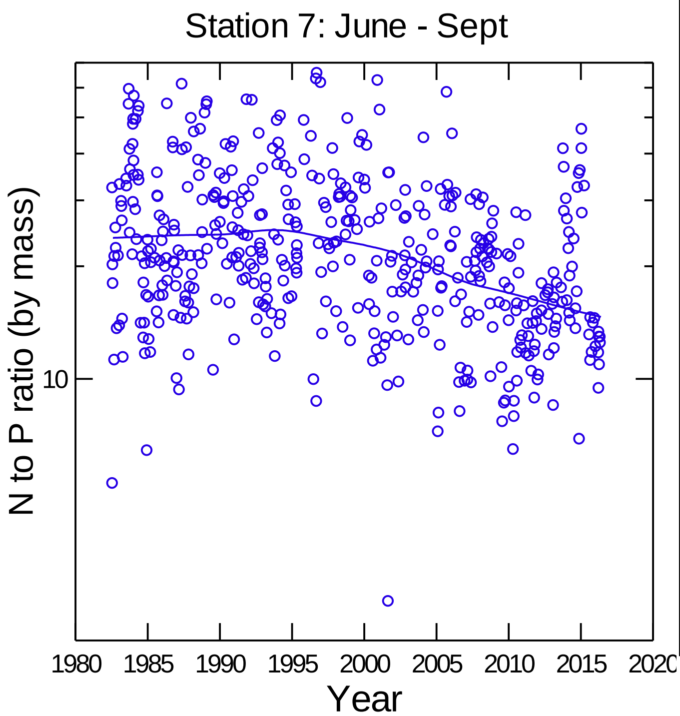
Organic nitrogen in the cove was fairly high in the 1980's and has since undergone a consistent decline through 2016 (Figure 109). In 1983 the trend line was at 1.5 mg/L and dropped below 0.7 mg/L by 2016. Regression analysis indicated a significant decline over the study period at a rate of about 0.047 mg/L per year or a total of 1.5 mg/L over the whole study period (Table 15).

Figure 109. Long term trend in Organic Nitrogen (Fairfax County Lab Data). Station 7. Gunston Cove.



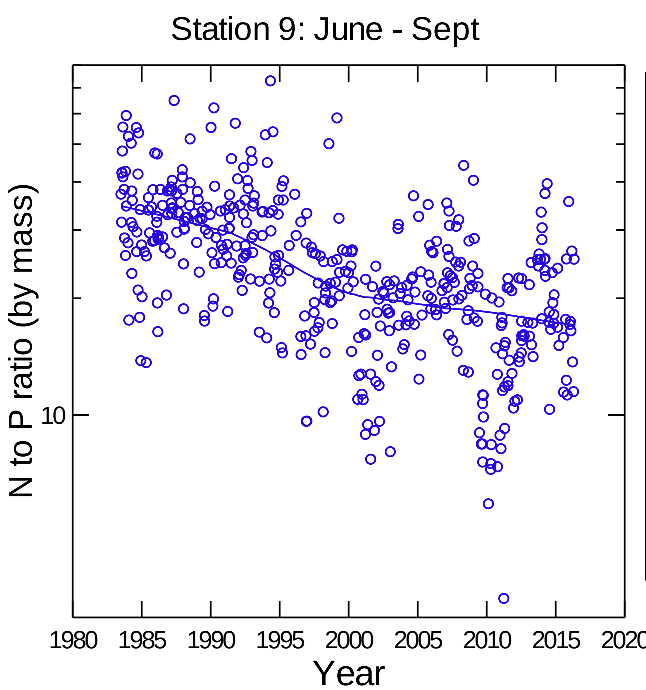
In the river organic nitrogen was steady from 1984 through 1995 and since then has shown perhaps a modest decline (Figure 110). The LOWESS line peaked at about 0.9 mg/L and has dropped to about 0.7 mg/L. Regression analysis indicated a significant linear decline at a rate of 0.01 mg/L when the entire period of record was considered for a total decline of 0.3 mg/L (Table 15).

Figure 110. Long term trend in Organic Nitrogen (Fairfax County Lab Data). Station 9. River mainstem.



Nitrogen to phosphorus ratio (N/P ratio) in the cove exhibited large variability, but the trend line was flat until about 1995. Since then, there has been a clear decline with the LOWESS line approaching 15 by 2016 (Figure 111). Regression analysis over the period of record indicates a statistically significant decline at a rate of 0.35 per year or about 11 units over the entire period (Table 15). Values in 2015 were well above the trend line, but were below the trend line in 2016. This ratio is calculated using nitrate, TKN, and TP values and are less accurate when any of those are below detection limits.

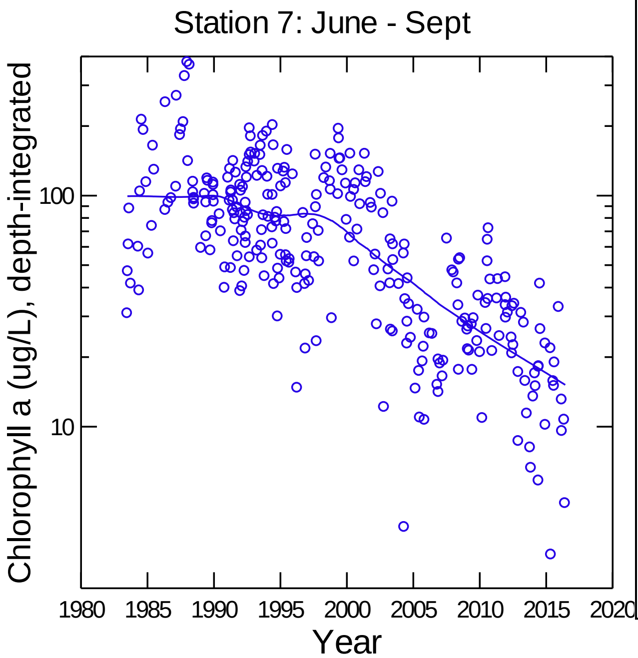
Figure 111. Long term trend in N to P Ratio (Fairfax County Lab Data). Station 7. Gunston Cove.



Nitrogen to phosphorus ratio in the river exhibited a strong continuous decline through about 2000 and has declined more slowly since then (Figure 112). The LOWESS trend line declined from about 35 in 1984 to 17 in 2016. Linear regression analysis confirmed this decline and suggested a rate of 0.6 units per year or a total of 20 units over the long term study period (Table 15).

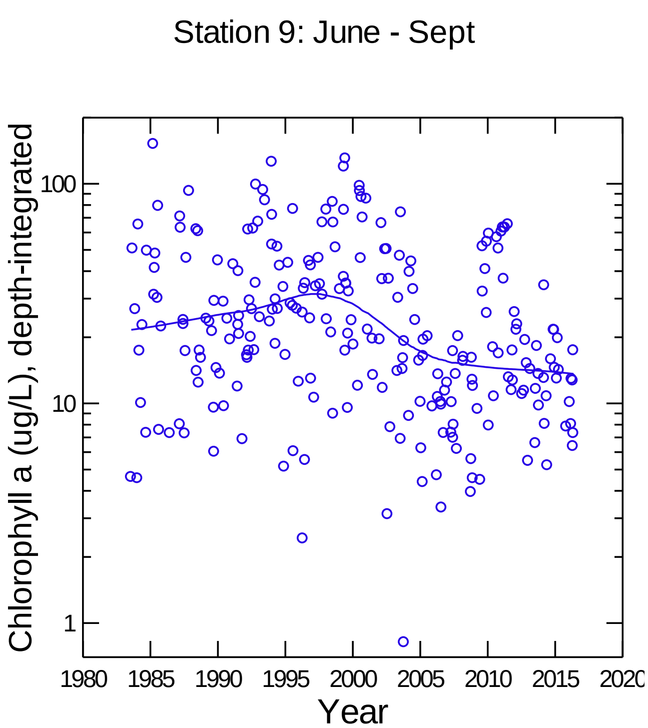
Figure 112. Long term trend in N to P Ratio (Fairfax County Lab Data). Station 9. River mainstem.

C. Phytoplankton Trends: 1984-2016



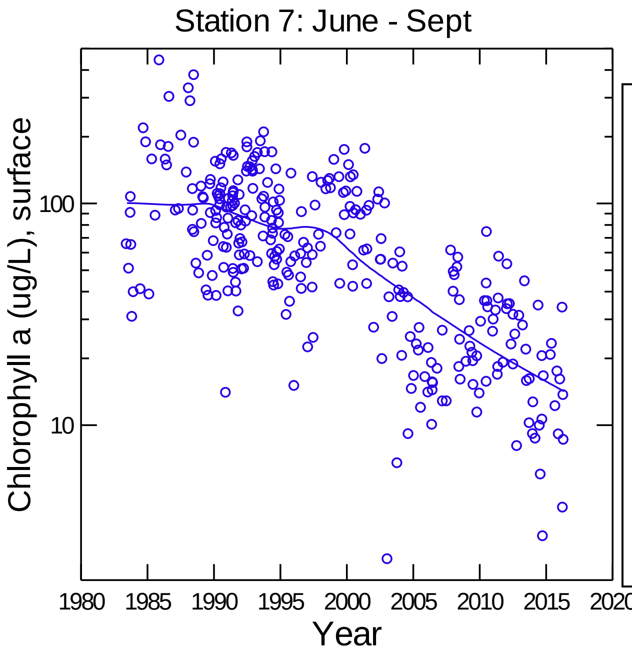
After increasing through much of the 1980’s, depth-integrated chlorophyll *a* in the cove demonstrated a gradual decline from 1988 to 2000 and a much stronger decrease since then (Figure 113). The LOWESS line has declined from about 100 µg/L to less than 15 µg/L in 2016. The observed decrease has resulted in chlorophyll values within the range of water clarity criteria allowing SAV growth to 0.5 m and 1.0 m (43 µg/L and 11 µg/L, respectively) (CBP 2006). This would imply adequate light to support SAV growth over much of Gunston Cove. Regression analysis has revealed a clear linear trend of decreasing values at the rate of 3.9 µg/L per year or 125 µg/L over the 32-year long term data set (Table 14).

Figure 113. Long term trend in Depth-integrated Chlorophyll *a* (GMU Lab Data). Station 7. Gunston Cove.



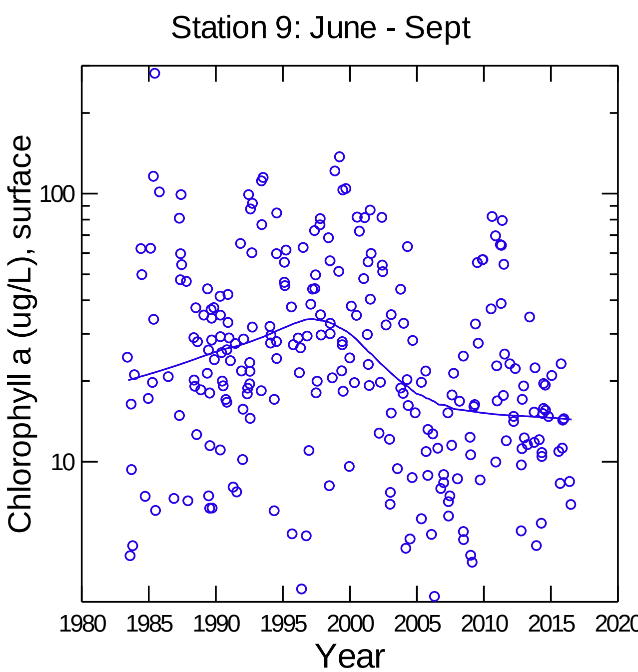
In the river depth-integrated chlorophyll *a* increased gradually through 2000 with the trend line rising from 20 to 30 µg/L (Figure 114). This was followed by a strong decline through about 2005 reaching about 18 µg/L with a further gradual decline to date. Regression analysis revealed a significant linear decline at a rate of 0.7 µg/L/yr when the entire period is considered (Table 14) yielding a total decline of about 22 ug/L.

Figure 114. Long term trend in Depth-integrated Chlorophyll *a* (GMU Lab Data). Station 9. River mainstem.



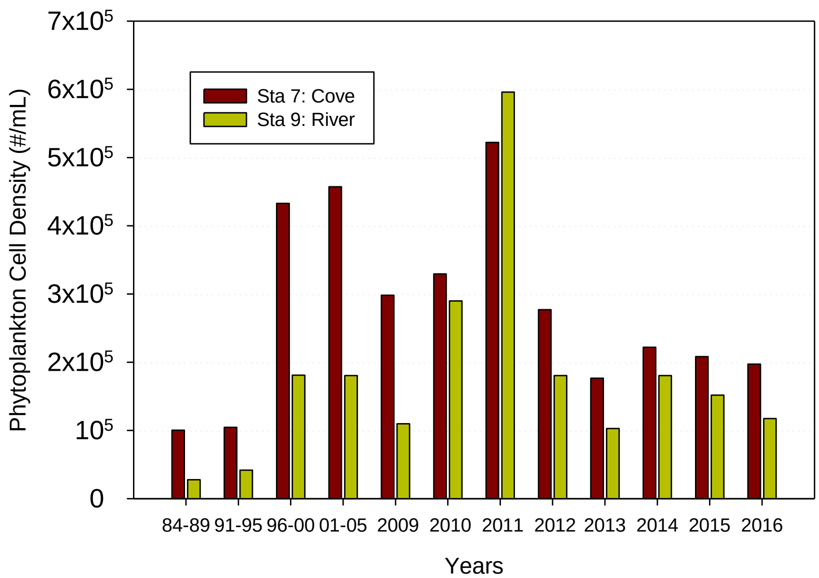
Surface chlorophyll *a* in the cove also exhibited a clear decline over the long term study period, especially since 2000 (Figure 115). Trend line values of about 100 µg/L in 1988 dropped to less than 15 µg/L in 2016. Linear regression confirmed the linear decline and suggested a rate of 4.0 µg/L per year or 128 µg/L over the entire study (Table 14).

Figure 115. Long term trend in Surface Chlorophyll *a* (GMU Data). Station 7. Gunston Cove.



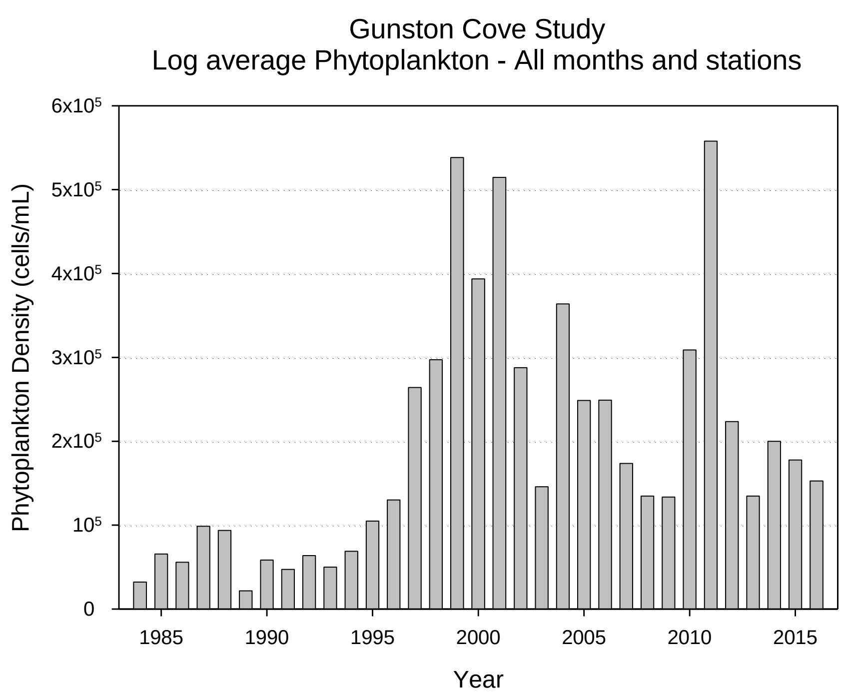
In the river the LOWESS line for surface chlorophyll *a* increased slowly from 1983 to 2000 and then declined markedly through 2005 (Figure 116). Values have stabilized since then at about 15 µg/L. Linear regression revealed a significant decline in surface chlorophyll across this period with a rate of 0.78 µg/L/yr or about 25 µg/L over the whole period (Table 14).

Figure 116. Long term trend in Surface Chlorophyll *a* (GMU Data). Station 9. River mainstem.



Phytoplankton cell density in both the cove and the river in 2016 was similar to values observed since 2012 (Figure 117). While cell density does not incorporate cell size, it does provide some measure of the abundance of phytoplankton and reflects the continuing decrease in phytoplankton in the study area which is expected with lower nutrient loading and should help improve water clarity.

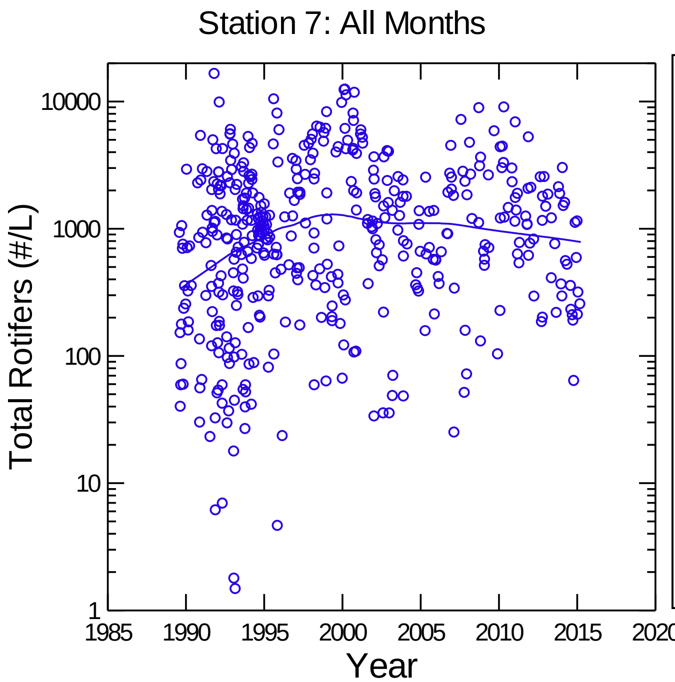
Figure 117. Interannual Comparison of Phytoplankton Density by Region.



By looking at individual years (Figure 118), we see that phytoplankton densities in 2016 remained lower than the high levels observed during the 1995 to 2005 period.

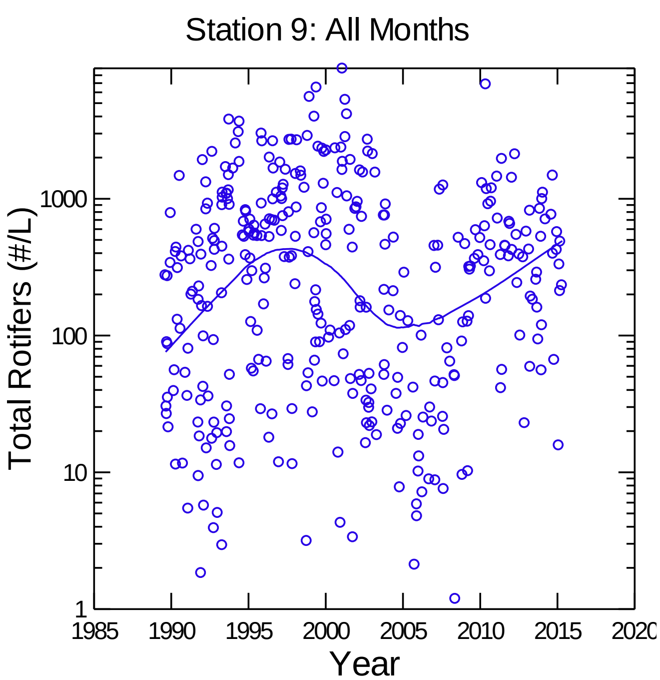
Figure 118. Interannual Trend in Average Phytoplankton Density.

D. Zooplankton Trends: 1990-2016



In the Cove total rotifers continued to exhibit a slow decline after an initial decade (1990-2000) of steady increase (Figure 119). The LOWESS fit line indicated about 800/L in 2013, up from about 400/L in 1990. Linear regression analysis continued to indicate a statistically significant linear increase in total rotifers over the period since 1990 (Table 16).

Figure 119. Long term trend in Total Rotifers. Station 7. Gunston Cove.



In the Potomac mainstem, rotifers exhibited an initial increase from 1990 to 1998, followed by a decline from 1999 to 2005 and more recently another increase (Figure 120). Trend line values in 1990 were about 80/L and now are about 500/L approaching 1998 values. However, when the entire 1990-2016 period was considered, total rotifers did not exhibit a significant linear trend in the river (Table 16).

Figure 120. Long term trend in Total Rotifers. Station 9. River mainstem.

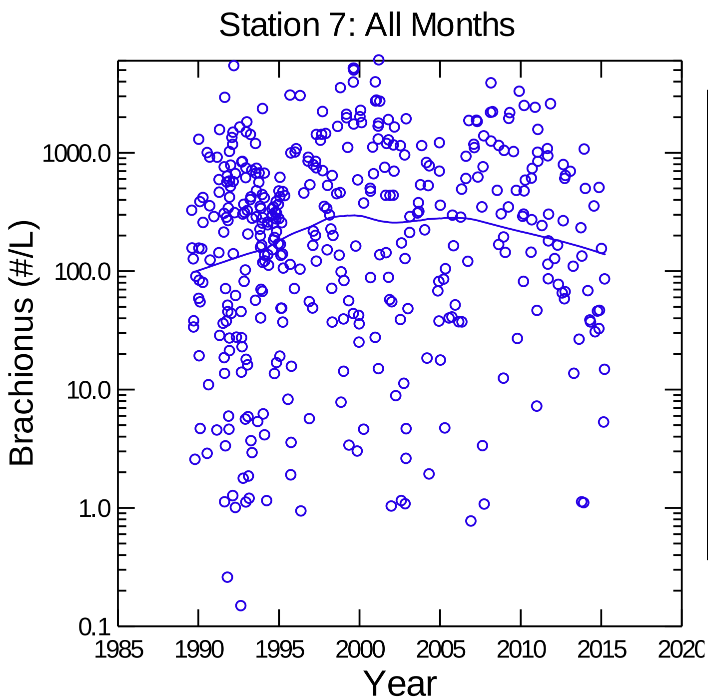
Table 16
Correlation and Linear Regression Coefficients
Zooplankton Parameters vs. Year for 1990-2016
All Nonzero Values Used, All Values Logged to Base 10

	Station 7			Station 9		
Parameter	Corr. Coeff.	Reg. Coeff.	Signif.	Corr. Coeff.	Reg. Coeff.	Signif.
Brachionus (m)	0.087 (426)	-----	NS	0.070 (349)	-----	NS
Conochilidae (m)	0.155 (377)	0.014	0.002	0.032 (298)	-----	NS
Filinia (m)	0.125 (369)	0.014	0.016	0.125 (252)	-0.011	0.048
Keratella (m)	0.309 (437)	0.030	<0.001	0.145 (362)	0.016	0.006
Polyarthra (m)	0.176 (416)	0.017	<0.001	0.081 (341)	-----	NS
Total Rotifers (m)	0.138 (454)	0.012	0.003	0.044 (374)	-----	NS
Bosmina (m)	0.014 (258)	-----	NS	0.049 (306)	-----	NS
Diaphanosoma (M)	0.141 (356)	-0.023	0.008	0.130 (261)	-0.017	0.035
Daphnia (M)	0.018 (281)	-----	NS	0.001 (186)	-----	NS
Chydorid cladocera (M)	0.135 (249)	0.014	0.033	0.109 (168)	-----	NS
Leptodora (M)	0.216 (203)	-0.026	0.002	0.273 (145)	0.273	0.001
Copepod nauplii (m)	0.433 (433)	0.033	<0.001	0.232 (370)	0.232	<0.001
Adult and copepodid copepods (M)	0.066 (549)	-----	NS	0.030 (414)	-----	NS

n values (# of data points) are shown in Corr. Coeff. column in parentheses.

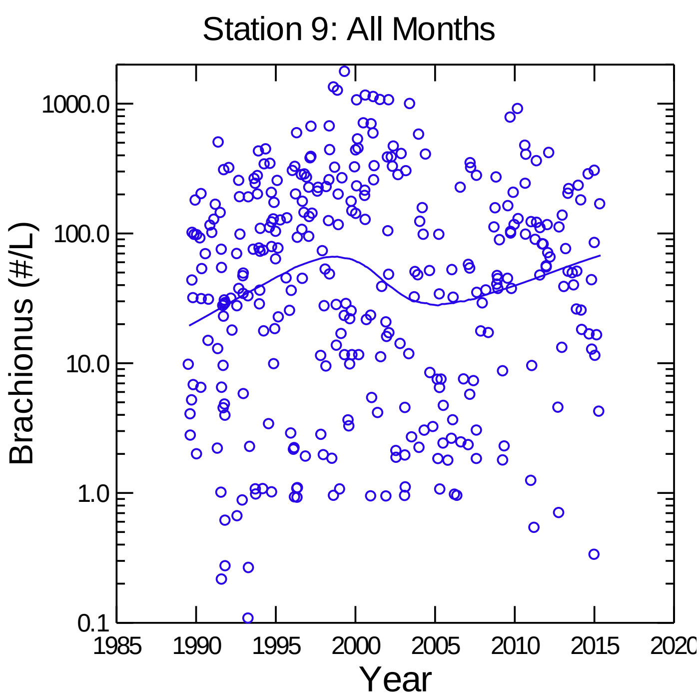
Significance column indicates the probability that a correlation coefficient this large could be due to chance alone. If this probability is greater than 0.05, then NS (not significant) is indicated. * = marginally significant.

M indicates species was quantified from macrozooplankton samples; m indicates quantification from microzooplankton samples.



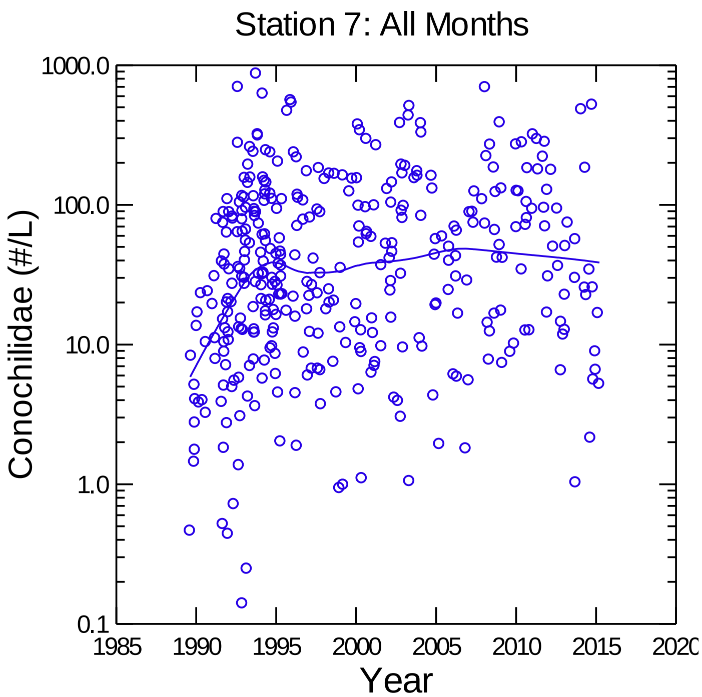
Brachionus is the dominant rotifer in Gunston Cove and the trends in total rotifers are generally mirrored in those in *Brachionus* (Figure 121). The LOWESS line for *Brachionus* suggested about 150/L in 2016, only slightly greater than the 100/L found in 1990. No linear trend was found over the study period (Table 16).

Figure 121. Long term trend in *Brachionus*. Station 7. Gunston Cove.



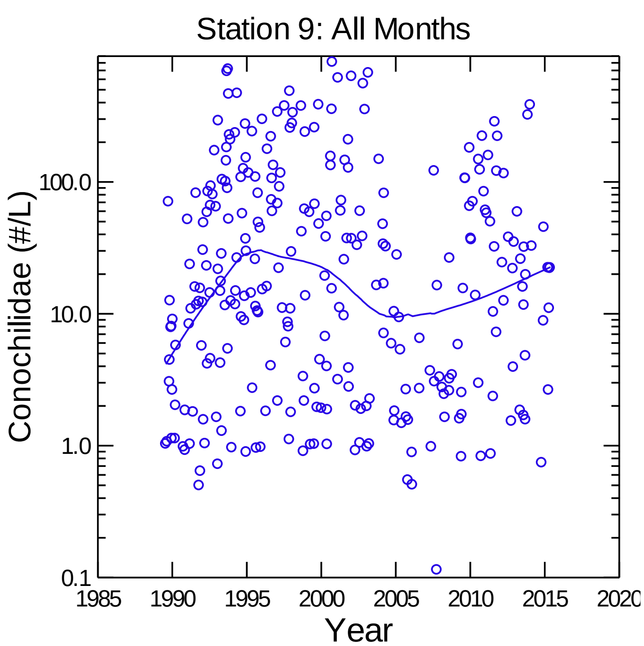
Brachionus was found at lower densities in the river. In the river the LOWESS line for *Brachionus* increased through 2000, but dropped markedly from 2000-2005.Since 2005 an increase has been noted, with the LOWESS value in 2016 of about 70/L, higher than the initial 20/L and near the previous peak of 80/L in 1999 (Figure 122). No linear trend was indicated when the entire study period was considered (Table 16).

Figure 122. Long term trend in *Brachionus*. Station 9. River mainstem.



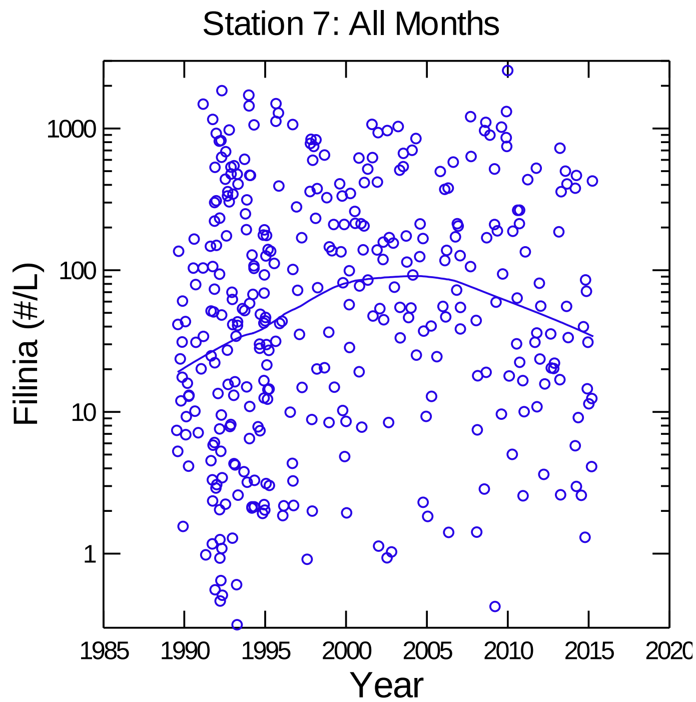
Conochilidae increased strongly from 1990-1995 and since then has leveled off. In 2016 the LOWESS trend line stood at about 40/L (Figure 123). This was well above levels of about 5/L in 1990. Over the entire period of record, a significant linear increase was found (Table 16).

Figure 123. Long term trend in Conochilidae. Station 7. Gunston Cove.



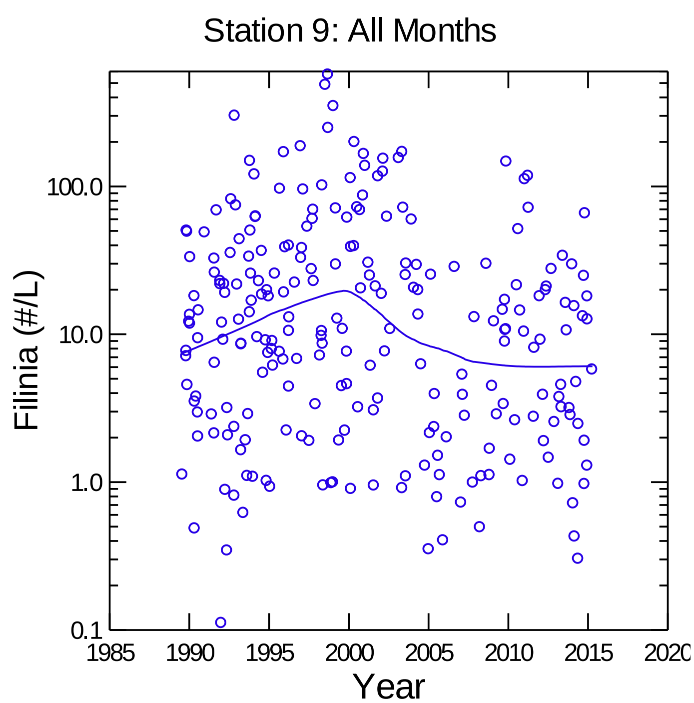
In the river, Conochilidae exhibited a strong increase in the early 1990's similar to that observed in the cove (Figure 124). This was followed by a period of decline and recently a renewed increase. The trend line has gone from 3/L in 1990 to 35/L in 1995 to 10/L in 2005 to 25/L in 2016. When the entire period of record was examined, there was not a significant linear trend (Table 16).

Figure 124. Long term trend in Conochilidae. Station 9. River mainstem.



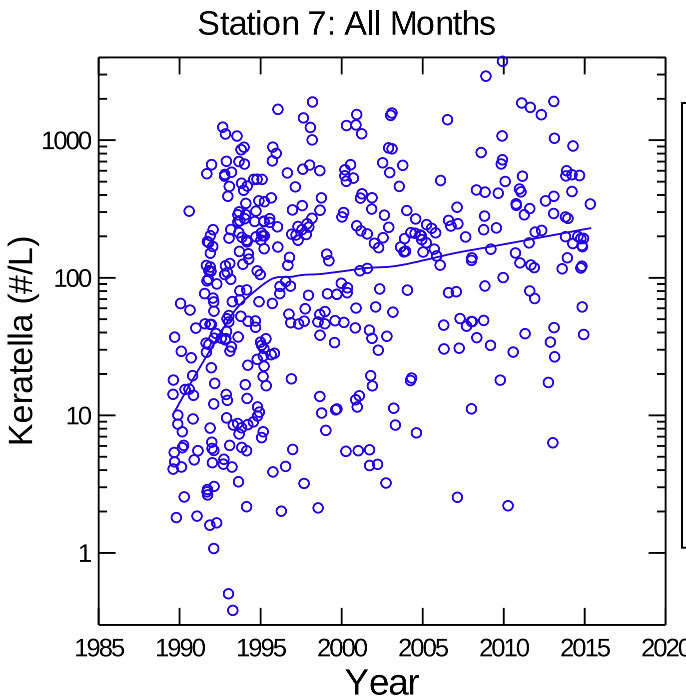
In the cove *Filinia* exhibited a steady increase from 1990 through 2000 rising from about 20/L to nearly 100/L (Figure 125). It has shown a gradual decline in recent years to about 30/L. When the entire period of record was considered, there is evidence for a linear increase in the cove despite the recent declines (Table 16).

Figure 125. Long term trend in *Filinia*. Station 7. Gunston Cove.



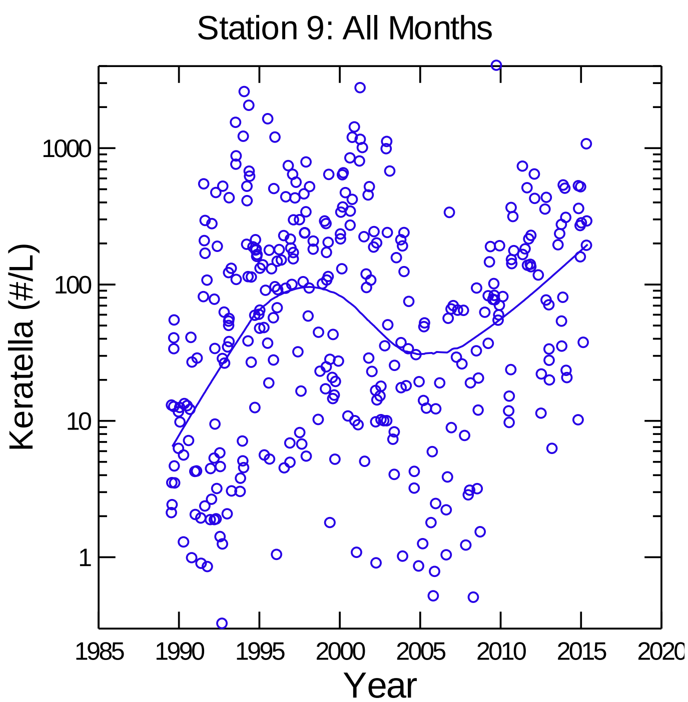
In the river *Filinia* demonstrated an increase through about 2001, declined from 2000-2005 and remained steady since. The trend line indicates about 6/L in 2016, about equal to the 7/L in 1990, but well below the peak of 20/L in 2000 (Figure 126). When the entire period of record was examined, there was a barely significant negative linear trend (Table 16).

Figure 126. Long term trend in *Filinia*. Station 9. River mainstem.



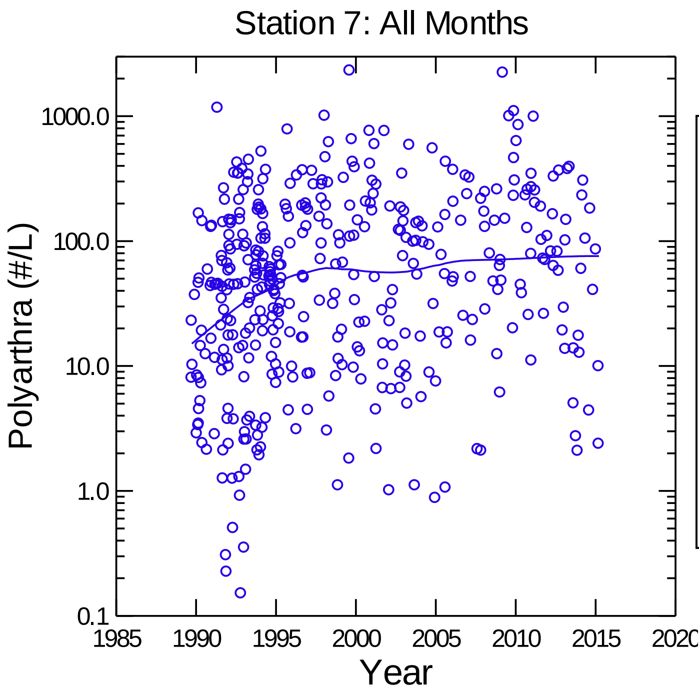
Keratella increased strongly from 1990 to 1995 and has shown a milder increase since then with the trend line approaching 250/L in 2016 (Figure 127). When the entire period of record was examined, there was a significant linear increase (Table 16).

Figure 127. Long term trend in *Keratella*. Station 7. Gunston Cove.



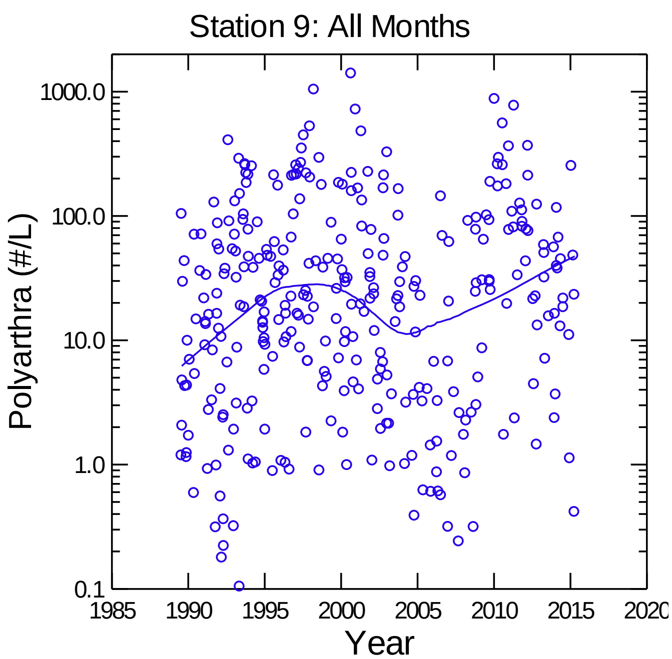
In the river *Keratella* increased from less than 10/L in 1990 to peak values of about 100/L in the mid to late 1990's (Figure 128). The trend line then declined to about 25/L, but since 2005 it has increased to about 200/L. Linear regression showed evidence of a linear increase when the entire study period was considered (Table 16).

Figure 128. Long term trend in *Keratella*. Station 9. River mainstem.



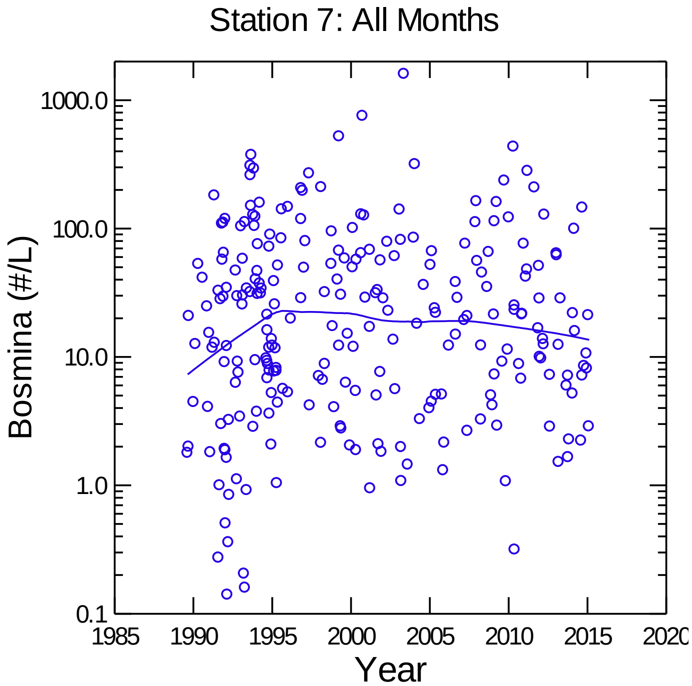
The trend line for *Polyarthra* in the cove increased steadily from 1990 to about 2000 rising from 15/L to about 60/L (Figure 129). Since 2000 densities have increased more slowly with the trend line reaching about 80/L by 2016. Regression analysis indicated a significant linear increase when the entire period of record was examined (Table 16).

Figure 129. Long term trend in *Polyarthra*. Station 7. Gunston Cove.



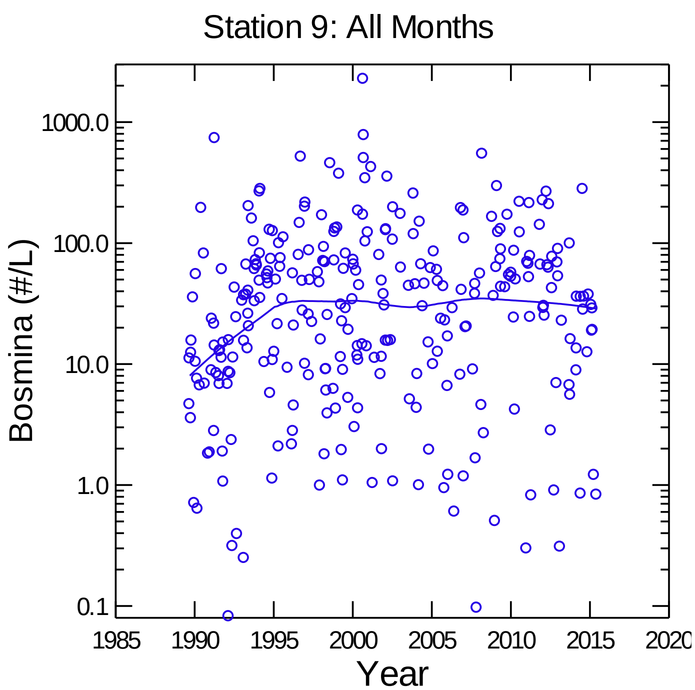
In the river *Polyarthra* showed a marked increase from 1990 to 2000 and then a decline to 2005. Recently values have increased again and by 2016 the trend line reached 50/L (Figure 130). Linear regression analysis did not indicate a significant positive trend over the period of record (Table 16).

Figure 130. Long term trend in *Polyarthra*. Station 9. River mainstem.



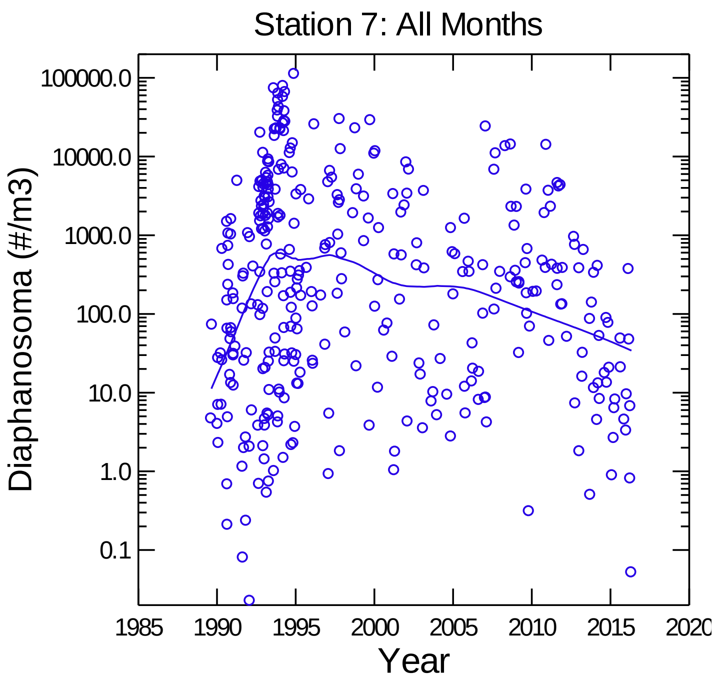
The trend line for *Bosmina* in the cove showed an increase from 8/L in 1990 to about 20/L in 2000 (Figure 131). Since 2000 densities have not changed much. Linear regression did not indicate a significant trend in the cove over the entire period of record (Table 16).

Figure 131. Long term trend in *Bosmina*. Station 7. Gunston Cove.



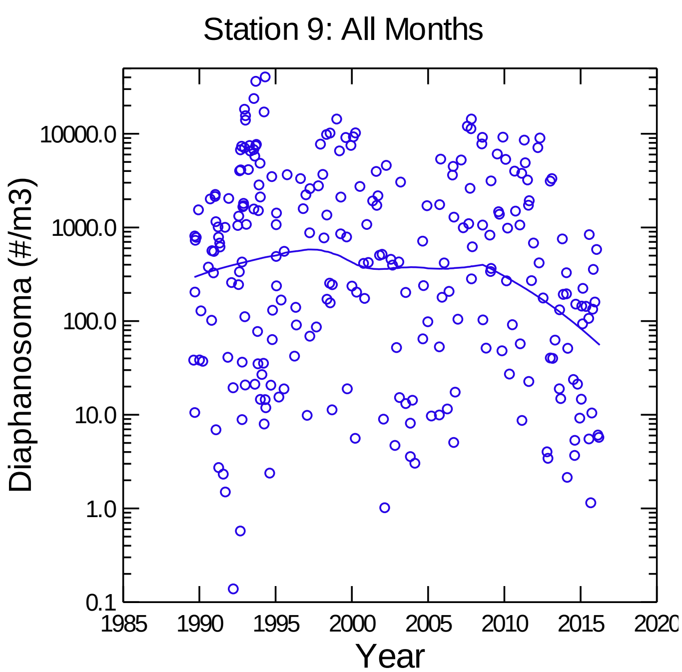
In the river mainstem the LOWESS curve for *Bosmina* increased from 1990 to 1995, and remained rather constant from 1995 to 2016 at about 30/L (Figure 132). Regression analysis did not indicate a significant linear increase over the entire period of record (Table 16).

Figure 132. Long term trend in *Bosmina*. Station 9. River mainstem.



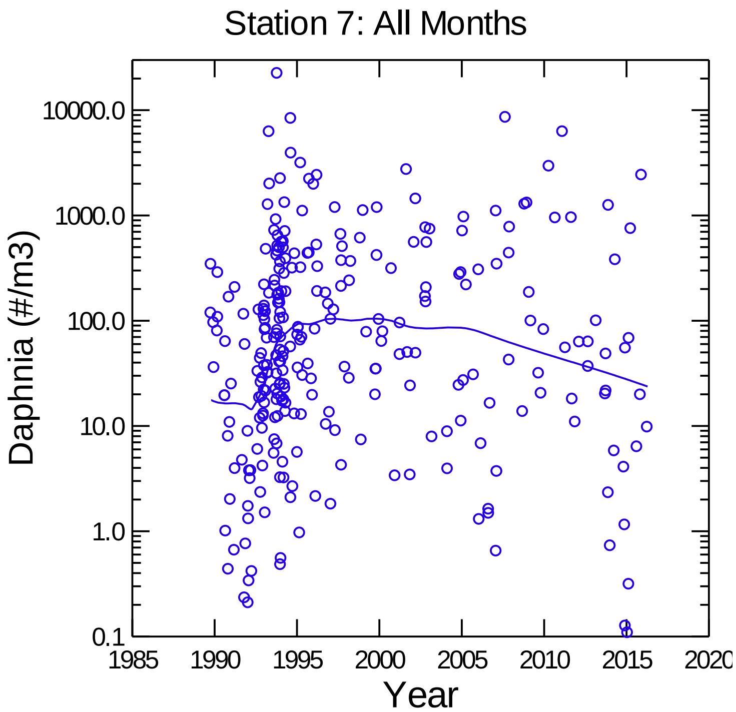
Diaphanosoma increased strongly in the early 1990s from about 12/m^3 nearly 1000/m^3. It gradually declined through 2016 to about 40/m^3 (Figure 133). Many 2016 values were below 100/m^3. Linear regression analysis of the entire period of record indicated a significant decline (Table 16).

Figure 133. Long term trend in *Diaphanosoma*. Station 7. Gunston Cove.



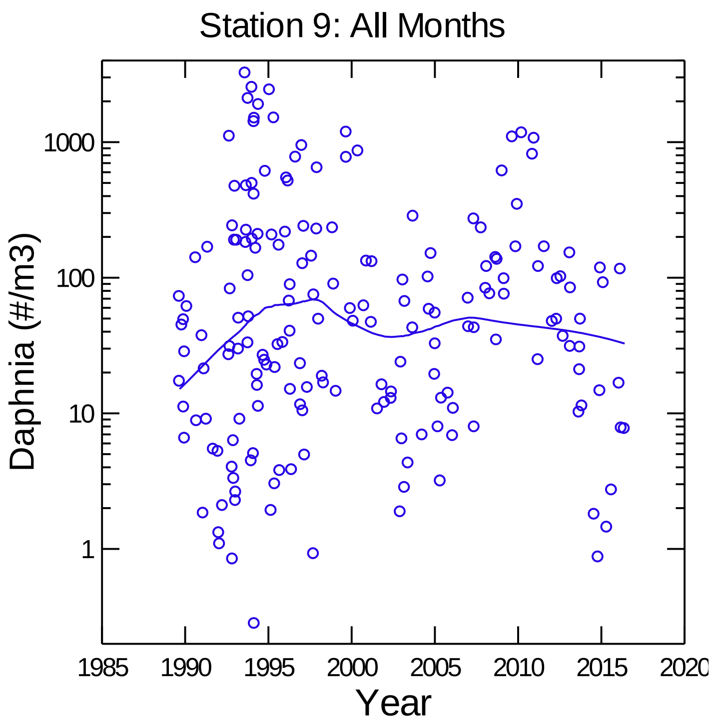
In the river the LOWESS line suggested a generally stable pattern in *Diaphanosoma* until 2010 until a decline set in (Figure 134). The trend line value of 50/m^3 found in 2016 compared with values as high as 600/m^3 in 1999. Regression analysis indicated significant declining trend over the period of record (Table 16).

Figure 134. Long term trend in *Diaphanosoma*. Station 9. River mainstem.



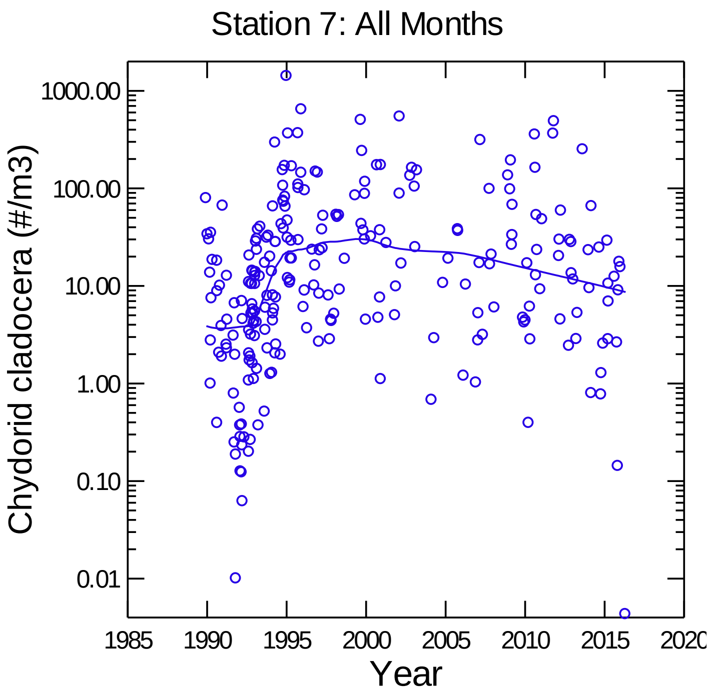
Daphnia in the cove has declined slowly since 1995 from about 100/m^3 to 30/m^3 (Figure 135). This is up slightly from the low of about 20/m^3 in the early 1990's. Regression analysis examining the entire period of record was not significant (Table 16).

Figure 135. Long term trend in *Daphnia*. Station 7. Gunston Cove.



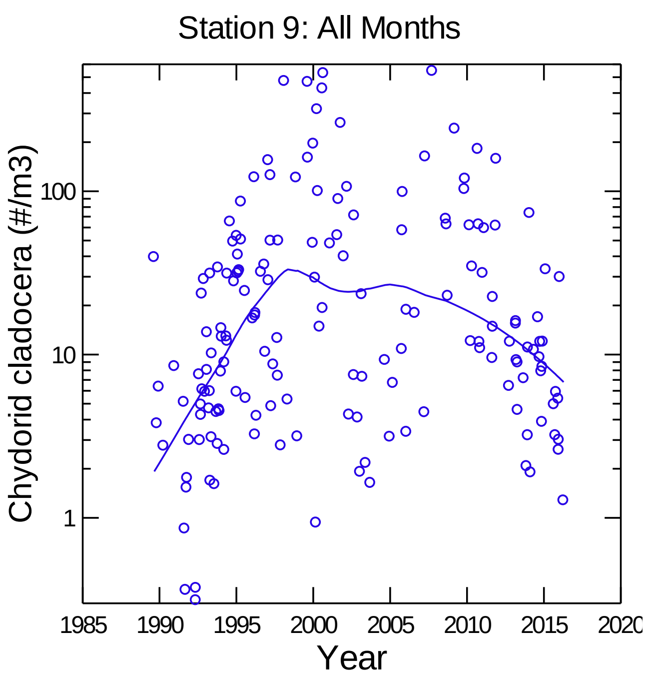
Daphnia in the river increased early on, but has since declined slightly (Figure 136). The trend line in 2016 approached 30/m^3, only slightly higher than the level observed at the beginning of the record in 1990. Regression analysis did not indicate a significant positive trend over the study period (Table 16).

Figure 136. Long term trend in *Daphnia*. Station 9. River mainstem.



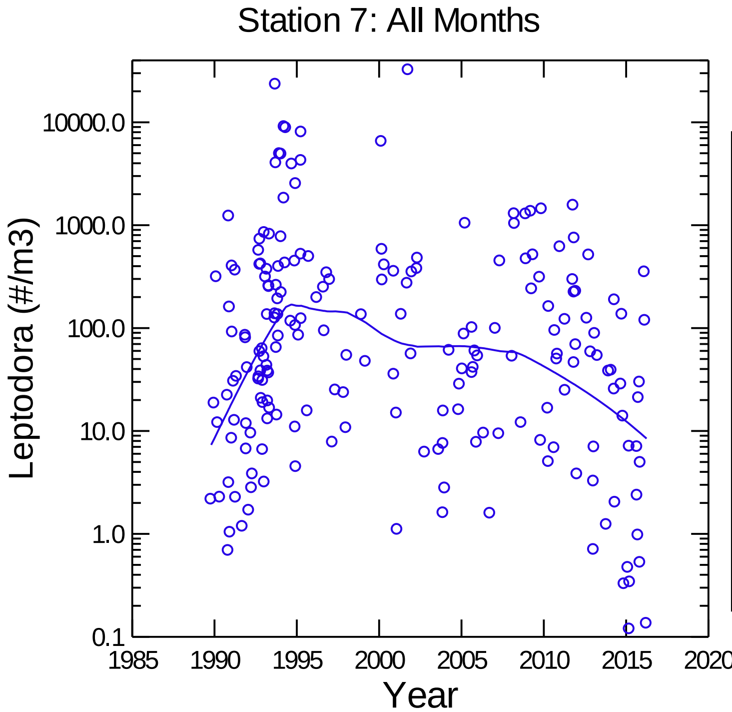
Chydorid cladocera in the cove have maintained a consistent population of about 10-20/m^3, substantially higher than the low of 4/m^3 in the early 1990's, but below trend line values of 30/m^3 observed in 2000 (Figure 137). Regression analysis gave evidence for a slight linear increase over the study period (Table 16).

Figure 137. Long term trend in Chydorid Cladocera. Station 7. Gunston Cove.



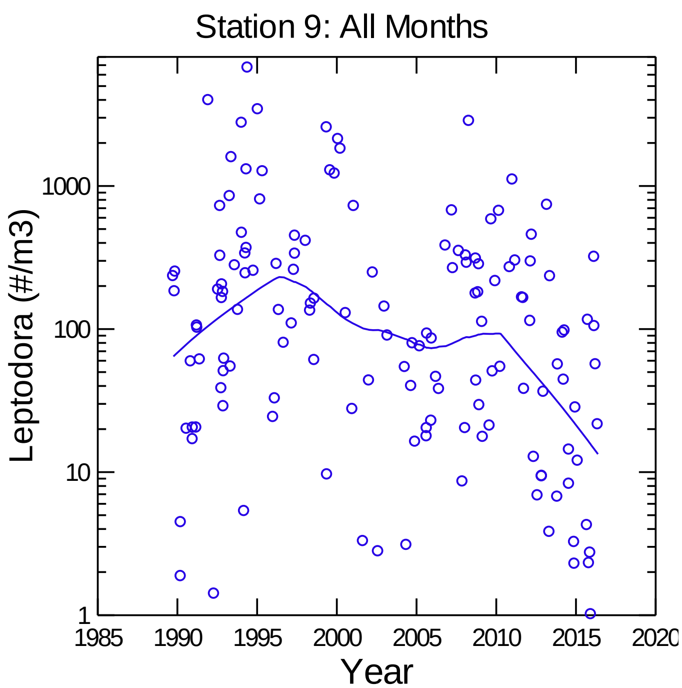
In the river chydorids continued a gradual decrease to about 7/m^3, slightly above the low of about 2/m^3 in the early 1990's (Figure 138). There was no evidence for a significan linear trend (Table 16).

Figure 138. Long term trend in Chydorid Cladocera. Station 9. River mainstem.



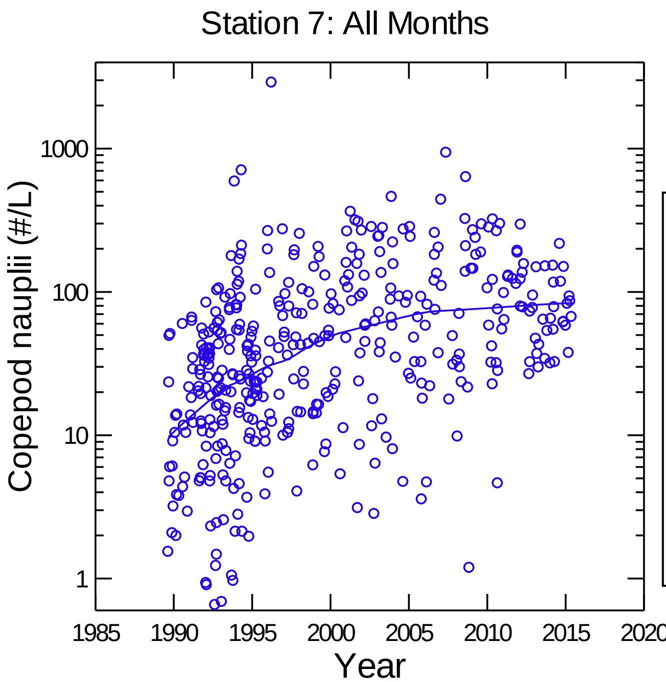
In the cove the trend line for *Leptodora*, the large predaceous cladoceran, has gradually decreased to about 8/m^3, down from its high of about 150/m^3 in 1994, and similar the 1990 value (Figure 139). There was weak evidence for a significant negative linear trend in *Leptodora* over the entire study period (Table 16).

Figure 139. Long term trend in *Leptodora*. Station 7. Gunston Cove.



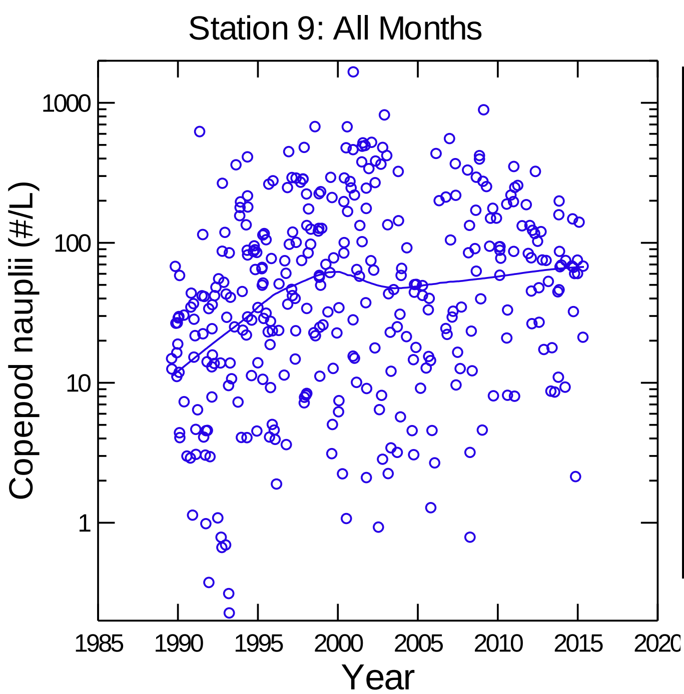
In the river, *Leptodora* densities continued a general decline which began in 1995 resulting in trend line values of about 15/m^3 for 2016 (Figure 140). These values are almost equal to those observed in 1990, and are well below the peak of 200/m^3 in 1994. Linear regression analysis detected a significant linear trend when the whole study period was considered (Table 16).

Figure 140. Long term trend in *Leptodora*. Station 9. River mainstem.



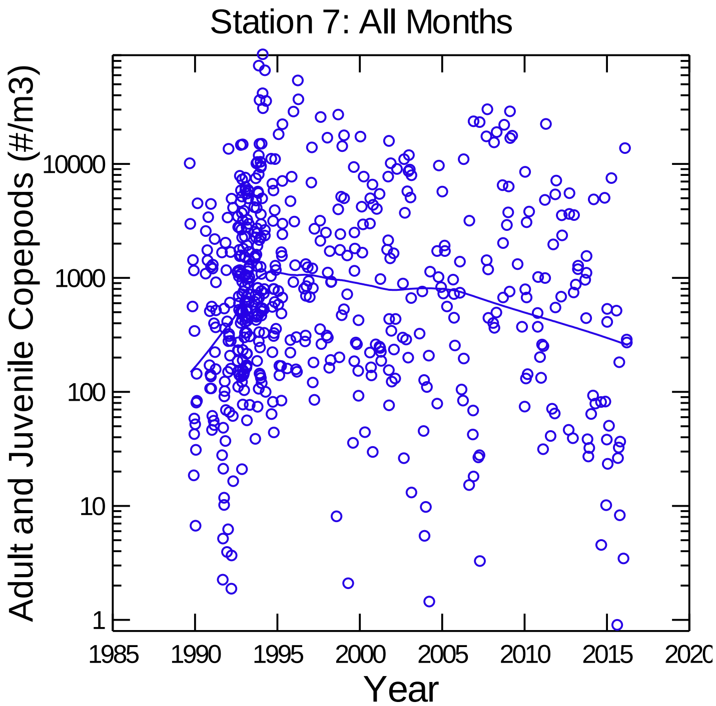
Copepod nauplii, the immature stages of copepods, continued their upward trend in the cove reaching almost 90/L (Figure 141). These values are well above the initial values of about 10/L in 1990. A strong linear increase was observed over the study period (Table 16).

Figure 141. Long term trend in Copepod Nauplii. Station 7. Gunston Cove.



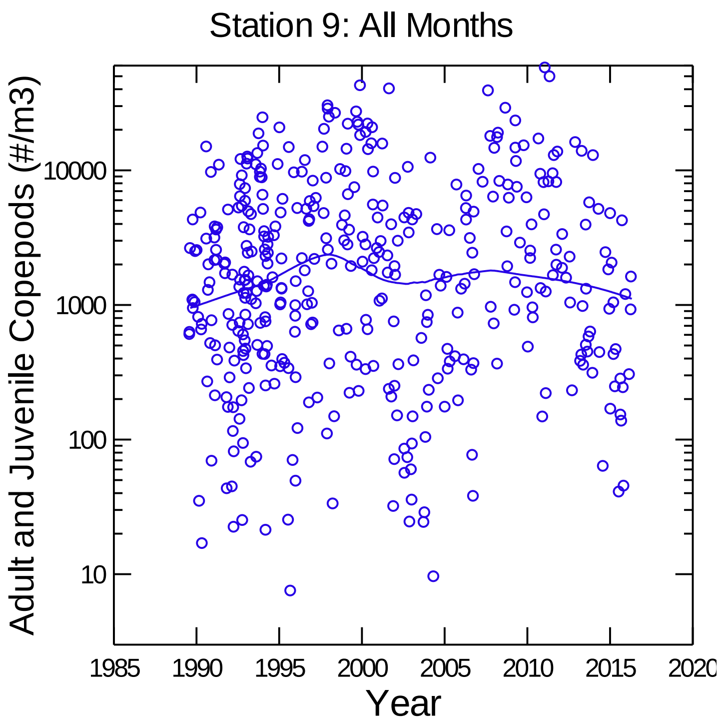
In the river, copepod nauplii showed a gradual increase since 2005 (Figure 142). The 2016 LOWESS trend line value was about 70/L, up from an initial value of 10/L in 1990, similar to the previous peak. A significant linear increase was found for nauplii over the study period (Table 16).

Figure 142. Long term trend in Copepod Nauplii. Station 9. River mainstem.



In the cove, adult and copepodid copepods increased strongly in the early 1990's and since have decreased slowly to about 250/m^3 (Figure 143). Copepods did not exhibit a significant linear trend in the cove over the study period (Table 16).

Figure 143. Long term trend in Adult and Copepodid Copepods. Station 7. Gunston Cove.



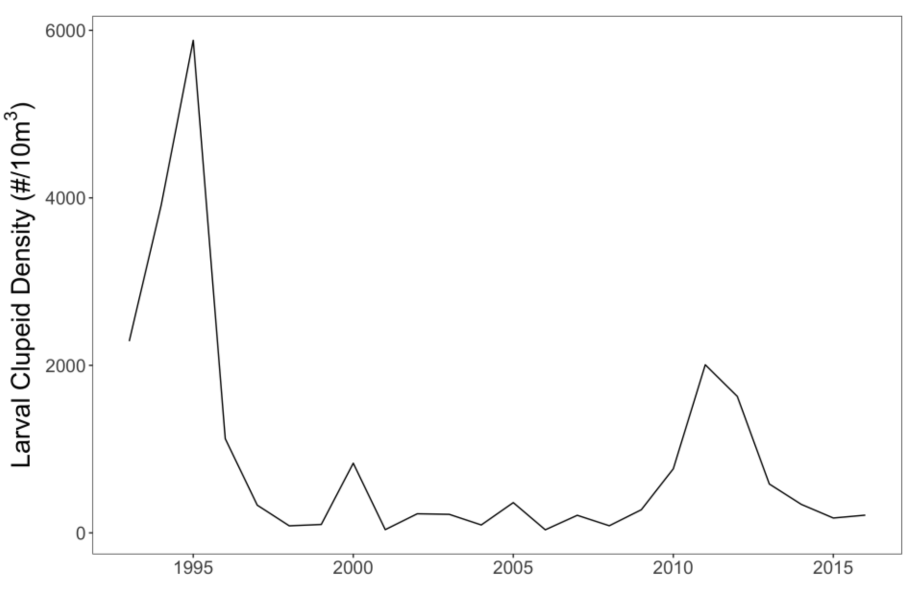
Adult and copepodid copepods have not changed greatly over the study period (Figure 144). The trend line in 2016 was about 1000/m^3 below the maximum of 2000/m^3 in 1998. No linear increase was found when the entire study period was considered (Table 16).

Figure 144. Long term trend in Adult and Copepodid Copepods. Station 9. River mainstem.

E. Ichthyoplankton Trends: 1993-2016

Ichthyoplankton monitoring provides a crucial link between nutrients, phytoplankton, zooplankton and juvenile fishes in seines and trawls. The ability of larvae to find food after yolk is consumed may represent a critical period when survival determines the abundance of a year-class. The timing of peak density of feeding stage fish larvae is a complex function of reproductive output as well as the temperature and flow regimes. These peaks may coincide with an abundance or scarcity of zooplankton prey. When the timing of fish larva predators overlaps with their zooplankton prey, the result is often a high abundance of juveniles that can be observed in high density in seines and trawl samples from throughout the cove. In addition, high densities of larvae but low juvenile abundance may indicate that other factors (e.g., lack of significant refuge for settling juveniles) are modifying the abundance of a year-class.

The dominant species in the ichthyoplankton samples, namely Clupeids (which are primarily river herring and Gizzard Shad), *Morone* sp. (mostly White Perch), Atherinids (Inland Silversides), and Yellow Perch, all exhibited a spike in density in 1995 followed by a decline in numbers until about 2008. The declines in Clupeid larvae were followed by increases starting in 2010 (Figure 145; Table 17). Especially 2010-2012 showed very high density of these larvae, while numbers decreased again in 2013. With continued relatively low densities in 2014 and 2015, the high densities of 2010-2012 appear to be a peak rather than a rebound to higher densities. It is possible that this is natural variation, and that these populations rely on a few highly successful year-classes.



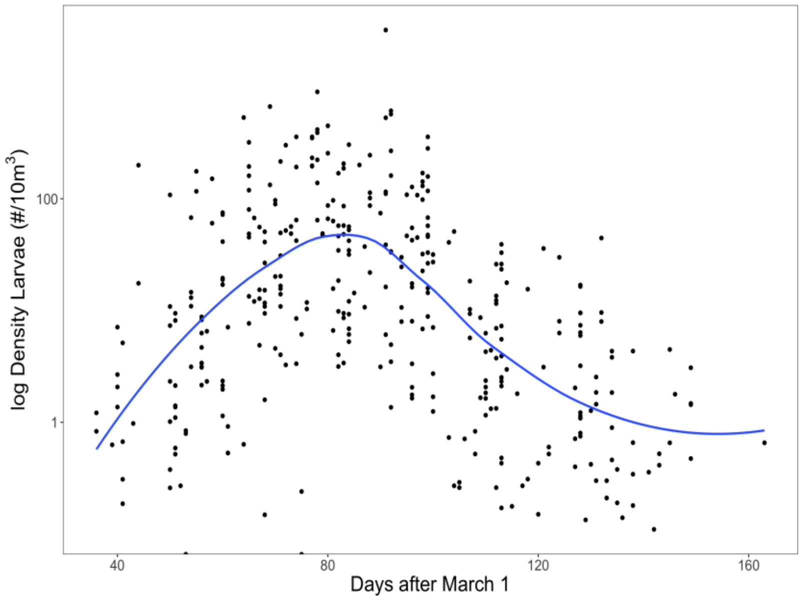
A graph of clupeid fish larvae averaged over all stations from 1993 through 2016 is shown in Figure 140. A peak from 2010-2012 sustains the population after years of declining densities since the mid 1990’s.

Figure 145. Long-term trend in Clupeid Larvae (abundance 10 m^{-3}).

Table 17. Density of larval fishes Collected in Gunston Cove and the Potomac mainstem (abundance per m^3)

Taxon	**Common Name**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**
Alosa sp.	Alewife, herring, or shad	17.20	3.60	37.90	246.60	149.10	1005.70	53.00	122.72	41.31	104.61
Dorosoma sp.	Gizzard and Threadfin Shad	208.7	85.0	276.1	1032.0	2006.9	1334.9	25.00	115.72	28.90	87.45
Lepomis sp.	sunfish	0.00	0.00	0.00	0.20	2.00	3.70	6.00	0.21	0.22	2.23
Morone sp.	perch and bass	39.60	60.5	58.10	88.10	62.80	640.90	27.00	0.21	1.70	100.61
Perca flavescens	Yellow Perch	12.00	1.20	0.30	14.80	0.40	0.40	3.00	1.04	0.00	0.14
Menidia beryllina	Inland Silverside	5.30	1.40	1.70	10.50	2.50	21.50	7.00	1.28	20.68	6.74

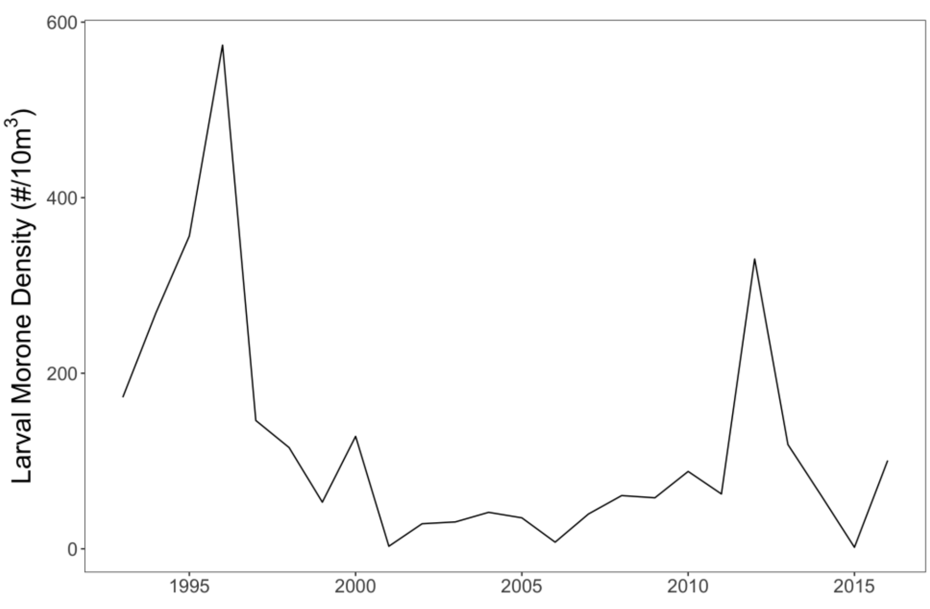
The peaks in abundance over the season reflect characteristic spawning times of each species (Figures 146, 148, 150, and 152). The earliest peak is from Yellow Perch (Figure 152), which may even be at its highest before our sampling starts. An early peak is also seen for *Morone sp*., which is mostly White Perch (Figure 148). White Perch begin spawning early and larval densities slowly taper off. Consequently, White Perch larvae are found throughout most of the sampling season. Clupeid larval density shows a distinct peak mid-May (Figure 146). Clupeid larvae are dominated by Gizzard Shad, which spawns later in the season than river herring (Alewife and Blueback Herring). However, river herring larvae are part of this peak as well; although their spawning season is from mid-March to mid-May, spawning occurs higher upstream, and larvae subsequently drift down to Gunston Cove. Silversides have a less pronounced peak in early June, with low densities continuing to be present throughout the season (Figure 150).



The seasonal pattern in clupeid larvae for 1993-2016 (Figure 146) shows that a peak in density occurs about 80 days after March 1, or mid-May. The occurrence of the peak late in the spring may indicate a dominance of Gizzard Shad larvae in the samples. Since river herring are spawned more upstream, this late spring peak also shows the time it takes for these larvae to drift down to Gunston Cove after river herring spawning season of mid-March to mid-May.

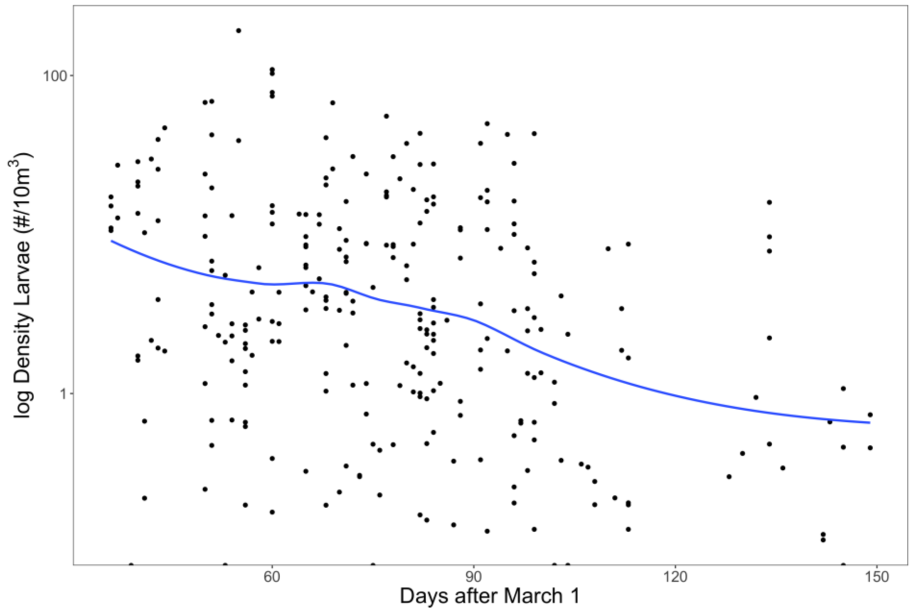
Figure 146. Seasonal pattern in Clupeid larvae (*Alosa* sp. and *Dorosoma* sp.; abundance 10 m^{-3}). The x-axis represents the number of days after March 1.

The long-term trend in annual average density of *Morone* larvae shows a high similarity with that of Clupeid larvae (figure 147). While densities are lower, the same pattern of high peaks in 1995 and 2012, and low densities in other years is seen. Looking at the seasonal pattern, we may miss high densities of larvae occurring in spring, as our sampling of larvae in Gunston Cove starts mid-April. With the high abundance of juveniles and adults each year, our *Morone* larval sample is likely not representative of the total larval production. White perch is also a migratory species, and juveniles may come in the system from elsewhere.



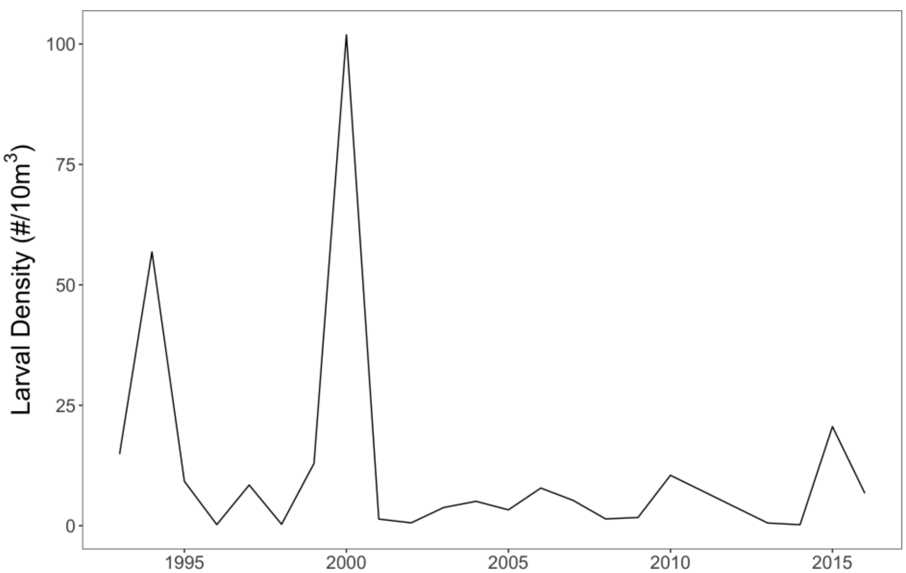
The trend in number of White Perch and Striped Bass larvae per 10 m^3 since 1993 is depicted in the graph in Figure 142. A slow increase can be seen since 2004, with a high peak in 2012, and subsequent decreases until 2015. 2016 may be the start of an upward trend.

Figure 147. Long term trend in *Morone* sp. larvae (abundance 10 m^{-3}).



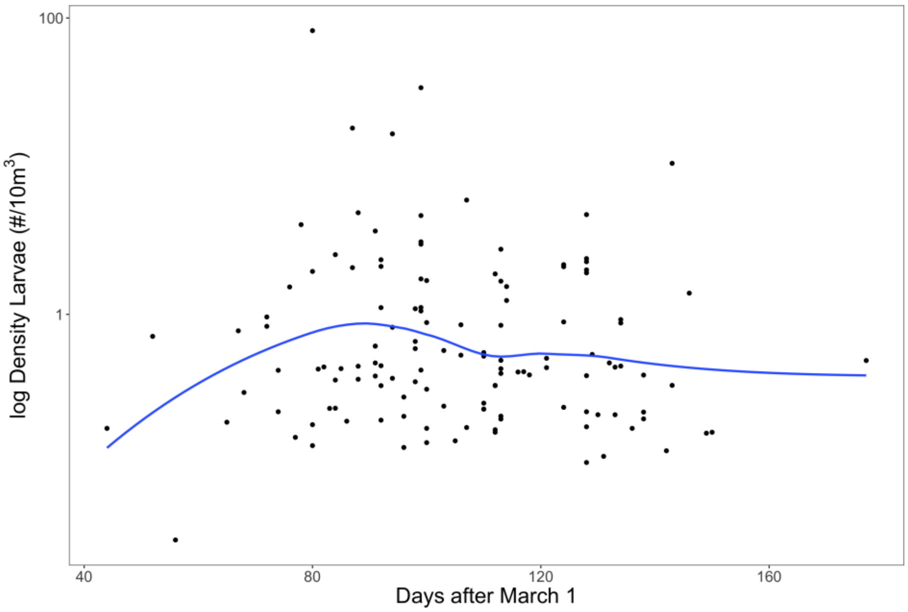
The seasonal density of *Morone* sp. larvae per 10 m^3 is shown in Figure 148. A small peak can be seen about 60 days after March 1 followed by declines in density thereafter. This peak occurs end of April, while highest densities are at the start of the sampling season.

Figure 148. Seasonal pattern in *Morone sp.* larvae (abundance 10 m^{-3}). X-axis represents days after March 1st.



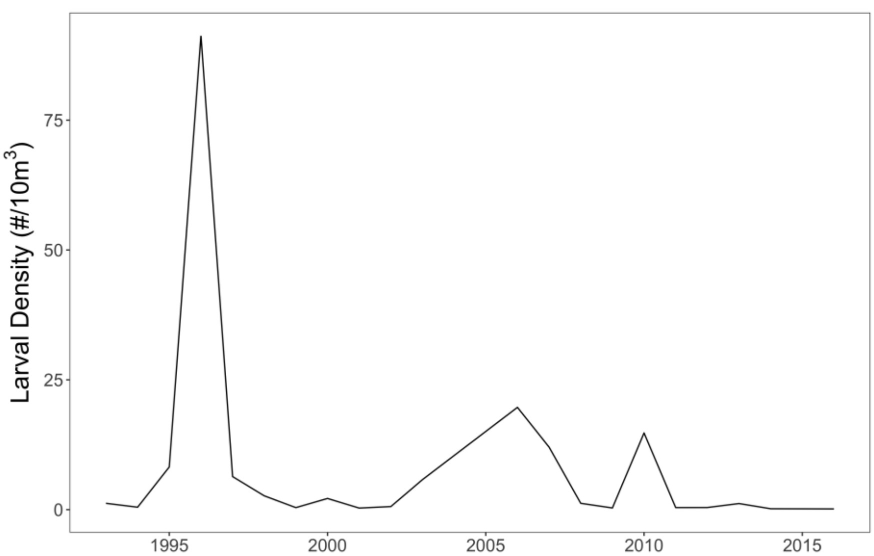
The long-term trend in density of Atherinid larvae (probably all Inland Silverside) is presented in Figure 149. After a high peak in 2000, densities have been moderate to low with some small peaks in 2006, 2010, and 2015.

Figure 149. Long-term trend in *Menidia beryllina* larvae (abundance 10 m^{-3}).



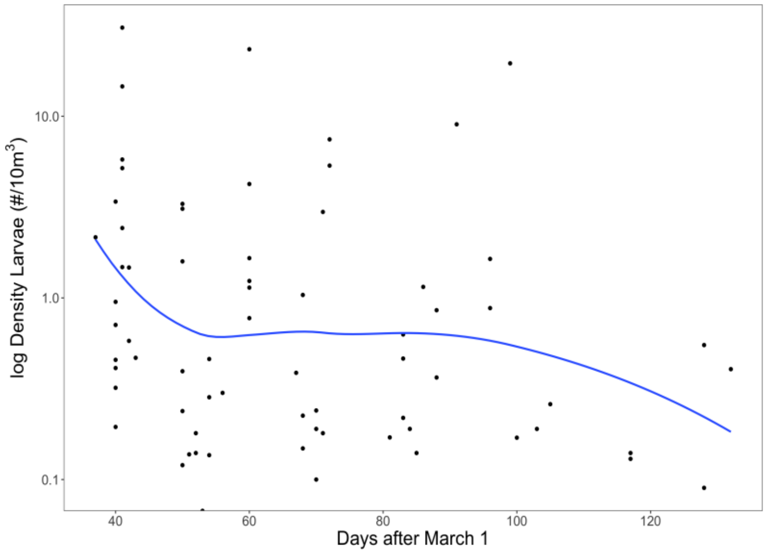
The seasonal occurrence of Atherinid larvae per 10m[3] is shown in a LOWESS graph in Figure 150. The pattern shows maximum density around 90 days after March 1, or around the first week of June.

Figure 150. Seasonal pattern in *Menidia beryllina* larvae (abundance 10 m^{-3}). The x-axis represents the number of days after March 1.



The long-term trend in density of Yellow Perch larvae since 1993. Following unusually high densities in 1996, abundances decreased while the general trend remains variable.

Figure 151. Long-term trend in Yellow Perch larvae (abundance 10 m^{-3}).



The long term pattern of seasonal occurrence of Yellow Perch larval density is presented in a LOWESS graph in Figure 152. The greatest densities occur in early to mid April, while spawning continues producing low densities throughout the season. Total density is low, which is likely the main reason for this unpronounced spawning pattern.

Figure 152. Seasonal pattern in Yellow Perch larvae (abundance 10 m^{-3}). The x-axis represents the number of days after March 1.

F. Adult and Juvenile Fish Trends: 1984-2016

Trawls

Overall patterns

Annual abundance of juvenile fishes inside Gunston Cove is indexed by mean catch per trawl in the inner cove (stations 7 and 10 combined; Table 18, Figure 153). Since 1984, this index has fluctuated by over an order of magnitude, and the pattern was predominately due to changes in the catch rate of White Perch (Figure 153). The one high peak in 2004 that was not caused by high White Perch abundance was caused by a large catch of Blueback Herring (Figure 154). On average, catch rates of fishes within the cove are approximately the same over the time of the survey; in other words, there is no significant increasing or decreasing trend over time. The overall catch rate for the inner cove (stations 7 and 10) in 2016 is similar to previous years and about the average of the last two years. Trawl catches in station 7 and 10 were dominated by White Perch and Spottail Shiner. *Alosa sp.* (herring or shad) were represented in the catches with high abundances as well.

Table 18. Mean catch per trawl of adult and juvenile fishes at Stations 7 and 10 combined. 1984-2016.

Year	All Spp.	White Perch	All Alosa Sp.	Blueback Herring	Alewife	Gizzard Shad	Bay Anchovy	Spottail Shiner	Brown Bullhead	Pumpkin seed
2016****	170.4	121.7	12.7	0.0	0.1	0.1	0.3	13.7	0.3	1.2
2015****	284.2	172.3	34.4	26.1	4.2	0.2	0.1	64.4	0.1	1.1
2014*	92.1	46.1	10.4	2.1	1.3	0.2	1.4	15.6	0.3	0.5
2013***	205.4	132.8	17.5	8.9	4.1	0.1	1.4	35.9	0.6	1.8
2012*	164.5	128.7	1.7	0.1	0.2	3.3	0.4	11.8	0.6	2.1
2011**	92.7	43.5	1.9	0.0	0.3	0.2	0.0	19.9	0.1	2.0
2010*	371.3	247.8	108.5	0.2	6.5	2.1	0.4	6.0	0.3	1.3
2009	90.1	18.3	46.6	1.0	45.2	0.6	6.2	2.7	0.1	2.7
2008	134.7	31.5	0.3	0.0	0.1	8.0	0.5	4.1	1.1	12.6
2007	227.3	141.4	37.3	23.6	8.9	0.2	15.8	20.1	0.2	2.6
2006	26.1	9.6	2.7	1.6	0.6	0.2	2.3	3.0	0.4	1.8
2005	70.7	22.0	34.6	12.1	17.3	1.1	0.0	6.4	0.0	1.4
2004	408.4	23.4	373.2	337.5	33.1	0.9	0.6	8.0	0.0	0.5
2003	54.2	13.2	23.9	18.8	3.5	0.0	7.4	2.8	0.1	0.4
2002	80.1	15.1	39.5	9.8	28.5	0.1	15.8	0.6	0.0	1.7
2001	143.5	47.0	50.6	40.5	9.9	0.3	35.1	2.8	3.3	1.4
2000	68.1	53.3	5.4	3.6	1.9	2.3	1.7	1.3	1.9	0.6
1999	86.9	63.2	4.7	4.2	0.5	1.0	5.4	4.8	2.4	1.8
1998	83.2	63.8	3.0	2.2	0.8	0.5	3.7	6.4	0.9	1.6
1997	81.4	61.7	2.9	1.9	1.0	5.0	2.6	2.9	1.5	1.2
1996	48.0	35.4	4.1	2.5	1.6	0.5	0.2	2.6	0.5	2.1
1995	88.6	69.7	6.2	4.1	2.1	0.4	3.0	3.0	1.9	1.8
1994	92.2	66.9	0.8	0.8	0.0	0.1	0.5	6.2	3.2	2.7
1993	246.6	216.0	2.0	1.4	0.6	1.4	0.6	7.3	4.5	3.4
1992	112.9	81.6	0.3	0.3	0.0	0.9	0.8	2.4	11.5	5.1
1991	123.7	90.9	1.5	1.0	0.5	8.1	2.6	2.9	12.5	1.7
1990	72.8	33.3	25.1	21.9	3.3	0.1	1.1	1.1	10.0	0.5
1989	78.4	14.9	16.4	16.1	0.3	42.4	0.3	0.5	3.0	0.6
1988	96.0	45.1	19.9	11.2	8.8	12.7	8.3	1.8	5.3	0.9
1987	109.2	54.6	19.6	16.4	3.2	5.6	8.8	0.7	17.2	1.4
1986	124.6	65.4	25.9	1.9	24.0	4.1	4.2	0.5	18.4	0.6
1985	135.9	43.9	25.8	8.6	10.7	2.9	48.2	1.1	9.8	0.1
1984	223.5	132.7	12.0	6.1	0.6	13.4	22.8	1.7	36.1	0.3

*Station 10 not sampled late July – September **Station 10 not sampled in August, *** station 10 not sampled in August-September, ****Station 10 not sampled in June-September.

Mean catch at station 9 was lower in 2016 than in the previous two years, and below the long-term mean (54; Table 19). The total catch at station 9 may be declining over time, and it would be interesting to pursue the research question whether and how blue catfish invasion has played a role in that. Blue catfish is regularly collected at station 9 the last 15 years, and not at the inner cove stations. The mean catch of all stations combined in 2016 is right at the long-term mean of

103 (Table 20). There was high variability between stations, with mean catch per trawl at station 9 at an all-time low (Table 22). The presence and location of SAV beds is partially responsible for the variability. Trawling is impeded at station 10 in the summer, until trawling becomes impossible at varying dates late summer (Table 21). This is likely responsible for the lower catch in station 10 than station 7. It is clear from the lower catch per trawl in station 9 than 7 and 10, that the inner cove is preferred habitat for fishes.

Table 19. Mean catch per trawl of selected adult and juvenile fishes for all months at Station 9. 1988-2016.

Year	All Spp.	All Alosa Spp,	Alewife	Blueback Herring	White Perch	Bay Anchovy	Spottail Shiner	Brown Bullhead	Channel Catfish	Blue Catfish	Tessellated Darter
2016	10.1	2.0	0.0	0.0	2.0	4.9	0.0	0.0	1.2	0.0	0.0
2015	17.5	11.3	8.6	0.2	1.6	0.5	0.2	0.2	3.2	0.2	0.0
2014	16.8	6.7	3.7	1.0	3.0	3.3	0.1	0.1	3.1	0.0	0.4
2013	12.2	3.9	2.1	0.6	1.5	1.6	0.0	0.0	4.5	0.0	0.2
2012	62.1	0.0	0.0	0.0	21.6	31.7	0.8	0.0	7.3	0.3	0.0
2011	33.8	0.1	0.1	0.0	21.3	0.0	0.2	0.1	5.2	6.4	0.3
2010	38.8	0.1	0.0	0.0	10.8	7.9	0.0	0.1	19.6	0.0	0.0
2009	36.8	2.4	0.5	0.4	13.8	7.8	0.5	0.2	10.5	0.6	0.1
2008	234.6	0.3	0.0	0.0	26.4	199.9	0.8	0.1	6.8	0.0	0.0
2007	253.8	52.7	17.2	2.5	195.7	0.7	1.1	0.0	1.8	0.0	0.9
2006	68.1	0.2	0.0	0.2	31.0	3.0	0.2	8.0	19.9	4.6	0.0
2005	91.1	15.0	14.7	0.3	36.5	12.1	1.8	0.0	18.3	4.7	0.1
2004	41.9	3.8	3.4	0.3	20.4	0.0	1.1	0.0	5.2	6.6	0.3
2003	65.8	0.3	0.1	0.1	32.6	0.0	0.6	0.0	7.4	14.4	1.2
2002	55.3	1.3	0.7	0.4	28.2	0.5	0.1	0.0	6.8	10.8	1.0
2001	77.1	0.1	0.1	0.1	40.1	22.2	0.1	0.9	2.7	5.5	0.8
2000	52.1	0.1	0.1	0.0	43.4	0.0	0.1	2.1	0.0	3.9	0.0
1999	23.1	0.0	0.0	0.0	18.9	0.3	0.0	0.3	0.0	2.4	0.0
1998	22.3	0.1	0.1	0.0	12.9	0.4	0.1	0.3	0.0	6.3	2.0
1997	49.6	0.0	0.0	0.0	37.2	0.0	1.1	0.9	0.0	9.2	0.4
1996	14.0	0.0	0.0	0.0	7.0	0.0	0.1	0.1	0.0	6.0	0.8
1995	31.9	0.3	0.3	0.0	17.4	0.2	0.2	4.4	0.0	8.5	0.1
1994	31.9	0.0	0.0	0.0	13.4	0.1	0.0	2.4	0.0	6.3	3.5
1993	31.2	0.1	0.0	0.1	6.4	0.0	6.2	1.4	0.0	6.8	7.5
1992	29.0	0.1	0.0	0.1	13.4	0.0	0.2	1.1	0.0	1.8	3.3
1991	67.9	0.1	0.1	0.0	42.4	1.9	0.1	1.3	0.0	13.2	0.4
1990	101.5	0.1	0.1	0.0	50.6	0.0	0.1	5.5	0.0	39.9	0.1
1989	14.3	1.0	0.3	0.8	7.9	0.4	0.0	1.6	0.0	2.0	0.3
1988	19.3	0.3	0.3	0.0	5.3	11.5	0.0	0.0	0.0	0.8	0.0

Table 20. Mean catch per trawl of selected adult and juvenile fishes for all months at Stations 7, 9, and 10 combined. 1984-2016.

Year	All Spp.	White Perch	Blueback Herring	Alewife	All Alosa Spp.	Gizzard Shad	Bay Anchovy	Spottail Shiner	Brown Bullhead	Channel Catfish	Blue Catfish
2016 ****	103.6	71.8	0.0	0.0	8.3	0.0	2.2	8.0	0.2	0.5	0.0
2015 ****	168.3	98.1	14.8	6.1	24.3	0.1	0.3	36.5	0.1	1.4	0.1
2014 *	62.0	28.8	1.6	2.3	8.9	0.1	2.2	9.4	0.2	1.3	0.0
2013 ***	131.1	82.3	5.7	3.3	12.3	0.2	1.5	22.1	0.4	2.3	0.0
2012 *	123.5	85.8	0.0	0.1	1.0	2.0	12.9	7.4	0.4	2.9	0.2
2011 **	71.8	35.6	0.0	0.2	1.3	0.1	0.0	12.9	0.1	2.0	2.3
2010 *	246.6	158.9	0.1	4.1	67.9	1.3	3.2	3.8	0.3	8.0	0.0
2009	71.8	16.8	0.8	29.9	31.4	0.4	6.8	1.9	0.2	3.6	0.3
2008	163.3	30.0	0.0	0.1	0.3	5.7	57.5	3.1	0.8	2.2	0.1
2007	236.1	159.5	16.6	11.6	42.4	0.1	10.7	13.8	0.1	0.7	0.0
2006	41.1	17.3	1.1	0.4	1.8	0.1	2.5	2.0	3.1	7.1	1.6
2005	78.2	27.3	7.7	16.3	27.4	0.7	4.4	4.7	0.0	7.3	1.8
2004	271.0	22.3	211.1	22.0	234.7	0.5	0.4	5.4	0.0	2.0	2.5
2003	58.1	19.7	12.6	2.3	16.0	0.0	4.9	2.1	0.1	2.5	5.4
2002	71.7	19.6	6.6	19.0	26.5	0.1	10.6	0.4	0.0	4.1	4.6
2001	122.3	44.8	27.6	6.8	34.5	0.3	31.0	1.9	2.5	0.9	1.8
2000	65.3	48.8	2.3	1.9	4.2	1.5	1.1	2.1	1.9	0.0	1.3
1999	65.6	48.4	2.8	0.3	3.1	0.7	3.7	3.2	1.7	0.0	0.8
1998	62.9	46.8	1.4	0.6	2.0	0.4	2.6	4.3	0.7	0.0	2.1
1997	70.8	53.5	1.3	0.7	2.0	3.3	1.7	2.3	1.3	0.0	3.1
1996	32.4	22.4	1.3	0.8	2.1	0.3	0.1	1.4	0.3	0.0	2.5
1995	74.0	53.3	2.5	1.3	3.8	1.1	3.0	2.3	1.9	0.0	4.8
1994	87.2	63.8	1.2	0.1	1.3	0.1	0.6	6.6	1.7	0.0	2.1
1993	162.4	131.7	2.0	0.4	2.3	1.0	2.2	7.6	1.9	0.0	2.1
1992	119.8	88.2	0.6	0.7	1.3	0.4	1.0	2.3	4.5	0.0	1.5
1991	150.5	82.5	13.1	5.1	18.2	5.4	26.6	2.9	4.7	0.0	2.6
1990	69.1	31.6	16.5	3.4	19.9	0.1	0.8	2.4	4.4	0.0	7.1
1989	64.1	9.3	27.1	0.5	27.7	22.0	0.6	0.4	1.5	0.0	0.6
1988	65.8	21.2	14.1	3.8	17.9	6.4	13.3	1.2	2.4	0.0	0.3
1987	104.1	49.7	14.1	1.2	15.3	6.5	20.5	1.2	7.2	0.0	0.1
1986	82.4	47.6	2.7	11.8	14.5	2.4	5.3	0.5	7.5	0.0	0.1
1985	93.1	33.0	7.7	5.6	18.7	1.4	29.4	1.4	4.6	0.0	0.3
1984	157.7	100.6	5.1	0.4	8.3	6.5	18.1	2.2	15.4	0.0	0.5

*Station 10 not sampled late July – September **Station 10 not sampled in August, *** station 10 not sampled in August-September, ****Station 10 not sampled in June-September.

Table 21. The number of trawls per station in each month at Stations 7, 9, and 10 in each year.

Year	Station	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2016	7	0	0	1	2	2	2	2	1	0	0	0
2016	9	0	0	1	2	2	2	2	1	0	0	0
2016	10	0	0	1	2	1	0	0	0	0	0	0
2015	7	0	0	1	2	2	2	2	1	0	0	0
2015	9	0	0	1	2	2	2	2	1	0	0	0
2015	10	0	0	1	1	0	0	0	0	0	0	0
2014	7	0	0	1	2	2	2	2	1	0	0	0
2014	9	0	0	1	2	2	2	2	1	0	0	0
2014	10	0	0	1	2	2	0	0	0	0	0	0
2013	7	0	0	1	2	2	2	2	1	0	0	0
2013	9	0	0	1	2	2	2	2	0	0	0	0
2013	10	0	0	1	2	2	1	0	0	0	0	0
2012	7	0	0	1	2	2	2	2	1	0	0	0
2012	9	0	0	1	2	2	2	2	1	0	0	0
2012	10	0	0	1	2	2	0	0	0	0	0	0
2011	7	0	0	1	2	3	2	2	1	0	0	0
2011	9	0	0	1	2	3	2	2	1	0	0	0
2011	10	0	0	1	2	3	2	0	1	0	0	0
2010	7	0	0	1	1	2	2	2	1	0	0	0
2010	9	0	0	1	1	2	2	2	1	0	0	0
2010	10	0	0	1	1	2	1	0	0	0	0	0
2009	7	0	0	1	2	2	2	2	1	0	0	0
2009	9	0	0	1	2	2	2	2	1	0	0	0
2009	10	0	0	1	2	2	2	2	1	0	0	0
2008	7	0	0	1	2	2	2	2	1	0	0	0
2008	9	0	0	1	1	2	1	2	1	0	0	0
2008	10	0	0	1	2	2	2	2	1	0	0	0
2007	7	0	0	1	2	2	2	2	1	0	0	0
2007	9	0	0	1	2	2	2	2	1	0	0	0
2007	10	0	0	1	2	2	2	2	1	0	0	0
2006	7	0	0	1	2	2	2	2	1	0	0	0
2006	9	0	0	1	2	2	2	2	1	0	0	0
2006	10	0	0	1	2	2	1	2	0	0	0	0
2005	7	0	0	1	2	2	2	2	1	1	0	0
2005	9	0	0	1	2	2	2	2	1	1	0	0
2005	10	0	0	1	2	2	1	2	0	0	0	0
2004	7	0	0	0	1	2	2	2	1	0	0	0
2004	9	0	0	1	1	2	2	2	1	0	0	0
2004	10	0	0	0	1	2	2	1	1	0	0	0
2003	7	0	1	0	1	2	2	1	1	1	0	0
2003	9	0	1	2	1	2	2	1	1	1	1	1
2003	10	0	0	0	1	2	2	1	1	0	1	0
2002	7	0	1	2	1	2	2	2	2	1	0	1
2002	9	0	1	2	2	2	2	2	2	1	0	0
2002	10	0	0	2	2	2	2	2	2	1	0	0
2001	7	0	1	2	2	1	2	3	2	1	1	1
2001	9	0	1	2	1	1	2	3	2	1	1	1
2001	10	0	1	2	2	1	2	3	2	1	1	1

Year	Station	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2000	7	0	1	2	2	3	2	2	2	1	1	1
2000	9	0	1	2	2	3	2	2	2	1	1	1
2000	10	0	1	2	2	3	2	2	2	1	1	0
1999	7	0	1	2	2	2	2	2	2	0	1	1
1999	9	0	1	1	2	2	2	2	2	1	1	1
1999	10	0	1	2	2	2	2	2	2	1	1	1
1998	7	0	1	2	2	2	2	2	2	1	1	1
1998	9	0	1	2	2	2	2	2	2	1	1	1
1998	10	0	1	2	2	2	2	2	2	1	1	1
1997	7	0	1	2	2	2	2	2	2	2	1	1
1997	9	0	1	2	2	2	2	2	2	2	1	1
1997	10	0	1	2	2	2	2	2	2	2	1	1
1996	7	0	1	2	2	1	2	1	2	1	1	1
1996	9	0	1	2	2	1	2	1	2	1	1	1
1996	10	0	1	2	1	2	2	1	2	1	1	1
1995	7	0	1	2	2	2	2	2	2	2	1	0
1995	9	0	1	2	2	2	2	2	2	2	1	0
1995	10	0	1	2	2	2	2	2	2	2	1	0
1994	7	0	1	1	1	2	2	0	2	2	1	0
1994	9	0	0	1	1	2	2	0	2	2	1	0
1994	10	0	1	1	1	2	2	0	2	2	1	0
1993	7	0	0	1	2	2	3	2	2	2	1	1
1993	9	0	1	1	2	2	3	2	2	2	0	1
1993	10	0	0	1	2	2	3	2	2	2	1	1
1992	7	0	1	1	1	1	1	1	1	1	1	1
1992	9	0	1	1	0	1	1	1	1	1	0	0
1992	10	0	1	1	1	1	1	1	1	1	1	1
1991	7	0	1	1	1	1	1	1	1	1	1	0
1991	9	0	1	1	1	1	1	1	1	2	1	0
1991	10	0	1	1	1	1	1	1	1	1	1	0
1990	7	0	1	1	1	1	1	1	1	1	0	0
1990	9	0	1	0	1	1	1	1	1	1	0	0
1990	10	0	0	1	1	1	1	1	1	1	0	0
1989	7	0	1	1	1	1	1	2	2	1	1	0
1989	9	1	1	1	0	0	1	2	2	1	1	0
1989	10	0	1	1	1	1	1	2	2	1	1	0
1988	7	0	1	1	1	2	2	2	2	1	1	0
1988	9	0	0	0	0	0	0	0	2	1	1	0
1988	10	0	1	1	1	2	2	2	2	1	1	0
1987	7	0	1	1	1	1	1	1	1	1	1	0
1987	10	0	1	1	1	1	1	1	1	1	0	0
1986	7	0	1	1	1	1	1	1	1	1	1	0
1986	10	0	1	1	1	1	1	1	1	1	1	0
1985	7	0	0	1	1	1	0	1	1	2	1	0
1985	10	0	0	1	1	1	0	1	1	2	1	0
1984	7	0	0	2	3	2	3	2	3	4	2	1
1984	10	0	1	2	4	2	3	2	3	4	2	1

Table 22. Mean catch per trawl of adult and juvenile fishes in all months at each station.

Year	Station 7	Station 9	Station 10
2016****	224.3	9.8	35.75
2015****	360	17.5	47.5
2014*	114.4	24	70.4
2013***	234.2	12.2	30.2
2012*	217.7	60.5	21.2
2011**	114	34	72.2
2010*	615.6	38.6	5.8
2009	142.8	40.4	45.3
2008	50.1	95	91.3
2007	390.1	253.8	64.4
2006	40.7	68.1	6.2
2005	104.6	91.1	21.4
2004	658.2	41.9	22.4
2003	61.3	62.5	39.4
2002	91.2	52.9	70.9
2001	157.9	68	112.1
2000	95.1	52.4	44.8
1999	117.2	23.1	56.6
1998	88.3	22.1	78.1
1997	111.5	49.6	51.4
1996	64.5	14	31.5
1995	107.6	31.9	69.6
1994	122.3	31.9	62.1
1993	354.9	31.3	109.2
1992	155.5	27.5	70.2
1991	173.9	67.9	73.6
1990	77.3	101.5	68.4
1989	52.6	14.3	104.3
1988	95.8	19.3	96.2
1987+	84.3	-	131.9
1986+	95.8	-	153.4
1985+	122.6	-	146.1
1984+	197.1	-	141.3

*Station 10 not sampled late July – September **Station 10 not sampled in August, *** station 10 not sampled in August-September, ****Station 10 not sampled in June-September, +Station 9 was not sampled from 1984-1987.

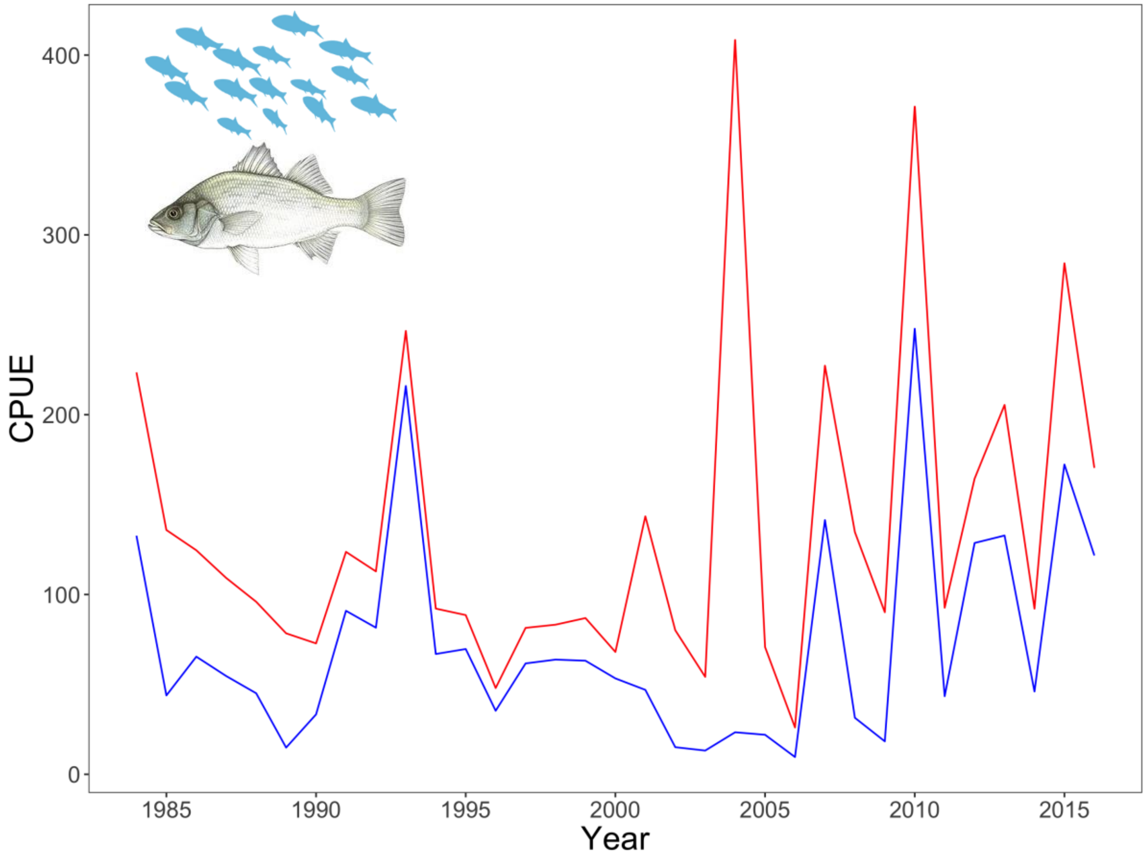


Figure 153. Trawls. Annual Averages. Catch per Trawl (CPUE). All Species (red) and White Perch (blue). Cove Stations 7 and 10. 1984-2016.

Mean total number of fish per trawl sample has remained steady over the course of the study; the pattern is highly dominated by catches of White Perch (Figure 153). Strong cohorts punctuated White Perch catch rates in 1993, 2007, 2010, 2012, and 2015. Overall, White Perch catches have remained similar and stable over the period of record. The higher frequency of strong year-classes after 2005 results in an overall small increase in trend starting that time.

The remaining component of the total catch (species other than White Perch) made up a moderate to large proportion of the catch until 1990; a relative small part of the catch between 1991 and 2000; and moderate to large proportion of the catch from 2001 to 2015. There was a high peak in catches other than White Perch in 2004, which was primarily due to exceptionally high catches of Blueback Herring (Figure 153; Figure 154). Annual trends in other dominant species captured by the trawl survey are presented below.

The high peak in Blueback Herring catches in 2004 stands out in otherwise low catches (Figure 155). Generally, both herring species have been found in higher abundances since 2000 than in the decade before that. We included *Alosa sp*. (unidentified herring or shad) in Figure 149 in 2016, so that abundances of herring or shad are not missed simply because they could not be identified to the species level. This revealed the second highest peak in Alosines in 2010 not previously reported.

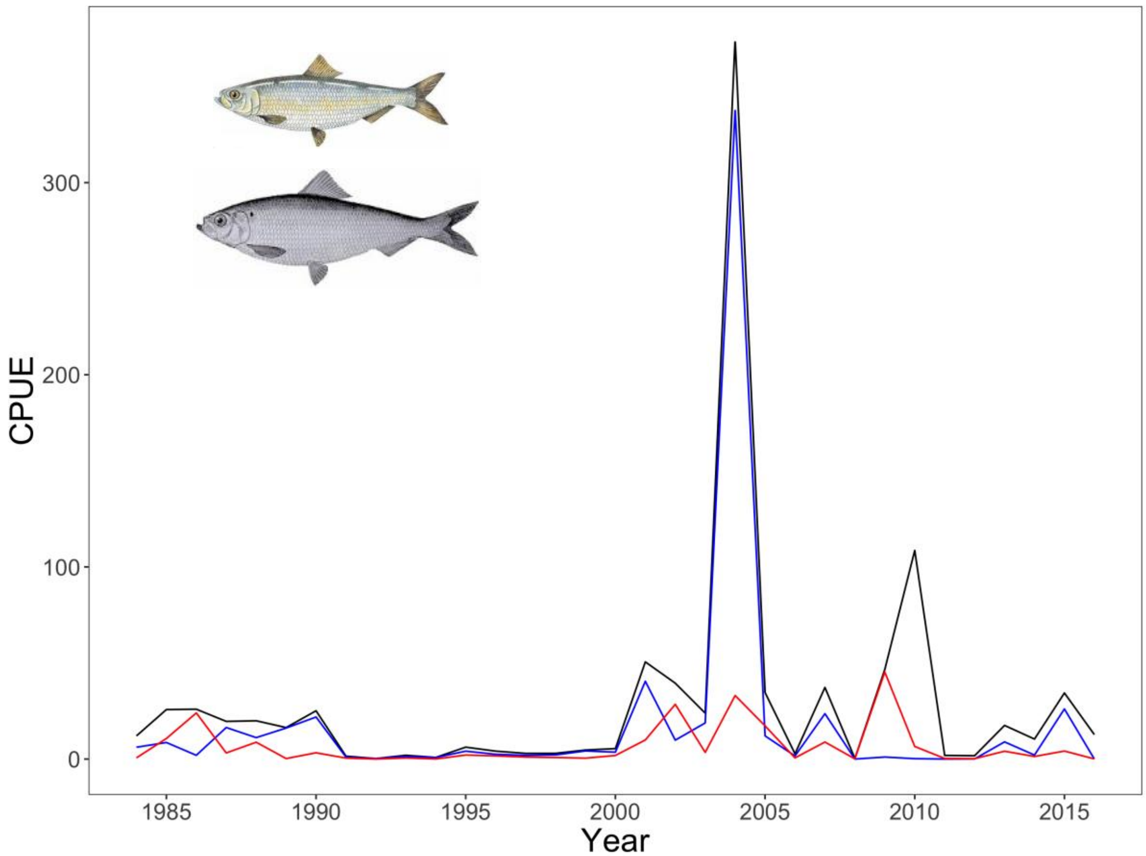


Figure 154. Trawls. Annual Averages. Blueback Herring (blue) and Alewife (red) and *Alosa sp*. (unidentified herring or shad; black). Cove Stations 7 and 10.

Gizzard Shad catch rates in trawls in 2016 were low which contributes to a pattern of low abundance after a high peak in 1989 (Figure 155). Smaller peaks later occurred in 1991, 1997, 2008, and 2012, that were all an order of magnitude lower than the 1989 peak. Bay Anchovy catch rates in 2016 were low like they were in 2015 at inner cove stations, and trends in the data suggests a sinusoidal but decreasing trend over the length of the survey. They are primarily resident in more saline portions of the estuary, and display sporadic occurrence in tidal freshwater. Any decreases in Gunston Cove therefore do not indicate a declining trend in the abundance of this species overall. Further years will determine whether the sinusoidal trend continues, or if the ecosystem of the inner cove has now shifted to a state (e.g. reduced open water/SAV bed ratio) that is less favorable for Bay Anchovy.

Spottail Shiner and sunfishes (Bluegill and Pumpkinseed) have been consistently collected in the majority of all trawl and seine samples (Figure 156). An increasing trend has been observed for Spottail Shiner since the beginning of the survey. In recent years (since 2000), a more sharply increasing pattern is seen in the midst of high variability, with high numbers in 2007, 2011, 2013, and 2015 (Figure 156). We collected an unprecedented high number of Spottail Shiner specimens in 2015. These individuals were mostly juveniles, indicating relatively high reproductive success as measured by this survey. 2016 had a much lower average catch again, but not to the extent that it changed the increasing trend in time. The trends for pumpkinseed showed lower overall abundance, with a decrease in abundance after a 2008 peak.

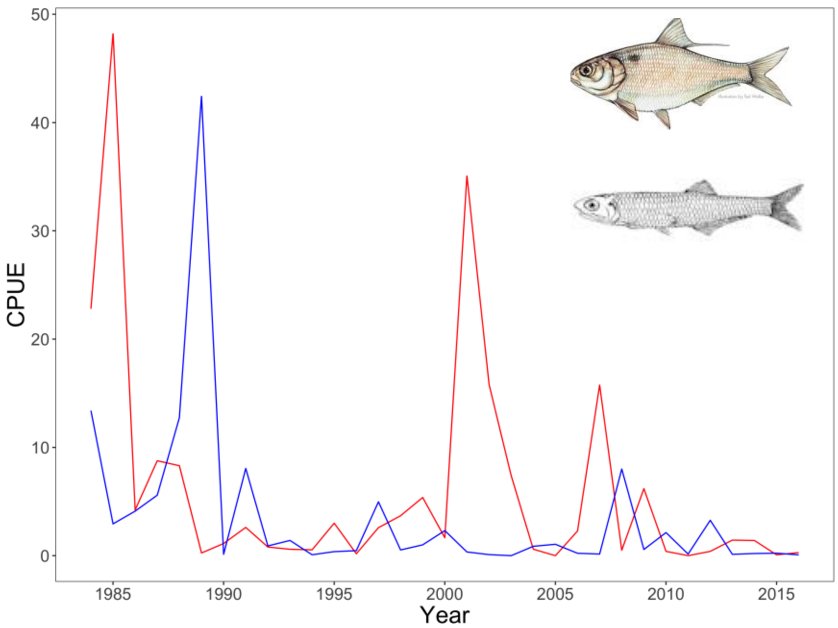


Figure 155. Trawls. Annual Averages. Cove Stations 7 and 10. Gizzard Shad (blue) and Bay Anchovy (red).

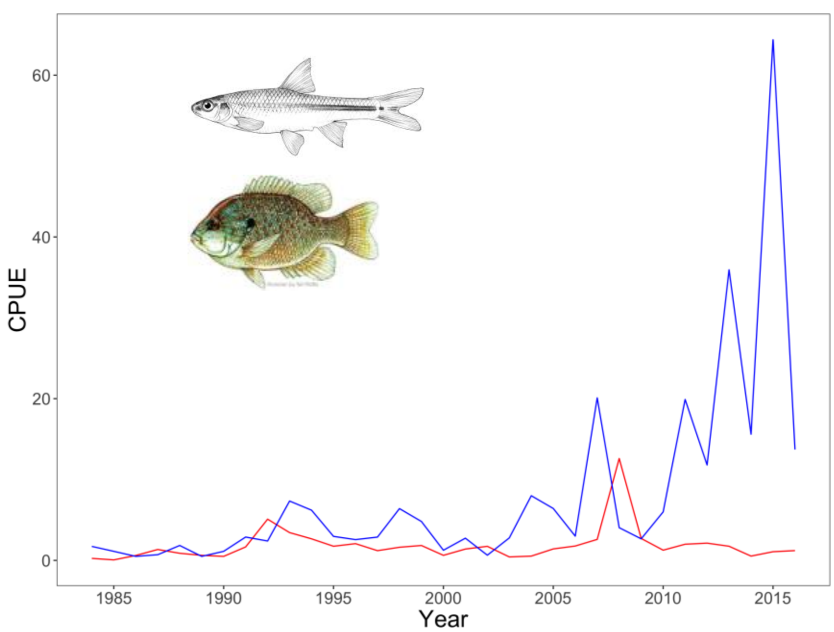


Figure 156. Trawls. Annual Averages. Spottail Shiner (blue) and Pumpkinseed (red). Cove Stations 7 and 10.

Very few Brown Bullhead specimens were captured in trawls in 2016, continuing a declining trend that has proceeded continuously since the start of the survey (Figure 157a). We did see a very small increase as compared to 2015. The fyke nets collect Brown Bullhead in low amounts as well, and next year's report will include a trend through time of fyke net collections as well, to ensure that all collected native catfishes are represented in the discussion. In 2016, we only collected 1 Brown Bullhead in the fyke nets.

Tessellated Darter was consistently encountered at low abundance in trawl samples. While average values remain low, the second highest peak in the period of record was recently observed in 2014, and the mean per trawl was relatively high in 2016 again (Figure 157b).

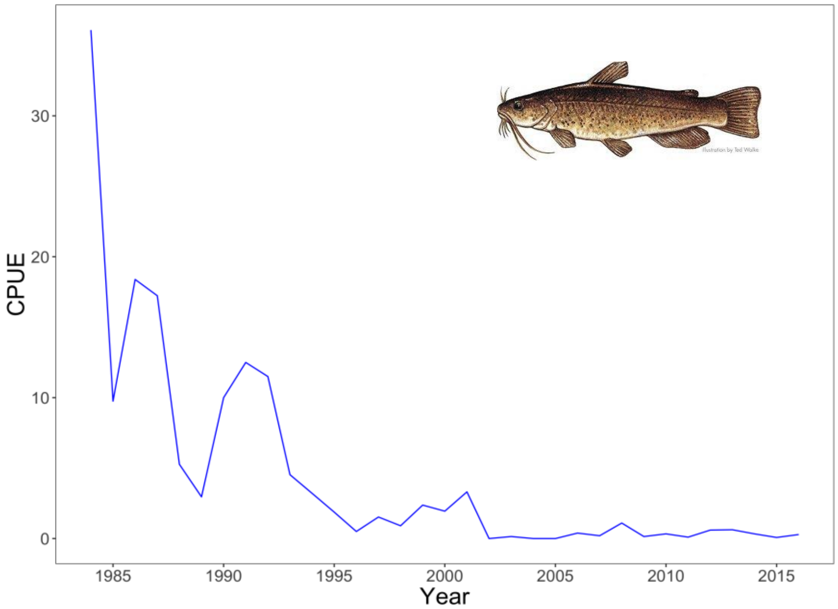


Figure 157a. Annual Averages. Brown Bullhead. Cove Stations 7 and 10.

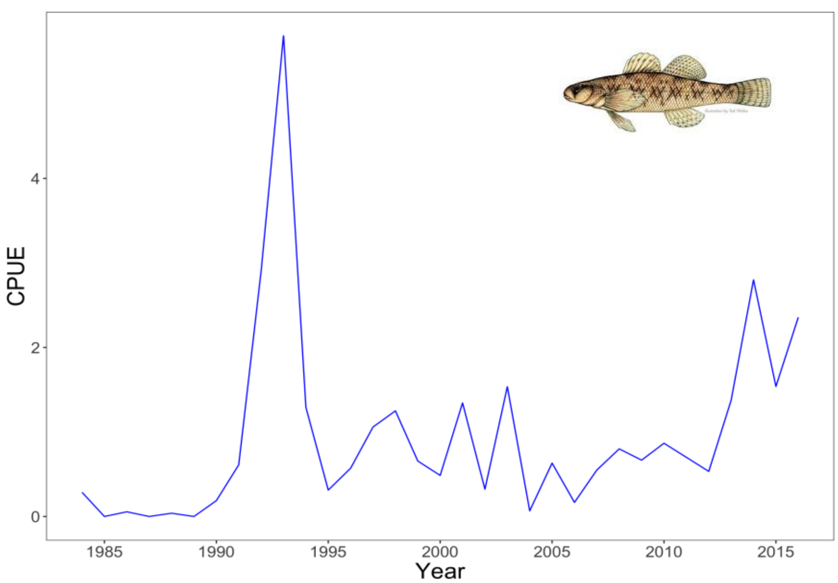


Figure 157b. Trawls. Annual Averages of Tessellated Darter (*Etheostoma olmstedi*). Cove stations 7 and 10.

At the river channel station (station 9), catches in 2016 were lower than the last two years (Figure 158), and lowest over the period of record. As in the inner cove, much of the variation at station 9 is directly attributable to the catch of White Perch. The fact that total catch in 2016 was lower than that in 2015 was not because of a lower amount of White Perch. Lower numbers of Alosines were mostly responsible for the difference.

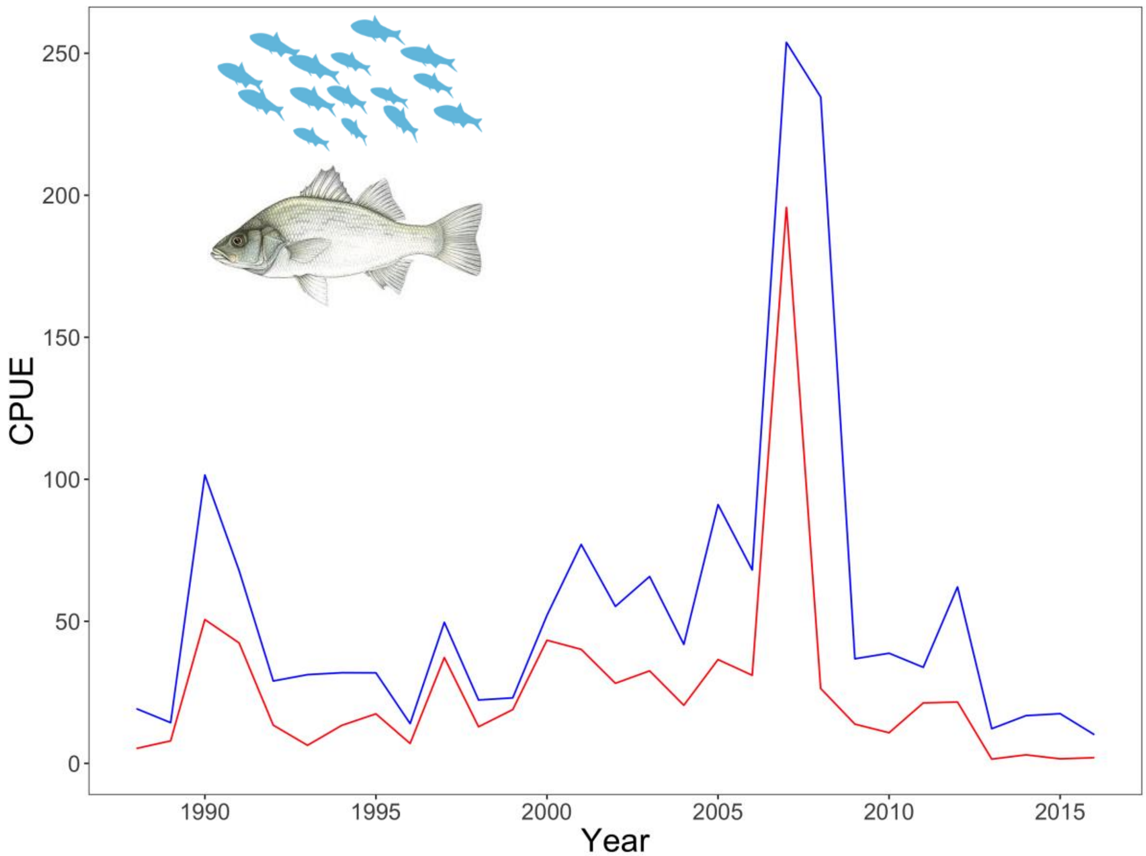


Figure 158. Trawls. Annual averages. River Station (9). Total catch (blue), White Perch (red).

Since 1988 when station 9 was incorporated as part of the survey, Bay Anchovy, Spottail Shiner, and American Eel have occurred sporadically at station 9 (Figure 159). We find high abundance of Bay Anchovy once every 5 years or so, with one very distinct peak in 2008. Overall, an increasing trend in Bay Anchovy abundances is observed (Figure 159). Spottail Shiner is found in low numbers every year at station 9, while American Eel has been rare since 1994.

Catch rates for native catfish species have been variable and low at station 9 since 2007 (Figure 160), with only a small peak from Channel catfish in 2011. No Brown Bullhead or Channel Catfish were observed in 2016. Long-term mean trends identify a decline in both Brown Bullhead and Channel Catfish (Figures 160). One species that warrants close attention is the invasive Blue Catfish, which was positively identified on the survey in 2001 and has been captured in high numbers relative to Channel Catfish and Brown Bullhead ever since (Figure 160). Since Blue Catfish occupy the same niche, but can grow to larger sizes, it generally outcompetes the native catfish population (Schloesser et al., 2011). Blue Catfish established itself in 2001 with relatively high numbers, but the trend has remained flat since then, and even may be somewhat declining (Figure 160). The system may have reached a new stable state that includes Blue Catfish in relative high numbers, and Channel Catfish and Brown Bullhead in low numbers. Continued monitoring in the growth of this population is warranted. Of note is that we are not capturing very large specimens with the otter trawl, and very large blue catfishes have been reported in this area.

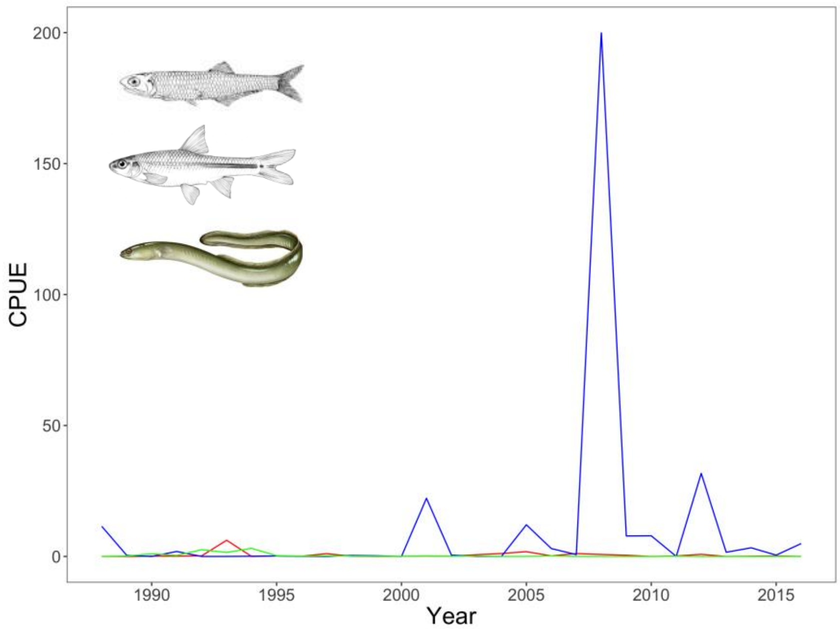


Figure 159. Trawls. Annual Averages. River Station (9). Bay Anchovy (Blue) Spottail Shiner (red) American eel (green).

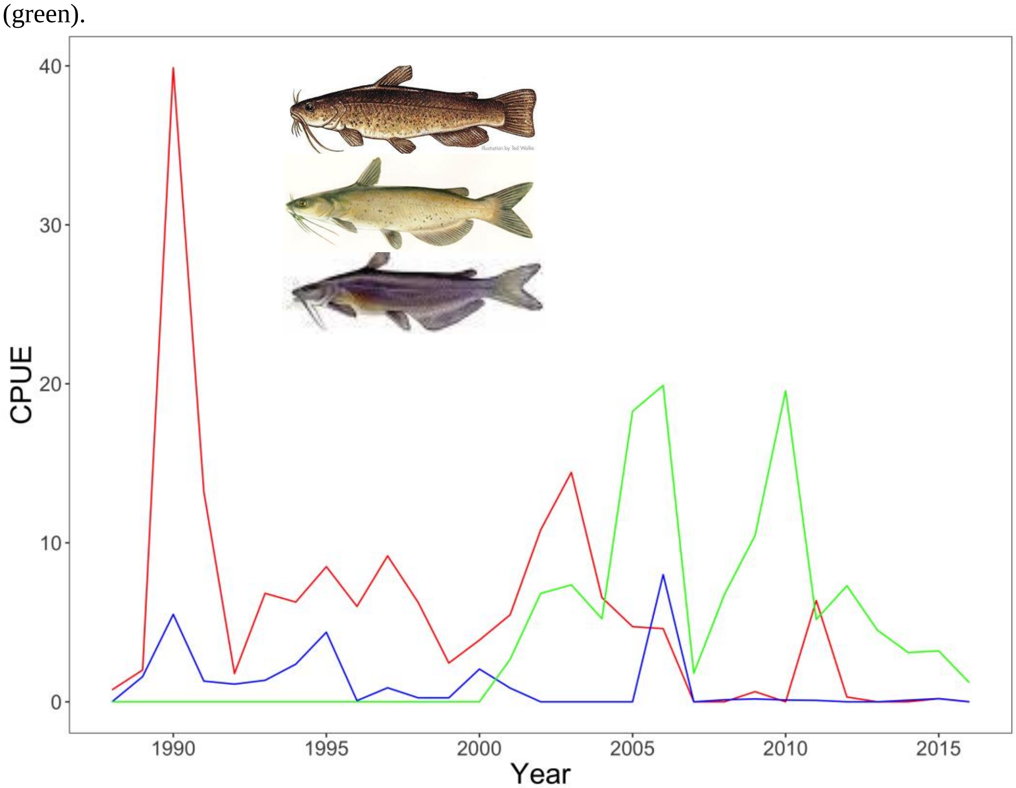


Figure 160. Trawls. Annual Averages. River Station (9). Brown Bullhead (blue), Channel Catfish (red), and Blue Catfish (green).

Station 9 represented low catch rates for the demersal species Tessellated Darter and Hogchoker (Figure 161). High catches have not occurred since 2004 (Figure 161) and neither of the two species was captured at station 9 in 2016. The mean annual trend seems to indicate a general decline in catch rates for each of these species in our Potomac mainstem site over the time-span of the survey (Figure 161).

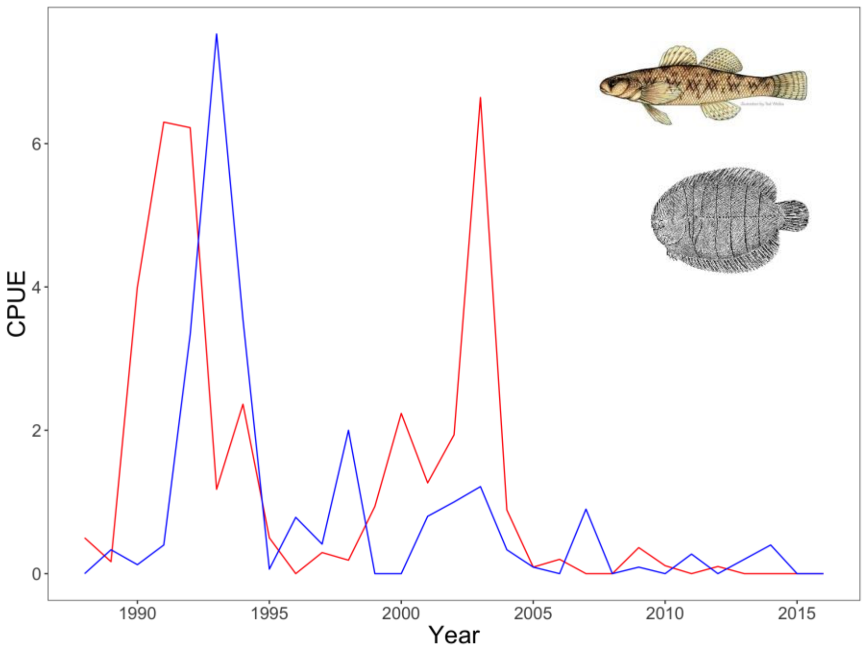


Figure 161. Trawls. Annual Averages. Tessellated Darter (blue) hogchoker (red). River Station (9).

Seines and fyke nets

Overall Patterns

Mean annual seine catch rates were generally higher than trawl catch rates. The long-term trend of seine catches shows a stable pattern of catches amidst inter-annual variability (Figures 162). The overall pattern shows a very slight increase in catches over the course of the survey (Figure 162). Of the three most abundant years high catches were due to a high abundance of Alosines those years: 1994 and 2004 were driven primarily by large catches of Alewife, whereas high catch rates in 1991 were a result of high catch rates of Blueback Herring (Table 23). Overall, Banded Killifish and White Perch have been the dominant species in seine samples throughout the survey. In 2016 the general trend of decreasing White Perch catches and increasing Banded Killifish catches over the period of record continues (Figure 163). The decrease in White Perch seen in seine catches is indication of the shifted ecosystem state to an SAV dominated system, since Banded Killifish prefers SAV habitat, while White Perch prefers open water. The decreasing trend in white Perch, and increasing trend in Banded Killifish, seems to be leveling out, and a new stable state in the relative contribution of these two species may have been reached. Subsequent years will determine whether this is indeed the case. The number of seine tows over the period of record is shown in Table 24. Fyke nets collected less specimens than the previous years, and collections were dominated by sunfishes. Like previous years, the relative contribution of other species in fyke nets is different than collected with trawl or seine nets, and mainly represents SAV-associated species such as Banded Killifish and several species of sunfishes.

Table 23. Mean Catch per Seine of Selected Adult and Juvenile Fishes at all Stations and all Months. 1985-2016.

Year	All Spp.	White Perch	Banded Killifish	Blueback Herring	Alewife	All Alosa Spp.	Spottail Shiner	Inland Silverside
2016	114.3	11.6	64.5	0.0	0.0	6.9	1.2	8.1
2015	171.2	33.1	76.1	0.5	0.4	17.1	5.2	4.7
2014	169.5	11.9	121.4	3.5	0.1	8.3	4.1	4.1
2013	117.3	8.3	92.4	0.1	0.2	2.1	0.4	0.7
2012	180.9	5.3	128.1	0.0	2.1	4.4	5.9	12.0
2011	137.1	31.0	76.3	0.0	0.6	1.3	2.4	1.5
2010	249.3	15.8	175.6	0.0	0.0	23.1	1.6	1.3
2009	186.5	18.7	67.4	0.3	0.2	1.5	3.6	6.9
2008	196.5	15.4	51.8	0.3	0.1	2.5	3.1	14.9
2007	130.4	15.0	40.6	6.7	2.2	17.6	3.4	2.3
2006	165.3	7.6	113.7	3.2	0.4	6.2	3.6	16.2
2005	230.4	37.8	139.9	1.3	6.7	9.0	10.7	6.6
2004	304.5	45.3	99.1	11.1	73.8	85.2	38.1	9.5
2003	100.6	7.5	42.9	2.3	2.8	7.5	7.3	4.8
2002	164.4	23.1	89.7	0.0	2.3	3.2	12.5	14.4
2001	134.0	30.2	54.6	0.0	4.9	5.6	14.3	7.6
2000	152.2	28.9	26.2	1.7	6.0	7.7	23.5	50.1
1999	108.1	18.3	19.0	14.4	0.4	14.8	12.3	25.0
1998	111.6	22.2	31.6	2.1	1.0	3.1	25.9	8.7
1997	107.5	14.1	37.0	19.5	1.6	21.1	5.0	16.1
1996	103.6	29.1	18.2	15.0	6.2	22.2	11.8	4.7
1995	88.8	26.1	16.3	2.1	2.8	5.0	5.8	12.5
1994	294.9	15.6	13.9	0.0	250.2	250.3	7.2	0.1
1993	73.6	13.4	26.1	3.2	1.3	4.5	8.5	9.1
1992	154.5	43.6	35.8	39.3	0.0	39.3	9.0	5.8
1991	215.1	31.7	44.1	71.8	0.2	71.9	18.8	6.5
1990	118.4	41.1	27.6	7.4	1.1	8.5	9.0	4.0
1989	130.8	39.9	25.8	1.8	0.5	2.3	8.1	1.9
1988	123.2	42.0	25.7	2.2	0.3	2.6	9.3	6.2
1987	108.9	36.7	31.9	0.0	0.0	0.0	8.0	11.6
1986	117.2	40.1	15.4	0.2	0.9	1.3	7.7	22.3
1985	122.0	37.4	11.9	0.0	0.1	0.2	13.2	30.0

Table 24. The number of seines in each month at Station 4, 4B, 6, and 11 in each year. 1985-2016.

Year	Station	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2016	4	0	0	0	1	2	1	0	0	0	0	0	0
2016	6	0	0	0	1	2	2	2	2	1	0	0	0
2016	11	0	0	0	1	2	2	2	2	1	0	0	0
2016	4B	0	0	0	1	2	2	2	2	1	0	0	0
2015	4	0	0	0	1	2	2	0	0	0	0	0	0
2015	6	0	0	0	1	2	2	2	2	1	0	0	0
2015	11	0	0	0	1	2	2	2	2	1	0	0	0
2015	4B	0	0	0	1	2	2	2	2	1	0	0	0
2014	4	0	0	0	1	2	2	2	1	0	0	0	0
2014	6	0	0	0	1	2	2	2	2	1	0	0	0
2014	11	0	0	0	1	2	2	2	2	1	0	0	0
2014	4B	0	0	0	1	2	2	2	2	1	0	0	0
2013	4	0	0	0	1	2	2	2	0	0	0	0	0
2013	6	0	0	0	1	2	2	2	2	1	0	0	0
2013	11	0	0	0	1	2	2	2	2	1	0	0	0
2013	4B	0	0	0	1	2	2	2	2	1	0	0	0
2012	4	0	0	0	1	2	2	1	0	0	0	0	0
2012	6	0	0	0	1	2	2	2	2	1	0	0	0
2012	11	0	0	0	1	2	2	2	2	1	0	0	0
2012	4B	0	0	0	1	2	2	2	2	1	0	0	0
2011	4	0	0	0	1	2	3	2	2	1	0	0	0
2011	6	0	0	0	1	2	3	2	2	0	1	0	0
2011	11	0	0	0	1	3	3	2	2	1	0	0	0
2011	4B	0	0	0	1	2	3	2	2	1	0	0	0
2010	4	0	0	0	1	1	2	2	2	1	0	0	0
2010	6	0	0	0	1	1	2	2	2	1	0	0	0
2010	11	0	0	0	1	1	2	2	2	1	0	0	0
2010	4B	0	0	0	1	1	2	2	2	1	0	0	0
2009	4	0	0	0	1	2	2	2	2	1	0	0	0
2009	6	0	0	0	1	2	2	2	2	1	0	0	0
2009	11	0	0	0	1	2	2	2	2	1	0	0	0
2009	4B	0	0	0	1	2	2	2	2	1	0	0	0
2008	4	0	0	0	1	2	2	2	2	1	0	0	0
2008	6	0	0	0	1	2	2	2	2	1	0	0	0
2008	11	0	0	0	1	2	2	2	2	1	0	0	0
2008	4B	0	0	0	1	2	2	2	2	1	0	0	0
2007	4	0	0	0	1	2	1	2	2	1	0	0	0
2007	6	0	0	0	1	2	1	2	2	1	0	0	0
2007	11	0	0	0	1	2	1	2	2	1	0	0	0
2007	4B	0	0	0	0	0	0	2	2	1	0	0	0
2006	4	0	0	0	0	0	0	1	2	0	0	0	0
2006	6	0	0	0	1	2	2	2	0	0	0	0	0

Year	Station	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
2006	11	0	0	0	1	2	2	2	2	1	0	0	0
2006	4B	0	0	0	1	2	1	0	0	1	0	0	0
2005	4	0	0	0	1	2	2	2	0	0	0	0	0
2005	6	0	0	0	1	2	2	1	0	0	0	0	0
2005	11	0	0	0	1	2	2	2	2	1	1	0	0
2004	4	0	0	0	1	1	2	1	0	0	0	0	0
2004	6	0	0	0	1	1	2	0	0	0	0	0	0
2004	11	0	0	0	1	1	2	2	2	1	0	0	0
2003	4	0	0	1	2	2	2	2	1	1	1	1	1
2003	6	0	0	1	1	2	2	2	1	1	1	1	1
2003	11	0	0	1	2	2	2	2	1	1	1	0	1
2002	4	0	0	1	2	2	2	2	2	2	1	1	1
2002	6	0	0	1	2	2	2	2	2	2	1	1	1
2002	11	0	0	1	2	2	2	2	2	2	1	1	0
2001	4	0	0	1	2	2	1	2	3	2	1	1	1
2001	6	0	0	1	2	2	1	2	3	2	0	1	1
2001	11	0	0	1	2	2	1	2	3	2	1	1	1
2000	4	0	0	1	2	2	3	2	2	2	1	1	1
2000	6	0	0	1	2	2	3	2	2	2	1	1	1
2000	11	0	0	1	2	2	3	2	2	2	1	1	0
1999	4	0	0	1	2	2	2	2	2	2	1	1	1
1999	6	0	0	1	2	2	2	2	2	2	1	1	1
1999	11	0	0	1	2	2	2	2	2	2	1	1	0
1998	4	0	0	1	2	2	2	2	2	2	1	1	1
1998	6	0	0	1	2	2	2	2	2	2	1	1	1
1998	11	0	0	1	2	2	2	2	2	2	1	1	0
1997	4	0	0	1	2	2	2	2	2	2	2	1	1
1997	6	0	0	1	2	2	2	2	2	2	2	1	1
1997	11	0	0	1	1	2	2	2	2	2	2	1	1
1996	4	0	0	1	2	2	2	2	1	2	1	1	1
1996	6	0	0	1	2	2	2	2	1	2	1	1	0
1996	11	0	0	1	2	2	2	2	1	2	1	1	1
1995	4	0	0	1	1	2	2	2	2	2	2	1	0
1995	6	0	0	1	2	2	2	2	2	2	2	1	0
1995	11	0	0	1	2	2	1	2	2	2	2	1	0
1994	4	0	0	0	0	1	1	0	0	1	1	0	0
1994	6	0	0	1	0	1	1	0	0	1	1	0	0
1994	11	0	0	1	0	1	1	0	0	1	1	0	0
1993	4	0	0	1	2	2	1	3	2	0	1	1	1
1993	6	0	0	1	1	2	1	3	2	0	1	1	1
1993	11	0	0	1	2	2	1	3	2	0	1	1	1
1992	4	0	0	1	1	1	1	1	1	1	1	1	0

Year	Station	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1992	6	0	0	1	1	1	1	1	1	1	1	1	0
1992	11	0	0	0	1	1	1	1	1	1	1	1	0
1991	4	0	0	1	1	1	1	1	1	1	1	1	0
1991	6	0	0	1	1	1	1	1	1	1	1	0	0
1991	11	0	0	1	1	1	1	1	1	1	1	1	0
1990	4	0	0	1	1	1	1	1	1	1	0	0	0
1990	6	0	0	1	1	1	1	1	1	1	0	0	0
1990	11	0	0	1	1	1	1	1	1	1	0	0	0
1989	4	0	0	1	1	1	1	1	1	1	1	1	0
1989	6	0	0	1	1	1	1	1	1	1	1	1	0
1989	11	0	0	1	1	1	1	1	1	1	1	1	0
1988	4	0	0	1	1	0	2	2	1	1	1	1	0
1988	6	0	0	1	1	1	2	2	2	1	1	1	0
1988	11	0	0	1	1	1	2	2	2	1	1	1	0
1987	4	0	0	1	1	0	1	1	0	0	1	1	0
1987	6	0	0	1	1	0	1	1	0	0	1	0	0
1987	11	0	0	1	1	0	1	1	0	0	1	1	0
1986	4	0	1	0	1	0	1	0	0	1	2	0	0
1986	6	1	1	0	1	1	1	0	0	2	1	0	0
1986	11	1	1	0	1	1	1	0	0	2	2	0	0
1985	4	0	0	0	0	0	0	0	1	1	1	2	0
1985	6	0	0	0	0	0	0	0	1	1	1	2	0
1985	11	0	0	0	0	0	0	0	1	1	1	2	0

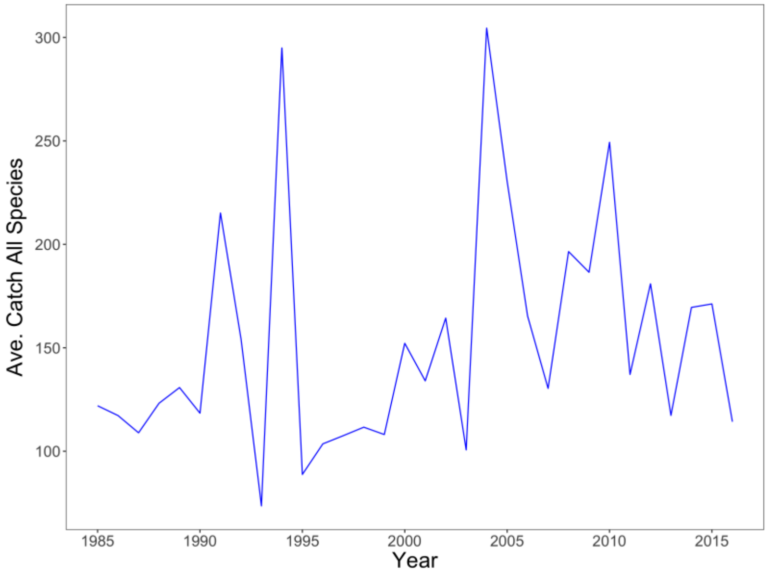


Figure 162. Seines. Annual Average over Stations 4, 4A, 6, and 11. All Species. 1985-2016.

Over the course of the survey mean annual seine catch rates of White Perch have exhibited a gradual decline (Figures 163). An important factor is the pronounced increase in SAV, which until 2012 was not effectively sampled and could potentially represent a significant alternative habitat for White Perch. In 2012, fyke nets were added to the sampling gear near Station 4 (seine station where SAV interferes halfway during the sampling season) and Station 10 (trawl station where SAV interferes with sampling halfway during the sampling season). For the first three years of fyke net collections (2012-2014), White Perch was not among the dominant species in fyke nets. However, in 2015 White Perch was the second most dominant species in fyke net collections, and was present again in 2016, indicating it is present within the SAV beds as well. Fyke nets did efficiently sample the SAV beds, and were dominated by SAV-associated species like Banded Killifish and sunfishes. Additional abundant species in the fyke nets in 2016 were Inland Silverside and Largemouth Bass. The state shift of the ecosystem to a SAV dominated system has resulted in a shift in the nekton community from open-water species to SAV-associated species.

The relative success of Banded Killifish is coincidentally (rather than functionally related) to declines in White Perch as these species show very little overlap in ecological and life history characteristics. Instead, as mentioned above, prominent increases in mean catch rates of Banded Killifish are associated with development of SAV in the cove since 2000. The SAV provides refuge for Banded Killifish adults and juveniles and may enhance feeding opportunities with epifaunal prey items. Essentially, the habitat of White Perch in Gunston Cove has decreased, while the habitat of Banded Killifish has increased. However, White Perch does reside in SAV covered areas as well, just in lower numbers.

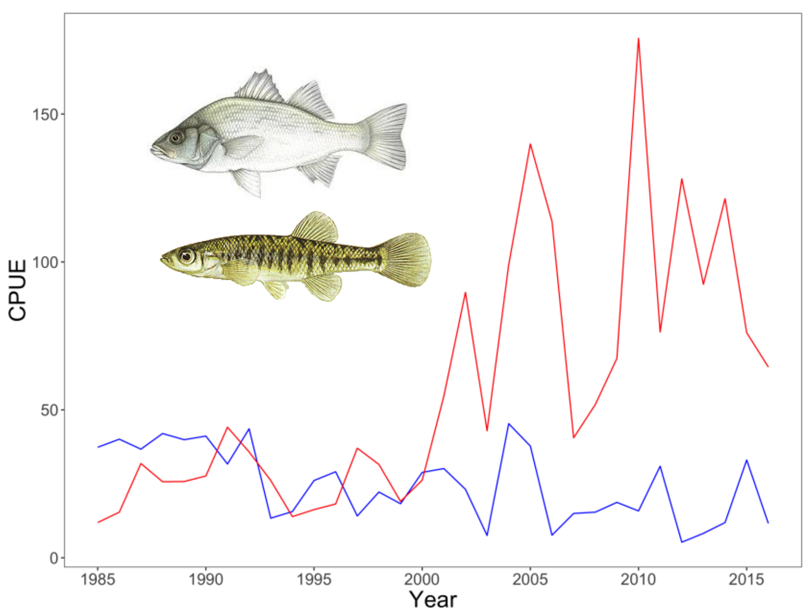


Figure 163. Seines. Annual Average Stations 4, 4A, 6, and 11. White Perch (blue) and Banded Killifish (red). 1985-2016.

Long-term trends in mean annual catch rates for the two dominant species in seine hauls have exhibited a negative association (r=-0.427) over the course of the survey. White Perch mean catches have declined steadily since the beginning of the survey, while Banded Killifish numbers have increased since the start of the survey, and experienced a prominent increase since 1999 (Figure 163). Mean catches from both species in 2016 may indicate a stabilization of these diverging trends.

Mean annual catch rates for river herring (Alewife and Blueback Herring) have exhibited sporadic peaks related to the capture of a large schools of fish (exceeding 200 for Alewife and approaching 100 individuals for Blueback Herring) in single hauls (Figure 164). Typically, less than 10 of either species were captured in a single sample. Though very variable, long-term trends indicate a decline in overall catches of Alewife and Blueback Herring. These species are both listed as species of concern and have experienced declines throughout the Chesapeake Bay watershed. The moratorium on river herring since January 2012 has been put in place as an aid in the recovery. If successful, the moratorium (on fishing) may results in an increase in river herring over time in future years. We added the category 'all *Alosa sp*.' to figure 164 in 2016 because a large portion of the Alosines cannot be identified to the species level. That revealed that Alosine abundances have been slightly higher since 2005 then just based on Alewife and Blueback Herring findings. For example relatively high peaks in Alosines have been found in 2007, 2010, and 2015. Abundances are not sufficiently high that the stocks can be considered recovered. Continued monitoring will be key in determining the success of the moratorium. The high numbers of spawning adult river herring in 2015 in Pohick Creek, as described in the 2015 Anadromous Report, could signal the start of the recovery of these species. The abundances in 2016 were lower than 2015 again, but still relatively high compared to the average of the period of record (see Anadromous Report).

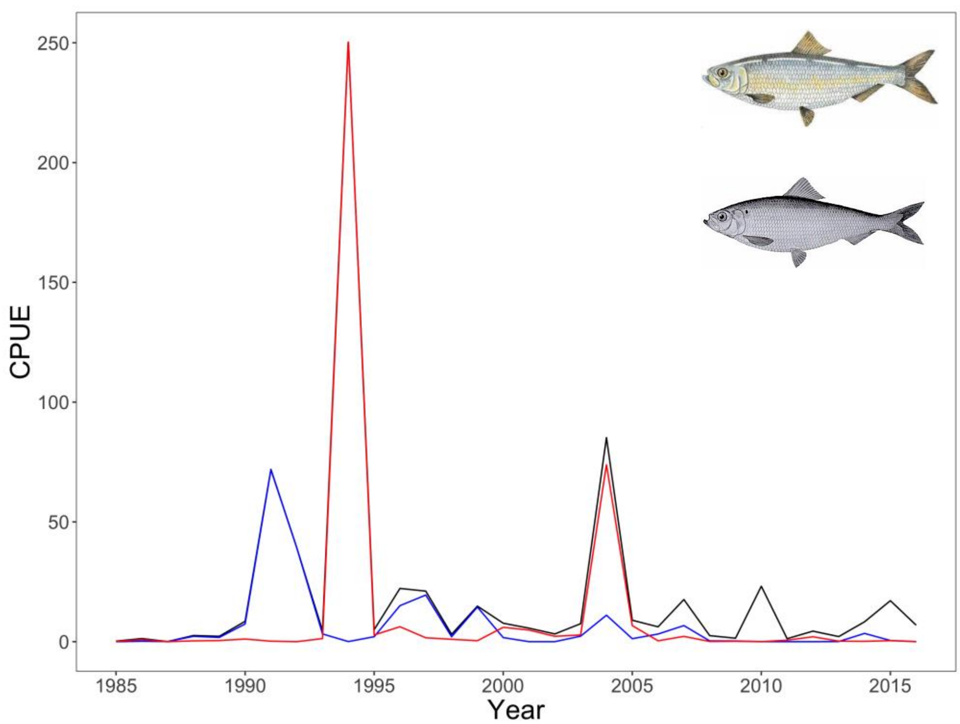


Figure 164. Seines. Annual Average over 4, 4A, 6, and 11 Stations. Blueback Herring (blue), Alewife (red), and all *Alosa* sp (black; Blueback Herring, Alewife, Hickory Shad, American Shad, and unidentified Herring and Shad species). 1985-2016.

Owing to their affinity for marginal and littoral zone habitats, Spottail Shiner and Inland Silverside were consistently captured at moderate abundances throughout the course of the survey (Figure 165). Although a few high abundance years (1985, 1991, 1998, 2000, and 2004) have occurred, a general declining trend in catches since 2000 was present (Figure 165). While the fyke nets did capture a high proportion of Spottail Shiner in 2014, only one was collected in 2016. Like 2015, Inland Silverside had a higher abundance in the fyke nets. With the variable record within the SAV-beds as represented by the fyke net catches, similar to the record of trawl catches, these species do not seem to have particularly concentrated in SAV beds, but rather have remained moderately abundant throughout the Cove and the survey when all gear is considered.

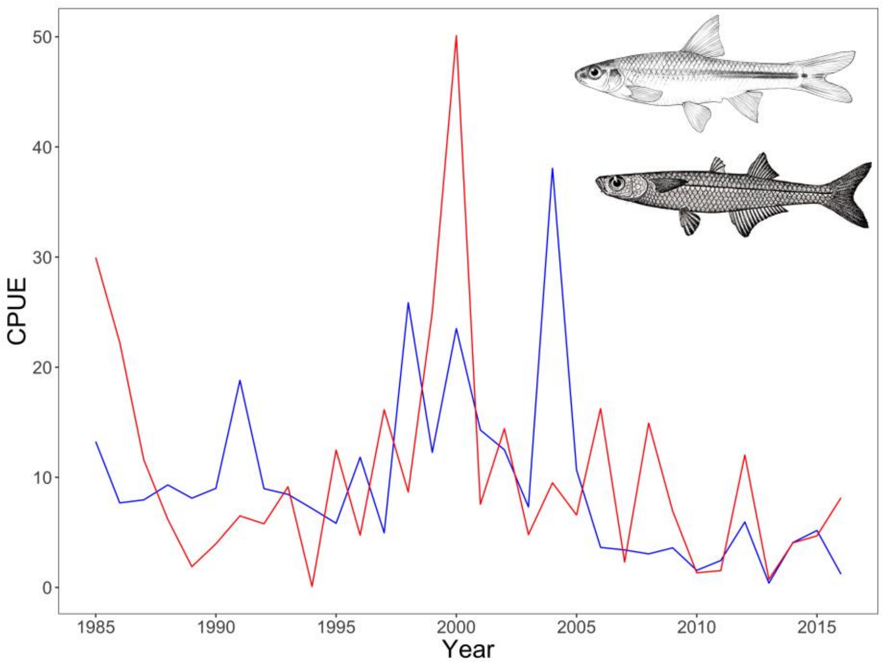


Figure 165. Seines. Annual Average over 4, 4A, 6, and 11 Stations. Spottail Shiner (blue) and Inland Silverside (red). 1985-2016.

Long-term Species Composition Changes

The species composition and community structure is changing throughout the time of the survey as indicated by trawl and seine catches. The expansion of SAV beds in the inner cove seems to be driving some of these changes. The main trend related to increasing SAV beds is a decline in White Perch and an increase in Banded Killifish. A detailed multivariate analysis of the community structure shifts in the Gunston Cove fish community sicne the start of the Gunston Cove survey has recently been published (De Mutsert et al. 2017). Another community shift can be seen in the catfishes. Since the introduction of the invasive Blue Catfish in Gunston Cove in 2001, Blue Catfish has become prevalent in the trawl catches, while the abundances of other catfishes (Brown Bullhead, Channel Catfish, White Catfish) have been declining. The trend in Blue Catfish abundance is currently not increasing, and seems to have reached a plateau. Potentially, a new stable state has been achieved with high Blue Catfish abundances and low abundances of other catfishes. We do collected some Brown Bullhead specimens in the fyke nets. More fyke net collections are needed to determine if there is a spatial shift of Brown Bullhead towards SAV beds, which would not be unusual for this species that prefers vegetated habitat. Another interesting community change is an increase in collections of Striped Bass. We only find Striped Bass in low numbers, but because of its high commercial and recreational value, it is worth mentioning. While Striped Bass is thought to occur in more saline waters, this semi-anadromous species does come up to tidal freshwater areas to spawn, and we find juvenile Striped Bass in our seine and trawl collections. Other observed long-term changes are the decline in Alewife and Blueback Herring. These declines are in concurrence with declines observed coast-wide, and do not have a local cause. It is a combination of declining suitable spawning habitat and overfishing (either targeted fishing that ended in 2012, or as bycatch of the menhaden fishery). Relative high abundances of juvenile Alosines in the trawl and seine samples in 2015 could be an indication of the start of a recovery since a moratorium on fishing was imposed in 2012. However, the numbers were not as high in 2016. The large cohort of spawning adults of Blueback Herring and Alewife in Accotink Creek and Pohick Creek, as reported in the 2015 Anadromous Report, could be the start of increasing numbers in years to come.

Summary

In 2016 ichthyoplankton was dominated by clupeids, most of which were Gizzard Shad and Alewife, and to a lesser extent, Blueback Herring, American Shad, and Hickory Shad. White Perch was a dominant species as well, with the same relative contribution to the total ichthyoplankton community as Gizzard shad. Striped Bass and Inland Silverside was found in relatively high densities as well. *Morone* species (White Perch and Striped Bass) were mostly found in the Potomac mainstem, confirming their affinity for open water. Other taxa were found in very low densities similar to the previous year. The highest density of fish larvae occurred in mid-May, which was driven by a high density of Clupeid larvae. Most clupeids are spawn from March –May, and are spawn closer to, or even further upstream from, the head of the tide. These larvae then drift down, and remain in tidal tributaries such as Gunston Cove until they are juvenile. They then usually remain several months as juveniles as well, and use Gunston Cove as a nursery.

The trawl, seine and fyke net collections continue to provide valuable information about long-term trends in the fish assemblage of Gunston Cove. The development of extensive beds of SAV over the past decade is providing more favorable conditions for Banded Killifish and several species of sunfish (Bluegill, Pumpkinseed, Redear Sunfish, Redbreast Sunfish, Bluespotted Sunfish, and Green Sunfish) among other species. Indeed, seine and trawl sampling has indicated a relative increase in some of these SAV-associated species. The abundance of some species such as White Perch are showing a decline (while relative abundance of White Perch in this area compared to other species than Banded Killifish remains high). This is likely due to a shift in nekton community structure as a result of the state shift of Gunston Cove to a SAV-dominated system. The shift in fish community structure was clearly linked to the shift in SAV cover with a community structure analysis (De Mutsert et al. 2017).

The SAV expansion has called for an addition to the sampling gear used in the survey, since both seines and trawls cannot be deployed where SAV beds are very dense. While drop ring sampling has been successfully used in Gunston Cove in previous years (Krauss and Jones, 2011), this was done in an additional study and is too labor-intensive to add to our semi-monthly sampling routine. In 2012, fyke nets were deployed to sample the SAV beds. The fyke nets proved to be an effective tool to sample the fish community within the vegetation. While fyke-nets do not provide a quantitative assessment of the density of species, it effectively provided a qualitative assessment of the species that reside in the SAV beds. The fyke nets collected mostly several species of sunfish and Banded Killifish, which are indeed species know to be associated with SAV.

Juvenile anadromous species continue to be an important component of the fish assemblage. We have seen declines in river herring since the mid 1990s, which is in concordance with other surveys around the Potomac and Chesapeake watersheds. In January 2012, a moratorium on river herring was put in effect to alleviate fishing pressure in an effort to help river herring stocks rebound. There were relatively high numbers of juvenile Blueback Herring, Alewife and other Alosines in trawls and seines in 2015. These abundances were lower again in 2016, but the successful spawning cohort of 2015 (reported in more detail in the 2015 Anadromous report) may be able to sustain the Alosine populations at higher levels than before 2015. The continued monitoring of Gunston Cove since the complete closure of this fishery will help determine if the moratorium results in a recovery of Blueback Herring and Alewife.

G. Submersed Aquatic Vegetation (SAV) Trends: 1994-2016

A comprehensive set of annual surveys of submersed aquatic vegetation in the Gunston Cove area is available on the web at http://www.vims.edu/bio/sav/. This is part of an ongoing effort to document the status and trends of SAV as a measure of Bay recovery. Maps of SAV coverage in the Gunston Cove area are available on the web site for the years 1994-2016 except for 2001 and 2011. Data was not available in 2011 due to severe weather and poor imagery issues. A plot of SAV vs. Chlorophyll *a* and Secchi disk depth revealed that chlorophyll remained at near record low levels in 2016 and that Secchi depth was near its all-time high (Figure 166). These values reflect the sustained partial recovery of Gunston Cove from eutrophication.

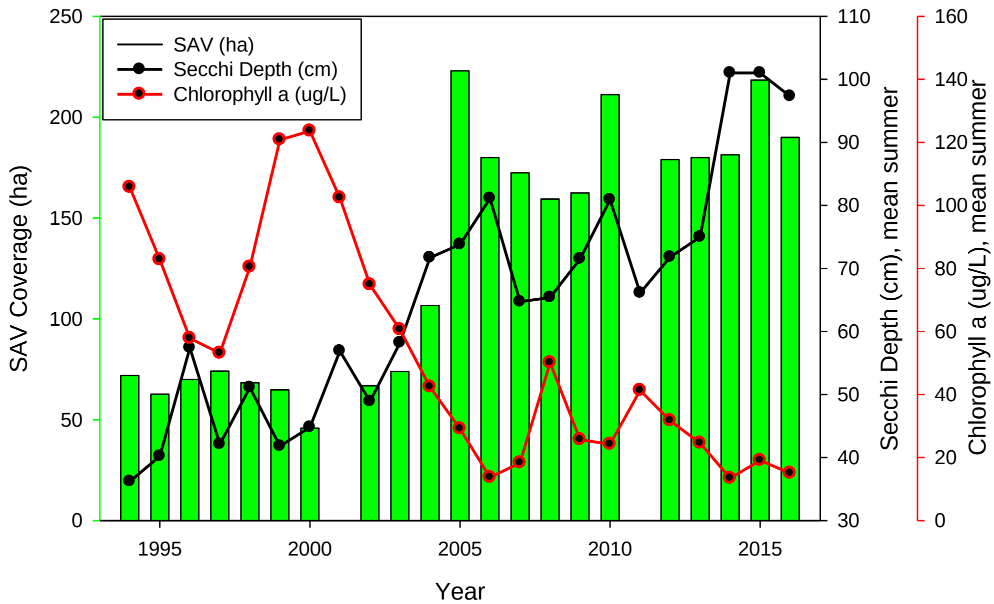


Figure 166. Gunston Cove SAV Coverage. Graphed with average summer (June-September) Depth-integrated Chlorophyll a (µg/L) and Secchi Depth (cm) measured at Station 7 in Gunston Cove. (2016 values are estimates).

H. Benthic macroinvertebrates

Benthic invertebrates have been monitored in a consistent fashion since 2009. Those data are assembled below (Table 25) and trends are generally consistent among years. The composition of the benthic macroinvertebrate community at these two sites seems to reflect mainly the texture of bottom substrates. In the cove at Station 7, the bottom sediments are fine and organic with anoxia just below the surface. These conditions favor chironomids and oligochaetes and are not very supportive of the other taxa found in the river. Interestingly, as SAV has become more established gastropods are becoming more abundant an chironmids (midge larvae) are declining. In the river sediments are coarser and are comprised of a mixture of bivalve shells (mainly *Corbicula*) and sand/silt. This type of substrate is supportive of a wider array of species. Oligochaetes are generally the most abundant taxon at both stations. In 2012 and 2013 chironomids were the most abundant taxa, but they declined strongly in 2014 and 2015. Amphipods are have generally occurred sporadically at low levels in the cove, but in substantial numbers in the river. In 2014 amphipods were the most abundant organism in the river, but returned to second place in 2015 and 2016. Isopods have been commonly found in the river since 2010 and sporadically in the cove; they reached their highest densities in both sites in 2016. Turbellaria (flatworms) and Hirundinea (leeches) are found in low numbers sporadically at both sites and were present in several river samples in 2014. The consistent finding of even small numbers of taxa other than chironomids and oligochaetes in the cove is encouraging and could be the result of improved water quality conditions in the cove.

Table 25. Benthic macroinvertebrates: annual averages (#/petite ponar)

	Station 7 (#/petite ponar)				Station 9 (#/petite ponar)			
Taxon	2009-13 Avg	2014	2015	2016	2009-13 Avg	2014	2015	2016
Oligochaeta	46.2	26.1	45.1	17.2	69.6	9.7	98.2	39.1
Amphipoda	1.6	1.7	4.4	3.4	23.5	32.6	33.9	11.9
Chironomidae	39.5	2.3	3.7	11.6	1.3	0.4	5.3	1.1
Corbicula	0.1	--	0.9	0.8	8.4	--	3.9	0.9
Gastropoda	0.4	--	11.9	0.8	5.2	--	12.4	1.2
Isopoda	0.02	0.1	0.7	1.2	1.9	1.7	6.4	6.8
Turbellaria	0.1	0.0	0.7	0.5	0.7	2.9	6.3	1.1
Hirundinea	0.4	0.2	0.6	0.1	0.2	1.2	0.1	0.0
Total	88.7	30.4*	68.2	36.4	111.1	48.5*	217.1	66.3

For 2009-10, n=8 per station; for 2011-12, n=6 per station; for 2013, 2015 and 2016, n=15 per station; for 2014, n=14 per station.

*Note that molluscs were not enumerated in 2014 due to processing error.

LITERATURE CITED

Bigelow, H.B. and W.C.Schroeder. 1953. Fishes of the Gulf of Maine. Fishery bulletin No. 74, Vol. 53. U.S. Government Printing Office. Washinton, D.C. 577 pp.

Carter, V., P.T. Gammon, and N.C. Bartow. 1983. Submersed Aquatic Plants of the Tidal Potomac River. Geological Survey Bulletin 1543. U.S. Geological Survey. 63 pp.

Chesapeake Bay Program. 2006 Ambient water quality criteria for dissolved oxygen, water clarity, and chlorophyll *a* for the Chesapeake Bay and its tidal tributaries. 2006 Addendum. Downloaded from Bay Program website 10/13/2006.

Cummings, H.S., W.C. Purdy, and H.P. Ritter. 1916. Investigations of the pollution and sanitary conditions of the Potomac watershed. Treasury Department, U.S. Public Health Service Hygienic Laboratory Bulletin 104. 231 pp.

Dahlberg, M.D. 1975. Guide to coastal fishes of Georgia and nearby states. University of Georgia Press. Athens, GA 187 pp.

De Mutsert, K., Sills, A., Schlick, C.J.C., and R.C. Jones. 2017. Successes of restoration and its effects on the fish community in a freshwater tidal embayment of the Potomac River, USA. Water 9(6), 421; doi:10.3390/w9060421

Douglass, R.R. 1999. A Field Guide to Atlantic Coast Fishes: North America (Peterson Field Guides). Houghton Mifflin Harcourt, Boston. 368 pp.

Eddy, S. and J.C. Underhill. 1978. How to know the freshwater fishes. 3rd Ed. W.C. Brown Co. Dubuque, IA. 215 pp.

Hildebrand and Schroeder. 1928. Fishes of the Chesapeake Bay. U.S. Bureau of Fisheries Bulletin 53, Part 1. Reprinted 1972. T.F.H. Publishing, Inc. Neptune, NJ. 388 pp.

Hogue, J.J, Jr., R.Wallus, and L.K. Kay. 1976. Preliminary guide to the identification of larval fishes in the Tennessee River. Technical Note B19. Tennessee Valley Authority. Knoxville, TN.

Islam, S. 2001. Seasonal dynamics of micro-, nanno-, and picoplankton in the tidal freshwater Potomac River in and around Gunston Cove. Ph.Dissertation. George Mason University. 127 pp.

Jenkins, R.E. and N.M. Burkhead. 1994. The freshwater fishes of Virginia. American Fisheries Society. Washington, DC. 1080 pp.

Jones, P.W., F.D. Martin, and J.D. Hardy, Jr. 1978. Development of fishes of the Mid-Atlantic bight. Volumes I-VI. Fish and Wildlife Service, U.S. Department of the Interior. FWS/OBS-78/12.

Kelso, D.W., R.C. Jones, and P.L. deFur. 1985. An ecological study of Gunston Cove - 1984-85. 206 pp.

Kraus, R.T. and R.C. Jones. 2011. Fish abundances in shoreline habitats and submerged aquatic vegetation in a tidal freshwater embayment of the Potomac River. Environmental Monitoring and Assessment. Online: DOI 10.1007/s10661-011-2192-6.

Lippson, A.J. and R.L. Moran. 1974. Manual for identification of early development stages of fishes of the Potomac River estuary. Power Plant Siting Program, Maryland Department of Natural Resources. PPSP-MP-13.

Loos, J.J., W.S. Woolcott, and N.R. Foster. 1972. An ecologist's guide to the minnows of the freshwater drainage systems of the Chesapeake Bay area. Association of Southeastern Biologists Bulletin 19: 126-138.

Lund, J.W.G., C. Kipling, and E.C. LeCren. 1958. The inverted microscope method of estimation algal numbers and the statistical basis of estimations by counting. Hydrobiologia 11: 143-170.

Mansueti, A.J. and J.D. Hardy, Jr. 1967. Development of fishes of the Chesapeake Bay region: an atlas of egg, larvae and juvenile stages: Part 1. Natural Resources Institute. University of Maryland. 202 pp.

Merritt, R.W. and K.W. Cummins. 1984. An introduction to the aquatic insects of North America. 2nd edition. Kendall/Hunt Publishing Co., Dubuque, IA. 722 pp.

Page, L.M., and B.M. Burr. 1998. A Field Guide to Freshwater Fishes: North America North of Mexico (Peterson Field Guides). Houghton Mifflin Harcourt, Boston. 448 pp.

Pennack, R.W. 1978. Fresh-water invertebrates of the United States. 2nd ed. Wiley-Interscience. New York, NY.

Schloesser, R.W., M.C. Fabrizio, R.J. Latour, G.C. Garman, G.C., B. Greenlee, M. Groves and J. Gartland. 2011. Ecological role of blue catfish in Chesapeake Bay communities and implications for management. American Fisheries Society Symposium 77:369-382.

Scott, W.B. and E.J. Crossman. 1973. Freshwater fishes of Canada. Bulletin 184. Fisheries Research Board of Canada. Ottawa, Canada. 966 pp.

Standard Methods for the Examination of Water and Wastewater. 1980. American Public Health Association, American Waterworks Association, Water Pollution Control Federation. 15th ed. 1134 pp.

Thorp, J.H. and A.P. Covich, eds. 1991. Ecology and classification of North American Freshwater Invertebrates. Academic Press. San Diego, CA. 911 pp.

Wetzel, R.G. 1983. Limnology. 2nd ed. Saunders. 767 pp.

Wetzel, R.G. and G.E. Likens. 1991. Limnological analyses. 2nd ed. Springer-Verlag. 391 pp.

Anadromous Fish Survey - 2016

Final Report
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Background

The commercially valuable anadromous fishes in the herring family (Clupeidae) live as adults in the coastal ocean, but return to freshwater creeks and rivers to spawn. In the mid-Atlantic region, four species are present: American Shad, Blueback Herring, Alewife, and Hickory Shad.

The American Shad grows to be the largest and spawns in the shallow flats along the Potomac River channel. In the 1700s and early 1800s, incredibly large numbers of American Shad were caught each spring as they came up the river to spawn. The records from 1814-1824 of just one fishery located at Chapman's Landing opposite Mason Neck, Virginia indicate that the annual catch varied from 27,939 to 180,755 American Shad (Massmann 1961). By 1982, the numbers caught in the entire river had dwindled so much that a moratorium was placed on both commercial and sport harvest of the species. In 1995, the Interstate Commission on the Potomac River Basin began a process of capturing ripe American Shad in gill nets off Dogue Creek and Fort Belvoir, stripping eggs from the females, and fertilizing the eggs with milt from males. The resulting young were raised in hatcheries for several days and then released, as fry, in the river below Great Falls (Cummins 2005). Through the 2002 season, over 15.8 million fry were released into the river, and by 2003 - the year after the restoration program ended - the population was judged strong enough to support a limited commercial fishery as bycatch in gill net fisheries. Moreover, a replacement stocking program continues (Jim Cummins, pers. comm.). The Virginia Department of Game and Inland Fisheries has also released some of the larvae at the boat ramp in Pohick Bay Regional Park in Gunston Cove (Mike Odom, USFWS; pers. comm.).

Prior to the 1900s, spawning occurred in the river as high as Great Falls (Smith and Bean 1899). In recent years spawning has occurred mostly downriver between Piscataway Creek and Mason Neck (Lippson et al. 1979). We do not normally catch individuals of this species as adults, juveniles, or larvae. The adults are not caught because our trawls mostly sample fishes that stay near the bottom of the water column, and the American Shad remain in the river where the water column is deeper. The juveniles mostly remain in the channel also, but sporadically some juvenile American Shad are captured at our seine stations. Hickory Shad has similar spawning habitats and co-occurs with American Shad, but is far less common than American Shad or river herring, and less is known about its life history. Coincident with the appearance of juvenile American Shad at our seine stations, we have also observed small numbers of juvenile Hickory Shad in recent years. Since 2010, we have been catching Hickory Shad adults in Pohick Creek and Accotink Creek.

The Alewife and Blueback Herring, collectively called river herring, are commercially valuable, although typically less valuable than American Shad. In past centuries, their numbers were apparently even greater than those of the American Shad. Massmann (1961) reported that from 1814 to 1824, the annual catch at Chapman's Landing ranged from 343,341 to 1,068,932 fish. The Alewife spawns in tributary creeks of the Potomac River and travels farther into these creeks than do the other species. The Blueback Herring also enters creeks to spawn, but may also utilize downstream tidal embayments to spawn.

River herring were listed in 2006 by NOAA as species of concern due to widespread declining population indices. Population indices of river herring in the Potomac are available from seine surveys of juveniles conducted by MD-DNR. Juvenile catch rate indices are highly variable but have been lower in the most recent decade for both species (Blueback Herring mean: 1998-2008=0.77 vs. 1959-1997=1.57; Alewife mean: 1998-2008=0.35 vs. 1959-1997=0.55). Since declines continued, a moratorium was established in January 2012, restricting all catches of Alewife and Blueback Herring (4VAC 20-1260-20). Causes of river herring decline are likely a combination of long-term spawning habitat degradation and high mortalities as a result of bycatch in the menhaden fishery. The establishment of a moratorium indicates that declines are widespread, and regular fishing regulations have not been sufficient to rebuild the stock. Using a moratorium to rebuild the stock is also an indication that the cause of the decline is largely unknown. Our monitoring of the river herring spawning population and density of larvae will aid in determining whether the moratorium is halting the decline in river herring abundance.

Another set of economically valuable fishes are the semi-anadromous White Perch and Striped Bass, which are sought after by both the commercial fishery and the sport-fishery. Both spawn in the Potomac River. Striped Bass spawn primarily in the river channel between Mason Neck and Maryland Point, while White Perch spawn primarily further upriver, from Mason Neck to Alexandria, and also in the adjacent tidal embayments (Lippson et al. 1979). Although spawning is concentrated in a relatively small region of the river, offspring produced there spread out to occupy habitats throughout the estuary. These juveniles generally spend the first few years of life in the estuary and may adopt a seasonal migratory pattern when mature. While most Striped Bass adults are migratory (spending non-reproductive periods in coastal seas), recent work indicates that a significant (albeit small) proportion of adults are resident in the estuaries.

Two other herring family species are semi-anadromous and spawn in the area of Gunston Cove. These are Gizzard Shad (*Dorosoma cepedianum*) and Threadfin Shad (*Dorosoma petenense*). Both are very similar morphologically and ecologically, but in our collections, Threadfin Shad are found downriver of Mason Neck, and Gizzard Shad are found upriver of Mason Neck. Neither is commercially valuable, but both are important food sources of larger predatory fishes.

For several years, we have focused a monitoring program on the spawning of these species in Pohick Creek, Accotink Creek, and, less regularly, Dogue Creek. We have sampled for adult individuals each spring since 1988 and for eggs and larvae since 1992. After 16 years of using block nets to capture adults, we shifted in the spring of 2004 to visual observations and seine, dip-net, and cast-net collections. This change in procedures was done to allow more frequent monitoring of spawning activity and to try to determine the length of time the spawning continued. We had to drop Accotink Creek from our sampling in 2005, 2006, and 2007 because of security-related access controls at Fort Belvoir. Fortunately, access to historical sampling locations from Fort Belvoir was regained in 2008. The block net methodology was taken up again in 2008 and has been continued weekly from mid-March to mid-May each year since then. The creeks continuously sampled with this methodology during this period are Pohick Creek and Accotink Creek. Results from our 2016 sampling are presented below. Since the 2015 report, we have included a summary table of the adult abundances from 2008 to present, which shows the changes observed since the period of record that the same sampling methods were used.

Introduction

Since 1988, George Mason University researchers have surveyed spawning river herring in Pohick Creek and adjacent tributaries of the Potomac River. The results have provided information on the annual occurrence and seasonal timing of spawning runs for Alewife (*Alosa pseudoharengus*) and Blueback Herring (*A. aestivalis*), but inferences on abundance have been limited for several reasons. The amount of effort to sample spawners has varied greatly between years and the methods have changed such that it is difficult to standardize the numbers captured or observed in order to understand annual fluctuations in abundance. River discharge was also not measured during the previous ichthyoplankton sampling. To maintain coherence with historical efforts while increasing the value of the data from surveys of Pohick and Accotink Creeks, we developed a modified protocol in 2008 with two main objectives: 1) quantify the magnitude of outdrifting larvae and coincident creek discharge rate in order to calculate total larval production; 2) quantify seasonal spawning run timing, size distribution and sex ratio of adult river herring using block nets (a putatively non-selective gear used throughout the majority of the survey). These modifications were accomplished with little additional cost and provided results that are more comparable to assessments in other parts of the range of these species. We have continued this sampling protocol in 2016 in Pohick Creek and Accotink Creek.

Methods

We conducted weekly sampling trips from March 15th to May 19th in 2016. Sampling locations in each creek were located near the limit of tidal influence and as close as possible to historical locations. The sampling location in Accotink creek was moved downstream a bit in 2014, which effectively moved the block net to an area before Accotink creek splits into two branches, which reduces the number of anadromous fishes that could escape through an unsampled branch of the creek. In Pohick Creek the block net remained in the same location. On one day each week, we sampled ichthyoplankton by holding two conical plankton nets with a mouth diameter of 0.25 m and a square mesh size of 0.333 mm in the stream current for 20 minutes. A mechanical flow meter designed for low velocity measurements was suspended in the net opening and provided estimates of water volume filtered by the net. The number of rotations of the flow meter attached to the net opening was multiplied with a factor of 0.0049 to gain volume filtered (m^3). Larval density ($\#/10m^3$) per species was calculated using the following formula:

Larval density ($\#/10m^3$) = 10N/(0.0049*(flow meter start reading-flow meter end reading))

Where N is the count of the larvae of one species in one sample.

We collected 2 ichthyoplankton samples per week in each creek, and these were spaced out evenly along the stream cross-section. Coincident with plankton samples, we calculated stream discharge rate from measurements of stream cross-section area and current velocity using the following equation:

Depth (m) x Width (m) x Velocity (m/s) = Discharge (m^3/s)

Velocity was measured using a handheld digital flow meter that measures flow in cm/s, which had to be converted to m/s to calculate discharge. Both depth and current velocity were measured at 12 to 20 locations along the cross-section.

The ichthyoplankton samples were preserved in 70% ethanol and transported to the GMU laboratory for identification and enumeration of fish larvae. Identification of larvae was accomplished with multiple taxonomic resources: primarily Lippson & Moran (1974), Jones et al. (1978), and Walsh et al. (2005). River herring (both species) have demersal eggs (tend to sink to the bottom) that are frequently adhesive. As this situation presents a significant bias, we made no attempts to quantify egg abundance in the samples. We were able to estimate total larval production (P) during the period of sampling by multiplying the larval density (m^{-3}) with total discharge (m^3) (Table 1).

The two river herring species (Blueback Herring and Alewife) are remarkably similar during both larval and adult stages, and distinguishing larvae can be extraordinarily time consuming. Our identification skills have improved over the time of the survey, and we do now distinguish Alewife from Blueback Herring in the larval stage as well as the adult stage. With the improved identification skills, we discovered that blue back herring sightings are common enough in our samples that they should be reported in this anadromous report, rather than Gizzard Shad, which is not an anadromous species. From the 2014 report on, the focus of this report is on the two true river herring species, Alewife and Blueback Herring, while presence of other clupeids (herring and shad species) such as Gizzard Shad will still be reported, but not analyzed to the detail of river herring.

The larval stages of two *Dorosoma* species are also extremely difficult to distinguish. However, only Gizzard Shad comes this far upstream, while Threadfin Shad has not been found higher up in the Potomac watershed than Mason Neck. Due to the absence of juveniles in seine and trawl samples from the adjacent Gunston Cove and adjacent Potomac River, we disregarded the possibility that Threadfin Shad were present in our ichthyoplankton samples.

The block net was deployed once each week in the morning and retrieved the following morning (see Figure 1). All fish in the block net were identified, enumerated, and measured. Fish which were ripe enough to easily express eggs or sperm/semen/milt were noted in the field book and in the excel spreadsheet. This also determined their sex. Any river herring that had died or were dying in the net were kept, while all other specimens were released. Fish that were released alive were only measured for standard length to reduce handling time and stress. Dead and dying fish were measured for standard length, fork length and total length. The dead fish were taken to the lab and dissected for ID and sex confirmation.

We used a published regression of fecundity by size and observed sex ratios in our catches to estimate fecundity, and to cross-check whether spawner abundance estimated from adult catches is plausible when compared to number of larvae collected. The following regression to estimate fecundity was used, this regression estimates only eggs ready to be spawned, which gives a more accurate picture than total egg count would (Lake and Schmidt 1997):

Egg # = -90,098 + 588.1(TL mm)

We used data from specimens where both standard length and total length was estimated to convert standard length to total length in cases we had not measured total length. Our data resulted in the following conversion: TL = 1.16SL + 6. The regression had an R^2 of 0.97.

Since the nets were set 24 hours per week for 10 weeks, we approximated total abundance of spawning Alewife and Gizzard Shad during the time of collection by extrapolating the mean catch per hour per species during the time the creeks were blocked off over the total collection period as follows:

Total catch/240 hours * 1680 hours = total abundance of spawners

Our total collection period is a good approximation of the total time of the spawning run of Alewife. To determine the number of females we used the proportion of females in the catch for Alewife as well as Blueback Herring, since we are able to sex Blueback Herring as well.

We did not determine the abundance of spawners based on the amount of larvae collected. Alewife and Gizzard Shad have fecundities of 60,000-120,000 eggs per female, and with the low numbers of larvae collected, we would grossly underestimate the abundance of spawning fish. Eggs and larvae also suffer very high mortality rates, so it is unlikely that 60,000-120,000 larvae suspended in the total discharge of a creek amount to one spawning female. Instead the method described above was used.

In response to problems with animals (probably otters) tearing holes in our nets in early years, we have been consistently using a fence device that significantly reduces this problem. The device effectively excluded otters and similar destructive wildlife, but had slots that allowed up-running fish to be captured. The catch was primarily Clupeids with little or no bycatch of other species.

Figure 1. Block net deployed in Pohick creek. The top of the block net is exposed at both high and low tide to avoid drowning turtles, otters, or other air-breathing vertebrates. The hedging is angled downstream in order to funnel up-migrating herring into the opening of the net.

Results

Our creek sampling work in 2016 spanned a total of 10 weeks, during which we collected 40 ichthyoplankton samples, and ten adult (block net) samples. We collected less adult clupeids than we did in 2015, which saw unprecedented high numbers, but more than we did in the years prior to 2015 (since the consistent block net collection method started in 2008). In 2010 Hickory Shad (*Alosa mediocris*) was captured for the first time in the history of the survey, after which we have continued to observe Hickory Shad in our samples. Hickory Shad are known to spawn in the mainstem of the Potomac River, and although their ecology is poorly understood, populations of this species in several other systems have become extirpated or their status is the object of concern. This year we captured a high number of adult Hickory Shad specimens in Accotink Creek, and some in Pohick Creek as well.

The abundance of *Alosa* larvae was a little bit higher than last year (184 versus 119 last year). There were less unidentified clupeids, with 108 unidentified clupeids versus 577 last year, which could be *Alosa* or *Dorosoma*; Gizzard Shad). We also collected 46 identified Gizzard Shad larvae. We found this year that the *Alosa* larvae consisted of Blueback Herring and Alewife larvae (Table 1). We did find adults of Hickory Shad, but no larvae.

Table 1. Larval and adult abundances of clupeids collected in both creeks in 2016.

	Pohick Creek		Accotink Creek	
Clupeid species	# larvae	# adults	# larvae	# adults
Alewife	111	94	66	76
Blueback Herring	7	80	0	9
Hickory Shad	0	21	0	108
Gizzard Shad	39	8	7	0
Unknown Clupeid	40	0	68	0

In 2016, as well the two previous years, *Dorosoma cepedianum* (Gizzard Shad) larvae are not the most abundant anymore. This is a good sign, since the reason for that are the increases in anadromous Alosines in our samples.

We measured creek discharge at the same locations and times where ichthyoplankton samples were taken. Discharge was much more variable this year in Accotink Creek than Pohick Creek and ranged from 0.12 to 4.85 $m^3 s^{-1}$, while Pohick Creek ranged from 0.68 to 2.65 $m^3 s^{-1}$ (Figure 2). On average and as in previous years, the discharge in Accotink Creek was lower than in Pohick Creek, with 0.97 $m^3 s^{-1}$ in Accotink Creek and 1.43 $m^3 s^{-1}$ in Pohick Creek. During the 70-day sampling period (which roughly coincides with the river herring spawning period), the total discharge was estimated to be on the order of 5.9 and 8.7 million cubic meters for Accotink and Pohick creeks, respectively (Table 2).

Larval density of Alewife exhibited a peak in Accotink Creek in late April (Figure 3a). Larval densities in Pohick Creek were lower and showed a small peak in mid-April. Given the observed mean densities of larvae and the total discharge, the total production of Alewife larvae was estimated at close to 2 and 1 million for Accotink and Pohick creeks, respectively (Table 2). Blueback Herring larval density was lower leading to total larval production estimates of 240 and 160 thousand for Accotink and Pohick creeks, respectively.

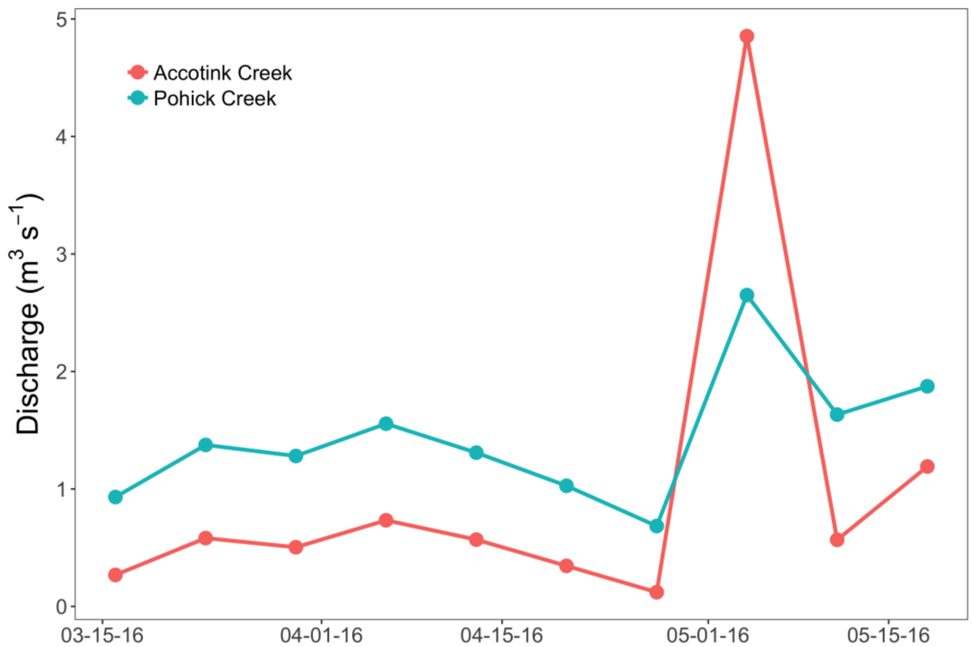

Figure 2. Discharge rate measured in Pohick and Accotink creeks during 2016.

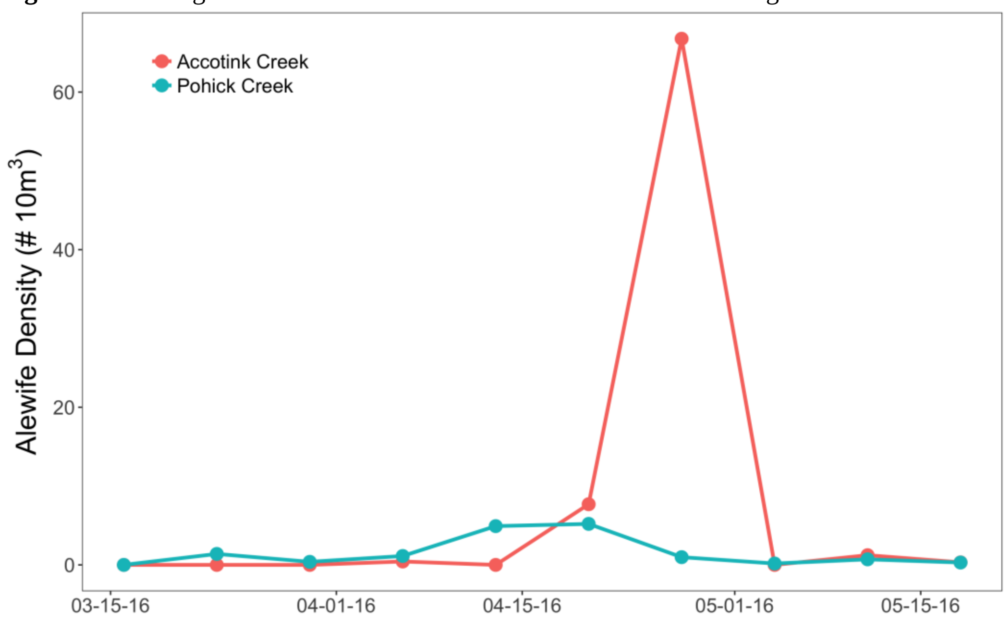

Figure 3a. Density of larval Alewife in # 10 m^{-3} observed in Pohick Creek and Accotink Creek in 2016.

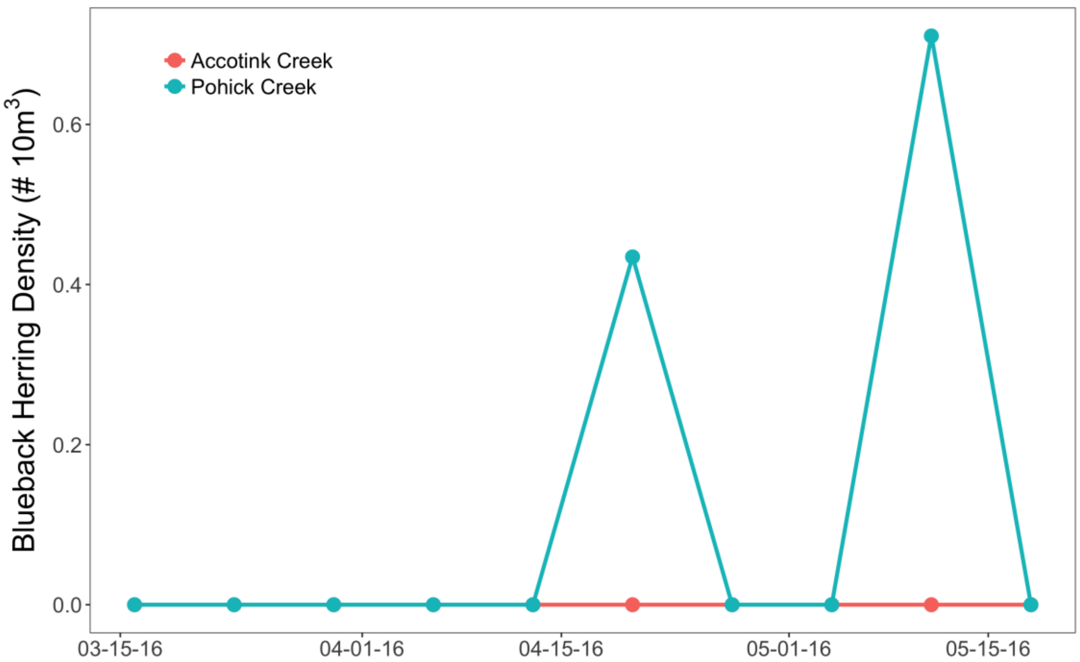

Figure 3b. Density of larval Blueback Herring in # 10 m^{-3} observed in Pohick Creek and Accotink Creek in 2016.

In the block net sets, a relatively high number of adults were captured for both Alewife and Blueback Herring; 170 and 89 respectively (Table 3). Both species were collected in unprecedented high numbers in 2015 relative to the rest of the period of record, and the abundance was a lot lower again in 2016. However, the abundance in 2016 is still high compared to years before 2015 (Table 3). Of those captured, 113 Alewife and 66 Blueback Herring were sexed, providing us with sex ratios (Table 2). Skewed sex ratios in fish populations are common. The total abundance of spawning Alewife was estimated to be 658 in Pohick Creek during the period of sampling, and 532 in Accotink Creek. The size of the spawning population of Blueback Herring is estimated to be 63 in Accotink Creek and 560 in Pohick Creek this year. Table 3 shows a summary of adult clupeid abundance collected in block nets from 2008-2016.

Table 2. Estimation of Alewife and Blueback Herring fecundity and spawner abundance from Accotink and Pohick creeks during spring 2015.

	Accotink Creek	Pohick Creek
Mean discharge ($m^3 s^{-1}$)	0.97	1.43
Total discharge, 3/15 to 5/19 (m^3)	5,887,728	8,654,688
Alewife		
Mean density of larval Alewife (10 m^{-3})	4.826	3.0983
Total larval production	2,831,317	2,681,481
Adult Alewife mean standard length (mm)	228.91	232.18
Alewife fecundity	75,413.40	77,626.80
Sex ratio (proportion female)	0.132	0.223
Estimated number of female Alewife	70	147
Estimated total number of Alewife	532	658
Blueback Herring		
Mean density of larval blueback (10 m^{-3})	0*	0.229
Total larval production	0*	198,192
Blueback mean standard length (mm)	215.5	221.5
Blueback Herring fecundity	65,928.05	69,971.7
Sex ratio (%F)	0.333	0.35
Estimated # of female Blueback Herring	21	196
Estimated total # of Blueback Herring	63	560

*No larval Blueback Herring were identified to species in Accotink Creek samples but the mean density of unidentified larval clupeids was 2.34.

Table 3. Total adult catch per year using block nets for 10 weeks during the spawning season of four Clupeid species that occur in this area.

	Pohick Creek				Accotink Creek			
	Blueback Herring	Hickory Shad	Alewife	Gizzard Shad	Blueback Herring	Hickory Shad	Alewife	Gizzard Shad
2008	0	0	8	2	0	0	0	0
2009	0	0	33	2	0	0	7	4
2010	0	31	130	9	0	0	79	4
2011	5	6	60	22	1	12	47	42
2012	7	3	58	5	0	0	12	2
2013	4	0	53	17	0	1	29	2
2014	27	6	52	21	0	1	8	28
2015	962	209	635	130	3	0	372	67
2016	80	21	94	8	9	0	76	108

Discussion

We caught 170 Alewife and 89 Blueback Herring; we have positively identified Blueback Herring in this survey since 2011. We also collected 21 Hickory Shad. These numbers are an order of magnitude lower than what we collected in 2015, but still high compared to what we have observed since at least 2008. The high abundance in 2015 could have been a combination of a strong year class, and the moratorium put in place in 2012. The estimated size of the spawning population of Alewife is still above a thousand fishes in 2016. We estimated about half that for Blueback Herring, which was found in relatively low numbers in Accotink Creek. This is likely a temperature effect. Blueback Herring prefer to spawn at higher temperatures than Alewife; >13 °C versus >10.5 °C for Alewife (Fay et al. 1983). By receiving effluent for the Noman Cole pollution control plant, Pohick creek is slightly warmer earlier in the season than Accotink Creek. It is possible that the Blueback Herring spawning season is actually taking place slightly later in Accotink Creek, rather than that the spawning population is smaller. Our sampling regime has been matched with the spawning season of Alewife when the understanding was that Blueback Herring does not use this area to spawn (the first Blueback Herring were identified in 2011). Continuing sampling into the summer would resolve whether the size of the Blueback Herring spawning population in Accotink Creek is small, or if the peak of the spawning period is simply taking place later. A spawning population of Blueback Herring has at least firmly established in Pohick Creek since 2011, and we will continue to provide population parameters of Blueback Herring in our reports, rather than Gizzard Shad (which is not a river herring).

With a moratorium established in 2012 in Virginia, in conjunction with moratoria in other states connected to the north Atlantic at the same time or earlier, the order of magnitude increase in Alewife and Blueback Herring abundance three years after this occurrence (in 2015) could be a result of the moratoria. The moratoria prohibit the capture and/or possession of river herring (Alewife and Blueback Herring). The three-year delay coincides with the time it takes for river herring to mature, which means this is the first year a cohort has been protected under the moratoria for a complete life cycle. The lower numbers in 2016 (while the moratoria are still in effect), indicate that the high abundances in 2015 are not just an effect of the moratoria, but perhaps a combination of that and having a good year class in 2015.

Through meetings with the Technical Expert Working group for river herring (TEWG; http://www.greateratlantic.fisheries.noaa.gov/protected/riverherring/tewg/index.html) it has become clear that not all tributaries of the Chesapeake Bay, in Virginia and elsewhere, have seen increased abundances as we saw here in 2015; some surveyors even reported declines (De Mutsert, personal communication). Since the general historic decline in river herring was related both to overfishing and habitat degradation, it could be the case that habitat in those areas has not recovered sufficiently to support a larger spawning population now that fishing pressure is released. This while the habitat in the Gunston Cove watershed is of suitable quality to support a larger spawning population now that reduced fishing pressure allows for more adults to return to their natal streams. The reduced numbers in 2016 as compared to 2015 may be the result of a density-dependent effect. The current available habitat or resources may not have been able to support an order of magnitude larger river herring population, and the numbers declined again because of that in 2016. Additional stressors could play a role in the variable success so far of the moratoria; while targeted catch of river herring is prohibited, river herring is still a portion of by-catch, notably of offshore midwater trawl fisheries (Bethoney et al. 2014). For the Gunston Cove watershed, 2015 was a highly productive year, and 2016 was less productive, but still above the 2008-2014 average. While it is too soon to tell what the long-term effects of the moratorium will be, and to what extent it affects the abundances in Potomac River tributaries, continued monitoring will determine whether some pattern of higher abundances is maintained in subsequent years.

Literature cited

Bethoney, N. D., K. D. E. Stokesbury, B. P. Schondelmeier, W. S. Hoffman, and M. P. Armstrong. 2014. Characterization of River Herring Bycatch in the Northwest Atlantic Midwater Trawl Fisheries. North American Journal of Fisheries Management 34(4):828-838.

Cummins, J.D. 2005. The Potomac River American Shad restoration project. 2004 Summary Report. Interstate Commission on the Potomac River Basin Report No. 05-2. 6 + 3 p.

Fay, C. W., R. J. Neves, and G. B. Pardue. 1983. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (mid-Atlantic). U.S. Fish and Wildlife Service, Blacksburg, VA.

Jones, P. W., F. D. Martin, and J. D. Hardy, Jr. 1978. Development of fishes of the Mid-Atlantic Bight: an atlas of egg, larval, and juvenile stages, volume 1. Acipenseridae through Ictaluridae. U.S. Fish and Wildlife Service, FWS/OBS-78/12.

Lake, T.R. and Schmidt. 1998. The relationship between fecundity of an Alewife (*Alosa pseudoharengus*) spawning population and egg productivity in Quassaic Creek, a Hudson River tributary (HRM 60) in Orange County, New York. Section II: 26 pp. *In* J.R. Waldman and W.C. Nieder (Eds). Final Reports of the Tibor T. Polgar Fellowship Program, 1997, Hudson River Foundation, NY.

Lippson, A. J., M. S. Haire, A. F. Holland, F. Jacobs, J. Jensen, R. L. Moran-Johnson, T. T. Polgar, and W. A. Richkus. 1979. Environmental atlas of the Potomac Estuary. Environmental Center, Martin Marietta Corp. 280 p.

Lippson, A. J., and R. L. Moran. 1974. Manual for the identification of early developmental stages of fishes of the Potomac River estuary. Maryland Department of Natural Resources, Baltimore.

Massmann, W.H. 1961. A Potomac River shad fishery, 1814 – 1824. Chesapeake Sci. 2 (1-2): 76-81.

NOAA (Department of Commerce). 2006. Endangered and Threatened Species; Revision of Species of Concern List, Candidate Species Definition, and Candidate Species List. Federal Register, Vol. 71, No. 200, Tuesday, October 17, 2006, pp. 61022-61025.

Smith, H.M., and B.A.Bean . 1899. List of fishes known to inhabit the waters of the District of Columbia and vicinity. U.S. Fish Commission Bulletin 18:179-187.

Walsh H.J., L.R Settle, and D.S. Peters. 2005. Early life history of Blueback Herring and Alewife in the lower Roanoke River, North Carolina. Transactions of the American Fisheries Society 134:910-926.

Development of an Benthic Index of Biotic Integrity for the Tidal Freshwater Potomac River

Final Report
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Introduction

Biological communities may serve as excellent indicators of the quality of environment as bioindicators.. Bioindicators represent the impact of environmental stress on a habitat, community or ecosystem (Mcgeoch 1998; Hodkinson et al. 2005). The utility of using aquatic invertebrates for assessing environmental conditions has been widely recognized, and a variety of biological monitoring tools are based on aquatic invertebrates (Hellawell 1986; Rosenberg and Resh 1993; Hodkinson and Jackson 2005). Benthic aquatic communities, in particular, have proved useful as most members are fixed in location. Therefore, their presence is related to the overall conditions at that site. Benthic macroinvertebrates are among the most useful of all indicator organisms and are the community of choice in the bioassessment of flowing streams worldwide. This is because they have life cycles which are long enough to integrate over a significant amount of time but can recolonize an area relatively quickly if conditions improve. In addition, aquatic insect larvae as also used as bioindicators because they are abundant, easily collected, and represent the trophic connection between and lower plants and higher trophic predators (Hodkinson and Jackson, 2005). For example, the presence of the orders Ephemeroptera, Plecoptera and Trichoptera (EPT; mayfly, stonefly and caddisfly) in aquatic environments indicates good water quality as they are sensitive to pollution (Rosenberg and Resh 1993; Siegloch et al. 2017). Some species of chironomid (midge larvae) are more pollution tolerant and found in both good quality and bad quality water (Halpern and Senderovich 2015). In comparison, taxa groups such as Oligochaetes (i.e., worms) are found in poor quality water with low dissolved oxygen and high organic load. Thus, the presence and absence of such benthic invertebrates are good bioindicators of water body health. Biomonitoring these species, including examining community characteristics such as abundance, diversity, and richness, can be an invaluable way to track trends in water quality over time.

While examining the macroinvertebrate communities over time can give some indication of overall changes, the magnitude of those changes in relation to particular stressors is hard to pinpoint without an adequate undisturbed habitat with which to compare. Therefore, establishment of similar reference sites relatively free of stressors (e.g., anthropogenic) provides a way to document site-level changes to macroinvertebrate communities on a temporal scale. Such a method is called an index of biotic integrity, or IBI. The index assigns categorical values for different metrics (e.g., habitat, water quality, macroinvertebrate taxa diversity and abundance) by comparison with observations at reference sites. In general, higher IBI scores represent unimpaired or unstressed benthic community conditions.

Below, we examine existing efforts which have been made to develop benthic indices of biotic integrity (B-IBIs) for tidal freshwater systems. We compiled a list of benthic taxa that have been found historically or are currently being collected in the tidal freshwater habitats of the Potomac River. The literature on the basic biology of these organisms was researched to determine their tolerance scores. We then lay out plans on how to develop and enhance the B-IBI specifically for tidal freshwater habitats of the Potomac River.

Existing efforts in developing B-IBIs for tidal freshwater systems

While many previous studies have established effective indices of biotic integrity (IBIs) for freshwater systems in general (e.g. Clements et al., 1992; Lenat, 1993; Kerans and Karr, 1994; Lang and Reymond, 1995), IBIs are now being developed and applied in tidal systems. For example, researchers in the Chesapeake Bay developed a benthic index of biotic integrity (B-IBI) that is applicable in a wide range of conditions throughout the Bay but is generally based on subtidal, unvegetated, infaunal microbenthic communities (Weisberg et al. 1997). In estuaries, salinity greatly affects the potential pool of species to be found at a location, and Weisberg et al. (1997) took that into account when developing the B-IBI. Therefore, seven distinct "habitats" were created on a salinity scale, and methods for creating a B-IBI were developed for each habitat. Four metrics are currently being used in the B-IBI for the "tidal freshwater" habitat, including total abundance of individuals (#/m^2), percent abundance of pollution-indicative taxa, percent of abundance of deep-deposit feeders, and tolerance scores (i.e., the range of contaminant or pollutant values a species is able to tolerate) (Llanso and Dauer 2002). Dauer et al. (2000) used these B-IBI scores to compare benthic community composition scores with measures of pollution exposure, finding that dissolved oxygen alone explained 42% of the variation in the B-IBI.

While the Chesapeake Bay B-IBI accounts for fluctuates in salinity that impact species composition, it has not been very successful in tidally influenced freshwater systems like the Potomac River in Northern Virginia. In an assessment of the Chesapeake Bay B-IBI, Alden et al. (2002) found that the classification effectiveness of the index increased with increasing salinity, and performed poorly in tidal freshwater systems. The tidal Potomac River is one of the major sub-estuaries of the Chesapeake Bay system, but shares only a portion of the same species with the larger Chesapeake Bay (Jones et al. 2008). Previous studies have shown that eutrophications processes in the Potomac River have been accelerated by human activities resulting in nutrient over enrichment, increased algal blooms, and hypoxia (Walker et al. 2000). As Gunston Cove receives a large input of wastewater, there are ongoing efforts to keep nutrient loading into the system at a minimum. Since research began in the 1980s, there has been a reduction in nutrients such as phosphorous, nitrogen, sodium, and effluent chlorine, leading to an overall improvement in water quality (Jones 2015). Past studies have focused on fish populations, submerged aquatic vegetation, ichthyoplankton, zooplankton and water quality. However, there are limited studies on aquatic benthic macroinvertebrates in this system. Indeed, in their assessment of the Chesapeake Bay B-IBI, Alden et al. (2002) suggest that the inaccuracy of the B-IBI in tidal freshwater systems stemmed from either "difficulties in reliably identifying naturally unstressed areas" or "differences in regional ecotones created by stress gradients". The later explanation refers to natural abiotic variability related to flow, sedimentation rates, and turbidity across various tidal freshwater habitats.

The current Chesapeake Bay B-IBI represents a starting point for development of a B-IBI for the tidal freshwater Potomac River, but much refinement is needed. For example, the only pollution-indicative taxa for the "tidal freshwater" habitat listed is Oligochaeta. Other taxa, such as species of Chironomidae (i.e., nonbiting midges), should be considered for addition to this list. Also, tolerance scores are not available for many of the taxa routinely found in Gunston Cove collections. Interestingly, in their assessment of the Chesapeake Bay B-IBI, Alden et al. (2002)

found that neither abundance nor biomass correctly discriminated between degraded and non-degraded tidal freshwater sites as single metrics, and a full B-IBI was needed in these systems. For tidal freshwater systems, the most important metrics in classifying sites were pollution-indicative species abundance and deep-dwelling deposit feeder abundance (Alden et al. 2002). Even then, the B-IBIs obtained for the tidal freshwater systems were extremely variable, highlighting the need for refinement (Alden et al. 2002). Using the same B-IBI, De la Ossa Carretero et al. (2016) found misclassifications of degraded or undegraded status of large water bodies in tidal freshwater systems 59% of the time. This was reduced to only 33% of the time in smaller water bodies (De la Ossa Carretero et al. 2016).

While not totally applicable to tidal freshwater systems, Astin (2007) created a B-IBI for non-tidal streams in the Potomac River basin that included various metrics for stream habitat, water quality, and macroinvertebrate taxa data. In her analysis, Astin (2007) found that there were seven macroinvertebrate community metrics that classified sites, including Ephemeroptera, Plecoptera and Trichoptera (EPT; mayfly, stonefly and caddisfly) richness, Hilsenhoff Family Biotic Index, percent clingers, percent collectors, percent dominance, percent EPT, and taxonomic richness.

Several others have incorporated various types of B-IBIs into other regional assessment systems (e.g., Tennessee Valley rivers – Kerans and Karr 1994; Persian Gulf – Doustshenas et al. 2009; Anacoastia River, DC – McGee et al. 2008; Northern California streams – Rehn et al. 2005; Maryland streams – Southerland et al. 2006), but none of these focus on the tidal freshwater ecosystem. Indeed, there are actually a multitude of potential biotic indices that have been developed to focus on specific regions or issues, but the vast majority of these are not applicable in tidal freshwater systems (see review by Pinto et al. 2009).

Resident macroinvertebrate community assemblages vary greatly due to a multitude of environmental factors, but one of the most influential is habitat and sediment characteristics. In the tidal freshwater Potomac River, there are several habitat types that can be sampled for benthic macroinvertebrates, including both coarse and fine sediments and aquatic plants. Therefore, in some cases, multiple B-IBIs must be created, one for each habitat type, as reference conditions in one habitat type may not be indicative of all habitat types. This leads to establishment of habitat-specific reference conditions (Kerans and Karr 1994). Astin (2007) used this approach to classify the Piedmont, Valleys, and Highlands streams of the Potomac River basin. Incorporating habitat type into our index (along with water quality and macroinvertebrate taxa data) will allow for a more comprehensive examination of the resident taxa across multiple habitat types.

Historical and current benthic taxa and tolerance scores in the tidal freshwater Potomac River

The first step in our project is to compile a comprehensive list of taxa which are expected to inhabit the tidal freshwater Potomac. This effort has been initiated by pooling information from three sources. First, the list of taxa recently collected in the Gunston Cove and Hunting Creek tidal benthic samples were accumulated. Then we accessed the list of macroinvertebrate taxa from the comprehensive "Environmental Atlas of the Potomac Estuary" compiled by Lippson et al. from studies funded by Maryland's Power Plant Siting Program. Finally, the taxa list from a study of macroinvertebrate communities in SAV beds by Thorp et al. 1997 was accessed.

Calculations of many benthic macroinvertebrate metrics rely on assigned taxonomic attributes, or traits. We assigned each taxon reported from our studies in the tidal freshwater Potomac River to the following traits: Functional Feeding Group, Habit, and Stress Tolerance Value reported in the literature (Barbour et al. 1999; USEPA 2008, 2012; Chalfant 2009; Bollman et al. 2010; Buchanan et al. 2011; WVDEP 2015; Smith 2016). Functional feeding group classifications divide taxa based on behavioral mechanisms of food acquisition rather than taxonomic identification. Thus, different taxa that acquire food in the same manner can be in the same functional feeding group. In general, aquatic macroinvertebrates are placed into the following functional feeding groups: scrapers/grazers (consume algae and biofilms), shredders (consume leaf litter and coarse particulate organic matter), collectors/gatherers (collect fine particulate organic matter from the stream bottom), filterers (collect fine particulate organic matter from the water column), and predators (feed on other consumers). Habit here denotes in what part of the substrate or habitat different taxa occupy and span five categories: burrowing into the sediment, swimming periodically through the water to change location, clinging to a surface (plant or rock), a combination of clinging and burrowing, and sprawling on top of the sediment surface. Another metric, stress tolerance values, range from 0-10, and represent the ability of that taxa to survive environmentally stressful conditions. These stressful conditions include abiotic examples like low dissolved oxygen, increased chemical concentrations, or increased temperature. Taxa that have low tolerance values do not survive well in stressful conditions, while taxa with higher tolerance scores can survive environmental stress for a period of time. Taxa with multiple attributes/traits were assigned a single attribute/trait based on the most frequently assigned attribute/trait or best professional judgement.

Starting with stressful tolerance values, in general, the majority of the taxa listed have high (>6) scores (Table 1), indicating that most of the taxa currently found in the freshwater tidal Potomac River are tolerant of environmental stressors. This could be interpreted two ways. It is possible that repeated and increasing levels of stress to this area over time have resulted in a population that is now only composed of those species that could tolerate the stressors. This line of reasoning is supported due to the fact that there are relatively few species (16%) that have low (<4) tolerances to stress (Table 1). However, these taxa may be the baseline community that was there the entire time and have always been exposed to some level of stress.

All five functional feeding groups were represented in the taxa list, with the majority (33%) being collector gatherers, followed closely by predators (27%) (Table 1). There was only one shredder in the list (Diptera: Tipulidae). Similarly, all five habits were represented, with the majority (31%) belonging to the clinger-burrower group (Table 1). There were relatively few swimmers (N=2; 6%) relative to the rest of the groups (Table 1).

In general, the level of taxa diversity in regards to tolerance scores, habit, and functional feeding groups is high in this region, indicating that a possible B-IBI could be established. Comparing to Astin (2007)'s list of seven macroinvertebrate community metrics that classified sites (Ephemeroptera, Plecoptera and Trichoptera (EPT) richness, Hilsenhoff Family Biotic Index, percent clingers, percent collectors, percent dominance, percent EPT, and taxonomic richness), it appears that this survey does not have many representatives of the order Plecoptera that could contribute to the overall EPT scores. However, there are taxon representatives from the other two groups. The other metrics listed by Astin (2007) could be easily obtained from a data set containing the taxa represented here. However, it must be stressed that in order to create a useable B-IBI, reference sites that represent unaltered conditions are needed. It could be assumed taxa from unaltered locations would have lower tolerance values and, therefore, would be able to be compared with other sites on a scale of impairment.

Moving forward

To create a specific B-IBI for the tidal freshwater portion of the Potomac River, we will need to embark on a stepwise process. First, we need to establish a complete potential taxa list. There are several other references lists including very early surveys that we are aware of and we will check them for further taxa. We also plan to survey investigators in other tidal freshwater habitats along the mid-Atlantic for additional taxa. Second, we need to identify reference sites with similar physical characteristics as the tidal freshwater Potomac, but with a much lower human impact level. Most of the rivers which empty into Chesapeake Bay have a tidal freshwater reach and some, like the Rappahannock have had a lower human imprint. We plan to identify potential sites based on consultation with other investigators and inspection of available data. Sampling these less impacted systems would help us to complete the taxa and establish reference conditions for metric development.Then we need to decide on and calibrate the various abiotic and biotic metrics that would go into the B-IBI (e.g., habitat, water quality, macroinvertebrate taxa data). After that, we would calibrate the B-IBI scores using the metrics we defined, and then test the B-IBI with previous and/or new data using jackknife validation. These are the steps taken by both the team that established the Chesapeake Bay B-IBI and Potomac River Basin B-IBI.
Using reference sites that are minimally disturbed is the most important component of IBI development (Southerland et al. 2007), and may be the hardest to find in the tidal freshwater Potomac River. Southerland et al. (2007) notes that if reference sites are only slightly less disturbed than other sites, developing robust IBI scores to differentiate between “degraded” and “undegraded” becomes difficult. Ultimately, a location-specific B-IBI for the tidal freshwater area of the Potomac River would provide a tool to determine the relative health of local waterways, investigate the anthropogenic impacts and subsequent watershed protection measures, evaluate trends in stream health, and identify areas in need or restoration or protection.

Goals for next year

We recommend that this project be continued. Goals for next year would be:

- Completion of the taxa list
- Identification and possible sampling of less impacted sites like those on the Rappahannock
- Development and calibration of metrics
- Formulation and testing of the B-IBI

We expect to be able to accomplish many of these goals. As we work through them, we may find that further data must be collected to create a valid index. In that case our report next year will identify further work that must be done.

Table 1. Benthic Macroinvertebrate taxa reported from the tidal freshwater Potomac River.

Group	Order	Family	Genus and species	Tol. Val.	FFG	Habit	REF
Amphipod	Amphipoda	Crangonyctidae	Stygobromus	5	PR	SP	1
Amphipod	Amphipoda	Gammaridae	Gammarus sp.	6	CG	SP	1,2
Amphipod	Amphipoda	Hyallelidae	Hyalella sp.	8	CG	SP	1,2
Amphipod	Amphipoda	Oedicerotidae	Monoculodes				2
Amphipod	Amphipoda	Talitridae		8	CG	-	1
Bivalve	Veneroida	Cyrenidae	Corbicula fluminea	6	CF	BU	1
Bivalve		Mactridae					2
Bivalve		Sphaeriidae		8	CF	BU	1
Bivalve		Unionidae					2
Bryozoan		Pedicellinidae					2
Bryozoan	Plumatelida	Lophopodidae					2
Flatworm	Catenulida (class)	Stenostomidae					2
Flatworm	Rhabdophora (class)	Microstomidae	Microstomum				2
Flatworm	Rhapditophora (class)	Macrostomidae					2
Flatworm	Tricladida	Dugesiidae					1,2
Insect	Coleoptera	Elmidae		5	CG	CN	1
Insect	Diptera	Ceratopogonidae		6	PR	SP	1
Insect	Diptera	Chaoboridae	Chaoborus sp.	8	PR	SP	1
Insect	Diptera	Chironomidae		6	CG	BU	1,3
Insect	Diptera	Ephydridae		6	CG	BU	1
Insect	Diptera	Simulium		5	CF	CN	1
Insect	Diptera	Tipulidae	Antocha sp.	4	CG	CN	1
Insect	Diptera	Tipulidae		4	SH	BU	1
Insect	Ephemeroptera	Baetidae		5	CG	SW	1,3
Insect	Ephemeroptera	Caenidae					3
Insect	Ephemeroptera	Ephemeridae					3
Insect	Odonata	Calopterygidae	Calopteryx	6	PR	CB	1
Insect	Odonata	Coenagrionidae	Amphiagrion	6	PR	BU	1
Insect	Odonta	Coenagrionidae	Argia sp.	6	PR	CN	1,3
Insect	Trichoptera	Hydropsychidae	Hydropsyche	5	CF	CN	1
Insect	Trichoptera	Hydroptilidae		4	PR	CB	1,3
Insect	Trichoptera	Leptoceridae		4	CG	CB	1,3
Insect	Trichoptera	Philopotamidae	Dolophilodes	1	CF	CN	1
Isopod	Isopoda	Anthuridae	Cyathura sp.				1
Isopod	Isopoda	Asellidae	Asellus				1,2
Isopod	Isopoda	Chaetiliidae	Chiridotea sp.				2
Leech (Annelid)	Hirudinea	Glossiphoniidae					2,3
Leech (Annelid)	Hirudinea	Piscicolidae					2
Water Mite	Subclass Acari	Hydrachnidae		6	PR	SW	1,3
Snail	Gastropoda (class)	Amnicolidae					2
Snail	Gastropoda (class)	Ancylidae		7	SC	CB	1,2
Snail	Gastropoda (class)	Hydrobiidae		7	SC	CB	1
Snail	Gastropoda (class)	Hydrobiidae					3
Snail	Gastropoda (class)	Lymnacidae	Lymnaea	7	SC	CB	1,2
Snail	Gastropoda (class)	Physidae					2
Snail	Gastropoda (class)	Planorbidae		7	SC	CB	1,2
Snail	Gastropoda (class)	Pleuroceridae		7	SC	CB	1
Snail	Gastropoda (class)	Valvatidae		7	SC	CB	1
Snail	Gastropoda (class)	Viviparidae		7	SC	CB	1
Worm (Annelid)	Oligochaeta	Aeolosomatidae					2
Worm (Annelid)	Oligochaeta	Nadidae					2
Worm (Annelid)	Oligochaeta	Tubificidae					2
Worm (Annelid)	Polycheata (class)	Nereidae					2

Notes: Functional feeding group abbreviations: CG = Collector Gatherer; PR = Predator; SH = Shredder; CF = Collector Filterer; SC = Scraper Grazer.
Habit abbreviations: BU = Burrower; SW = Swimmer; CN = Clinger; CB = Clinger Burrower; SP = Sprawler.
Reference abbreviations: 1-Gunston Cove Study Reports. 2-Lippson et al. 1981. 3-Thorp et al. 1997.

References

Alden, R. W. III, D. M. Dauer, J. A. Ranasinghe, L. C. Scott, and R. J. Llansó. 2002. Statistical Verification of the Chesapeake Bay Benthic Index of Biotic Integrity. *Environmetrics* 13: 473-498.

Astin, L. E. 2007. Developing biological indicators from diverse data: The Potomac Basin-wide Index of Benthic Integrity (B-IBI). Ecological Indicators 7:895-908.

Barbour, M. T., Gerritsen, J., Snyder, B. D. and J. B. Stribling. 1999. Rapid bioassessment protocols for use in wadeable streams and rivers. Periphyton, Benthic Macroinvertebrates, and Fish (2nd ed.). US Environmental Protection Agency, Office of Water, Washington, DC EPA.

Bollman, W., Bowman, J. and D. Winter. 2010. Analysis of Biological Samples: District of Columbia Phytoplankton, Zooplankton, and Benthic Macroinvertebrate Samples: 2005-2009. Rhithron Associates, Inc., Missoula, Montana.

Buchanan, C., Foreman, K., Johnson, J, and A. Griggs. 2011. Development of a Basin-wide Benthic Index of Biotic Integrity for Non-tidal Streams and Wadeable Rivers in the Chesapeake Bay Watershed: Final Report to the Chesapeake Bay Program. Interstate Commission on the Potomac River, ICPRB Report:11–1.

Chalfant, B. 2009. A benthic index of biotic integrity for wadeable freestone streams in Pennsylvania. Pennsylvania Department of Environmental Protection, Division of Water Quality Standards. URL: http://www. Depweb. State. Pa. Us/watersupply/lib/watersupply/ibi_rifflerun. Pdf

Clements, W. H., Cherry, D. S., and J. H. Van Hassel. 1992. Assessment of the impact of heavy metals on benthic communities at the Clinch River (Virginia): evaluation of an index of community sensitivity. Canadian Journal ff Fisheries and Aquatic Science 49:1686–1694.

Dauer, D.M., J. A. Ranasinghe, and S. B. Weisberg. 2000. Relationships between benthic community condition, water quality, sediment quality, nutrient loads, and land use patterns in Chesapeake Bay. *Estuaries* 23:80-96.

De-la-Ossa-Carretero, J.A., M. F. Lane, R. J. Llansó, and D, M. Dauer. 2016. Classification efficiency of the B-IBI comparing water body size classes in Chesapeake Bay. *Ecological Indicators* 63:144–153.

Doustshenas, B., Savari, A., Nabavi S. M. B., Kochanian, P., and M. Sadrinasab. 2009. Applying benthic index of biotic integrity in a soft bottom ecosystem in north of the Persain Gulf. Pakistan Journal of Biological Sciences 12 (12):902-907.

Halpern, M., and Y. Senderovich. 2015. Chironomid Microbiome. Microbial Ecology 70:1-8.

Hellawell, J. M. 1986. Biological indicators of freshwater pollution and environmental management. Elsevier, London.

Hodkinson, I. D., and J. K. Jackson. 2005. Terrestrial and aquatic invertebrates as bioindicators for environmental monitoring, with particular reference to mountain ecosystems. Environmental Management 35:649–666.

Jones, R. C. 2015. The Ongoing aquatic monitoring program from the Gunston Cove Area of the tidal freshwater Potomac River. Final Report.

Jones, R. C. 2008. Long term response of water quality to changes in nutrient loading at a mainstem site in the tidal freshwater Potomac River. *Verhandlungen Internationale Vereinigung Limnologie* 30(4):633-637.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecological Applications 4:768–785.

Lang, C. G., and O. Reymond. 1995. An improved index of environmental quality for Swiss rivers based upon benthic invertebrates. Aquatic Science 57:172–180.

Lenat, D. R. 1993. A biotic index for the southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings. Journal of the North American Benthological Society 12:279–290.

Lippson, A.J., M.S. Haire, A.F. Holland, F. Jacobs, J. Jensen, R.L. Moran-Johnson, T.T. Polgar, and W.A. Richkus. 1981. Environmental Atlas of the Potomac Estuary. Power Plant Siting Program. Maryland Department of Natural Resources. 280 pp.

Llansó, R. J., L. S. Scott, D. M. Dauer, J. L. Hyland and D. E. Russell. 2002. An estuarine benthic index of biological integrity for the Mid-Atlantic region of the United States. I. Classification of assemblages and habitat definitions. *Estuaries* 25:1219-1230.

Mcgee, B. L., Pinkney, A. E., Velinsky, D. J., Ashley, J. T., Fisher, D. J., Ferrington, L. C. and Norberg-King, T. J., 2009. Using the Sediment Quality Triad to characterize baseline conditions in the Anacostia River, Washington, DC, USA. Environmental Monitoring and Assessment 156:51-67.

Mcgeoch, M. A. 1998. The selection, testing and application of terrestrial insects as bioindicators. Biological reviews of the Cambridge Philosophical Society 73:181–201.

Pinto, R., Patrício, J., Baeta, A., Fath, B. D., Neto, J. M. and J. C. Marques. 2009. Review and evaluation of estuarine biotic indices to assess benthic condition. Ecological Indicators 9(1):1-25.

Rehn, A. C., Ode, P. R., May, J. T. 2005. Development of a benthic index of biotic integrity (B-IBI) for wadeable streams in Northern Coastal California and its application to regional 305(b) assessment. Final Technical Report. California Water Boards.

Rosenberg, D. M. and V. H. Resh. 1993. Introduction to freshwater biomonitoring and benthic macroinvertebrates. In: Rosenberg, D. M. and Resh, V. H. (Eds): Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York.

Siegloch, A. E, Schmitt, R., Spies, M., Petrucio, M., and M. I. M. Hernandez. 2017. Effects of small changes in riparian forest complexity on aquatic insect bioindicators in Brazilian subtropical streams. Marine & Freshwater Research 68:519-527.

Smith, A. J. 2016. Standard Operating Procedure: Biological Monitoring of Surface Waters in New York State. Standard Operating Procedure, New York State Department of Environmental Conservation, Division of Water.

Southerland, M. T., Rogers, G. M., Kline, M. J., Morgan, R. P., Boward, D. M., Kazyak, P. F., Klauda, R. J. and S. A. Stranko. 2007. Improving biological indicators to better assess the condition of streams. Ecological Indicators 7(4):751-767.

Thorp, A.G., R.C. Jones, and D.P. Kelso. 1997. A comparison of water-column macroinvertebrate communities in beds of differing submersed aquatic vegetation in the tidal Potomac River. Estuaries 20: 86-95.

USEPA. 2008. NRSA 0809 Benthic Taxa List and Autecology - Data. Environmental Protection Agency.

USEPA. 2012. Freshwater Biological Traits Database. United States Environmental Protection Agency.

Walker H. A., J. S. Latimar, and E. H. Dettmann. 2000. Assessing the effects of natural and anthropogenic stressors in the Potomac Estuary: Implication for long-term monitoring. Environmental Monitoring and Assessment 63:237-251.

Weisberg, S. B., Ranasinghe, J. A., Dauer, D. M., Schaffner, L. C., Diaz, R. J., and J. B. Frithsen. 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries 20:149–158.

WVDEP. 2015. Watershed Assessment Branch Benthic Macroinvertebrate Taxa Autecology Data Table. West Virginia Department of Environmental Protection, Division of Water and Waste Management, Watershed Branch, Charleston, WV.

Status and Diversity of Native Freshwater Mussels In the Tidal Freshwater Potomac River

Final Report
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By

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Introduction

Freshwater mussels (Mollusca: Order Unionoida) are important components of rivers, streams, lakes, and tidal freshwater systems, providing a diversity of ecosystem functions and services such as filtering water, biodeposition of organic matter, enhancing deposition of fine sediment, and providing habitat complexity for other species (Ragnarsson and Raffaelli 1999; Lopes-Lima et al. 2015). Freshwater mussels are a morphologically and genetically diverse group found globally in most freshwater systems, but the greatest diversity (300 species) occurs in North America (Howard et al. 2004). They are sensitive to anthropogenic inputs and alterations to the environment, both of which have increased over the past century and have contributed to the decline of freshwater mussel populations worldwide (Howard et al. 2004). This has led to the classification of over 65% of unionoids in the United States and Canada as presumed extinct, possibly extinct, critically imperiled, imperiled, or vulnerable (Howard et al. 2004).

Freshwater mussel populations have also been negatively impacted by the introduction of other bivalve species such as the Asian clam (*Corbicula fluminea*). *Corbicula fluminea* first appeared in the United States in the 1920s and within 40 years it had spread coast to coast (Sousa et al. 2008). The rapid growth and development of *C. fluminea* as well as its broad tolerance of a wide variety of substrates and water quality contributed to its dispersal across the United States (Sousa et al. 2014). *Corbicula fluminea* has significant negative impacts on native mussels by competing for space and resources as well as being a vector for parasites and pathogens (Darrigan 2002; Sousa et al. 2008). The first report of *C. fluminea* in the tidal Potomac River, Maryland occurred in 1977 and was described by Dresler and Cory (1980). Individuals are currently found from the tidal headwaters into the tidal-freshwater estuary (Bogan and Ashton 2016). The bivalve fauna in the tidal freshwater habitats of the Potomac River was mainly restricted to *C. fluminea* during the latter part of the 20th century. However, native river mussels have been collected recently with greater frequency, including specimens of both *Anodonta* sp. and *Elliptio* sp. in 2016. These may constitute a harbinger of recovery of the native bivalve community.

In order to establish the status and trends in abundance of these imperiled freshwater mussels, we conducted a literature review of the native freshwater mussels in the tidal freshwater Potomac River. We also include the invasive Asian Clam (*Corbicula fluminea*) for comparative purposes. We have included the scientific and common names for each species, along with species specific overviews of their reproductive mode, season, and habitat (Table 1); conservation status for each species in both Virginia and Maryland (Table 2); and all scientific name synonyms (Table 3). We also describe major current anthropogenic threats for local species. We end by describing sampling methods for collecting and assessing freshwater mussels and make recommendations on how these should be implemented in the future.

Species List

A literature review for mussels in the tidal freshwater Potomac River shows that there are 15 extant species, two locally extirpated species, and two species with an unknown history (Table 2). These include 19 native species in the Order Unionoida and one non-native species in the Veneroida (*Corbicula fluminea*). We include the invasive Asian clam, *C. fluminea*, in our report since it directly competes with and impacts the distribution of native mussels (Sousa et al. 2008).

Of the 19 species of Unionoid mussels in the tidal freshwater Potomac River, *Elliptio, Alasmidonta,* and *Lampsilis* are the most taxonomically diverse genera. *Elliptio* contains five species, *Alasmidonta* with four, and *Lampsilis* with three. All other genera are represented by only one species in the tidal freshwater Potomac River.

Historically, there has been confusion over the taxonomic status of *Elliptio* species in Virginia. Johnson (1970) mistakenly called both species *Elliptio fisheriana* and *Elliptio producta* an entirely different species - *Elliptio lanceolata* (Chazal and Roble 2011). In addition, *Elliptio angustata*, *Elliptio fisheriana,* and *Elliptio producta* were often used interchangeably in both peer-reviewed manuscripts and museum collections during the 1990s and 2000s, further adding to the confusion. Recent molecular work showed that *Elliptio fisheriana* is the most abundant in Virginia (Bogan et al. 2003). Due to the confusion of this group, the original determination is used for this report, as has also been used by Bogan and Ashton (2016) to describe freshwater mussels in Maryland.

Life History

Many freshwater mussels can be long-lived, with some species living up to 10 years and others over 100 (Bauer and Wächtler 2000). Certain groups, such as the genera *Pyganodon, Leptodea,* and *Utterbackia*, trade off longer life for faster growth (Cummings and Bogan 2006). However, the life cycle for most freshwater mussels includes an internal brooding stage of fertilized eggs, followed by release of glochidia larvae that are obligatory parasites on a variety of fish hosts (Lopes-Lima et al. 2015). Of the mussel species with known host fish species found in the tidal freshwater Potomac River, the majority (87%) are generalists in terms of what host fish their glochidia can parasitize (Table 1). Only two species (*Leptodea ochracea* and *Ligumia nasuta*) are specialists that can only parasitize one or two species of fish (Bogan and Ashton 2016) (Table 1). However, there are four species whose host fish are currently unknown (Table 1).

After attaching to the fish, the fish responds by forming a cyst around the glochidia, allowing the glochidia to feed directly on the hosts body fluids (Arey 1932). Development for most species of mussels while on the gills or skin of the fish host is usually 3 to 6 months (but can be between 3 days and 10 months), and after this period the free-living juvenile mussel is released (Johnson 1970; Bogan 2008). Transformation from juvenile to adult occurs with the development of a foot (Cummings and Bogan 2006), and the sexually mature, adult mussel finds a suitable habitat to begin reproducing (Johnson 1970). In freshwater mussels, the sexes are usually separate, but two species in the tidal freshwater Potomac River, the Paper Pondshell

(*Utterbackia imbecillis*) and Green Floater (*L. subviridis*) are hermaphroditic (Cummings and Bogan 2006). A summary of the reproductive strategies of the locally occurring species is presented in Table 1.

There are two main types of freshwater mussel reproduction that vary in the length of time between fertilization and larval hatching. Bradytictic reproduction is when spawning takes place in the late summer, and the female broods the fertilized eggs overwinter and releases the glochidia the following spring. Tachytictic reproduction is when spawning takes place in the spring and glochidia are released during the following summer. All mussel species existing in the tidal freshwater Potomac River have glochidia larvae, and most species (79%) are bradytictic, spawning between April and July (Table 1). The fecundity of North American mussels varies across species, but can range from anywhere to 2000 to 10 million larvae per spawning event (Haag 2013).

Other groups of freshwater bivalves have slightly different methods of reproduction. In the tidal freshwater Potomac River, the invasive Asian clam, *Corbicula fluminea*, is a simultaneous hermaphrodite, capable of self-fertilization (Sousa et al. 2008). *Corbicula fluminea* does not need a host fish, but instead broods larvae, and direct larval development leads to crawl-away juveniles (average N = 68,678) (Sousa et al. 2008). Once released into the water, juveniles can settle onto vegetation or hard surfaces via byssal threads, bury into the substratum, or be re-suspended by water turbulence and dispersed over long distances downstream (Boltovskoy and Cataldo 1999; McMahon 2000). *Corbicula fluminea* reaches maturity in the next 3-6 months and can live up to 5 years (McMahon 2000). The reproductive period varies by ecosystem and is most likely related to temperature and food availability, with some populations reproducing once, twice, or three times a year (Doherty et al. 1987; Darrigran 2002). However, the majority of populations reproduce twice a year (i.e., spring to summer and late summer to autumn). *Corbicula fluminea* has high fecundity, low juvenile survivorship and high adult mortality that results in populations dominated by juveniles, although this too can vary by ecosystem (McMahon 2000).

Distribution

Various factors can impact the distribution of freshwater mussels, such as host fish abundance for mussels requiring a host fish for larval development, habitat loss and fragmentation such as dams and channelization, and abiotic factors that can impact water quality by lowering survival rates of larvae such as mercury or arsenic contamination (Eisler 1988; Lopes-lima et al. 2015). In general, large-scale patterns of mussel abundance are regulated by host fish abundance, and small-scale patterns are determined by water current, velocity, and sediment type and quality (Chazal and Roble 2011). Stream hydrology can also impact substrate stability, which can impact mussel species richness and abundance (Allen and Vaughn 2010). Small-scale patterns of distribution are discussed in more detail in the habitat section, and large-scale factors are elaborated on in the conservation status section.

Habitat

The most common habitat for freshwater mussels is streams, although species specific habitat preference varies, with some species associated with a particular sediment or substrate type and others associated with a particular flow regime (Cummings and Bogan 2006) (Table 1). The range of habitats in which freshwater mussels can be found includes anything from shallow lakes and ponds to large rivers or artificial impoundments. Hydraulic and substrate variables often interact to limit the distribution of mussels with some variables being more limiting depending on the species (Allen and Vaughn 2010). The full list of specific habitat preferences for mussels found in the tidal freshwater Potomac River is presented in Table 1.

In general, adult mussels bury themselves in a soft substrate or attach to hard substrates for much of their lives, and the substrate preference varies between species (Ortmann 1919) (Table 1). For the tidal freshwater Potomac River mussel species, the most commonly used substrates are sand (84%) and gravel (74%), while others prefer fine substrates like mud (58%) and clay (16%) (Table 1). Most mussel species in the tidal freshwater Potomac River are habitat generalists and are found in a variety of substrates, but others, such as *Ligumia nasuta* that prefers soft, fine substrates and *Alasmidonta marginata* which are most common in coarse or fine gravel, are more selective (Ortmann 1919; Swartz and Nedeau 2007) (Table 1). Substrate type can also be an indicator of where certain species can be found, which can be useful when trying to locate endangered or threatened species. For example, the genera of *Elliptio* and *Lampsilis* exhibit a preference for clay substrates (Strayer 1993; Bogan and Ashton 2016) and the genera *Alasmidonta* is commonly found in sand and gravel substrates (Swertz and Nadau 2007; Bogan and Ashton 2016).

Hydrology can impact freshwater mussel habitat preference as well, as the flow rates of the stream play a role in substrate availability and can be an important factor in determining the presence or absence of particular species (Allen and Vaughn 2010). Hydrology is the most important predictor for presence or absence of some tidal freshwater Potomac River genera such as *Alasmidonta*, *Lasmigona*, and *Strophitus* (Strayer 1993). Substrate availability during high flows has also been shown to be an important predictor for habitat suitability, and many species often prefer shallow pools or riffles with lower flow rates (e.g., the genera *Ligumia* and *Lasmigona*) (Ortmann 1919). However, even within a genus, there can be varied preference for stream flow rates. For example, *Elliptio angustata* is commonly found in substrates with higher flow rates, while other species of *Elliptio* prefer slower moving water such as in ponds or pools (Bogan and Alderman 2004).

Bivalve water preference can also be impacted by physiological constraints. For example, the invasive clam *Corbicula fluminea* is usually restricted to larger streams or rivers with faster flowing water because it requires high levels of dissolved oxygen (Sousa et al. 2008). The mussel genera *Alasmidonta* and *Anodonta* have also been shown to be highly sensitive to changes in calcium levels, so calcium levels are a major predictor for presence for these groups (Strayer 1993). Additionally, other abiotic factors, such as salinity inundation, can impact habitat suitability for mussels as is the case for *Elliptio complanata*, *Lampsilis cariosa*, *Leptodea ochracea*, *Ligumia nasuta*, *Pyganodon cataracta*, and *Strophitus undulatus* which have a salinity tolerance range of zero, whereas *Corbicula fluminea* can tolerate mild salinity (0-2 ppt) (Kreeger and Krauter 2010; Najjar 2015).

Conservation status

Mussels face many threats in their freshwater habitats across the United States, and, of the species found in the tidal freshwater Potomac River, 63% (N=12) are listed as endangered, threatened or of special concern at the state level in Virginia or Maryland (Table 2). The importance of mussels in freshwater ecosystems and the decline in populations is of special concern to conservationists (Howard et al. 2004). There are several major issues contributing to the decline of mussels nationwide such as habitat loss and fragmentation, pollution and eutrophication, loss of host fish, climate change, and invasive species, and many of these are intricately interconnected (Bogan 1993; Lopes-lima et al. 2016).

Habitat loss and fragmentation occur nationwide as streams and rivers get diverted or artificial impoundments and dams are constructed, which all serve to limit the dispersal of mussels and restrict the range where they are found. This also prevents recolonization in suitable mussel habitats after a severe disturbance and impacts gene flow at the metapopulation level (Geist and Kuehn 2005; Haag 2012). Human alteration of the streams, such as through the construction of dams, can alter the substrate, flow and temperature and make water unsuitable for mussels that are sensitive to any of these changes (Vaughn and Taylor 1999). For this reason, some of these species are used as indicator species to determine water and environmental quality. For example, because *Corbiucla fluminea* and *Elliptio complanata* are much more tolerant to pollution and environmental change, they are often the only species that can survive in the most altered habitats. Therefore, the Chesapeake Bay Benthic Monitoring Program uses them as indicator species for their surveys (Llanso and Dauer 2002).

Pollution and eutrophication of freshwater ecosystems is another factor impacting mussels nationwide and in the tidal freshwater Potomac River, especially. For example, recent work has shown high levels of pharmaceuticals and herbicides in the Potomac and Anacostia Rivers due to inefficient wastewater treatment (Foster and Cui 2008; Hwang and Foster 2008; Shala and Foster 2010; Huff and Foster 2011). The glochidia larvae are more sensitive to pollution than the adults; therefore, eutrophication can prevent recruitment and colonization of new individuals and new populations to highly polluted systems, leading to functional extinction of many species (Bogan 1993). In addition, specific pollutants can impact physiological processes in mussels. For example, road deicing salts can alter the filtering behavior of mussels, and heavy metals can impair shell development (Pynnönen 1995; Hartmann et al. 2015).

In addition to the direct impact of habitat loss and eutrophication on mussels, these two issues further indirectly impact the distribution and abundance of mussels as they alter host fish abundance. For example, structures such as dams can impact the distribution of host fish which could negatively impact some specialist mussel species that can use only specific fish species as hosts (Douda et al 2013). Alternatively, if the water quality is degraded enough that it becomes unsuitable for the host fish, this will lead to severe depletion of mussels in that area (Bogan 1993). In the Potomac River, fish host limitation is a problem that could impact mussels that are specialists such as the genera *Leptodea* and *Lasmigona* (Lopes-lima et al. 2015).

Another significant cause of freshwater mussel decline in the tidal freshwater Potomac River is the introduction of invasive species such as the Asian clam *Corbicula fluminea*. In invaded habitats, *C. fluminea* directly competes with native mussels for substrate space but does not compete for fish hosts since *C. fluminea* does not have larvae requiring a host (Sousa et al. 2008). *Corbicula fluminea* has a relatively short lifespan, fast sexual maturity and high fecundity making it an aggressive competitor of native species in most places where it is found, however some native populations do coexist (Vaughn and Spooner 2006). For example, 2016 surveys in Gunston Cove documented the coexistence of *C. fluminea* with juvenile fingernail clams (Veneroida: Sphaeriidae) (Jones et al., in prep). Due to the filter feeding behavior of bivalves, invasive bivalves have the potential to play a large role in nutrient cycling in freshwater systems. Indeed, *C. fluminea* can cause a wide variety of habitat changes where it is found as it consumes suspended particles, alters the ecosystem engineering, and bioaccumulates pollutants in the ecosystem (Sousa et al. 2014). In addition, the introduction of *C. fluminea* has the potential to introduce new parasites and diseases to native bivalve species (Sousa et al. 2008).

Sampling design

There is a wide variety of sampling methods for collecting freshwater mussels, but the two common sampling designs include timed qualitative searches (presence/absence) and quantitative quadrat sampling (density estimates). Both approaches have drawbacks, as timed searches tend to underestimate small and buried species and quadrat searches can underestimate rare species and the total number of species (Vaughn et al. 1997). A combination of both methods should be applied whenever possible. Due to the drawbacks for each type of sampling method, the objective of the sampling should be considered when assessing which method to use (Metcalfe-Smith et al. 2000).

Because of the critical conservation status of many species in the tidal freshwater Potomac River, a qualitative approach would be valuable for assessing the distribution of some of the rare and endangered species for monitoring. When performing a qualitative approach, it is important to assess all available habitat types, including pools, runs, riffles, backwater areas, etc. (Cummings and Bogan 2006). In addition, areas where dead shells are commonly observed should also be included in the assessment as that is usually indicative of a nearby population.

After we have an idea of the distribution of different freshwater mussel species, quantitative assessments could be used for tracking populations of stable species such as the invasive *C. fluminea.* In deeper waters of the Potomac River, this could be accomplished using a brail, which is an 8 to 16 foot long bar that has 100 thin pieces of rope or chains hanging from it. Each of the pieces of rope or chain has four to six heavy gauge wire hooks. As the brail is pulled downstream, the wire hooks drag along the sediment, hold the bar off the bottom, and serve as irritants to any mussels that come in contact with the wire. The mussel will close its valves on the wire, and as the brail continues to be pulled, the mussel is pulled from the substrate. In shallower waters, the same technique can be employed using rakes and dredges. Due to the endangered and threatened status of many of these species, all freshwater mussels collected should be released back into the wild at the same location. The best time of year to find mussels is the late summer or early fall due to the rivers and streams being at the lowest levels (Cummings and Bogan 2006).

A similar study completed in 2012 in the upper reaches of the Potomac River in Paw Paw, West Virginia and Mason Island, Maryland is a good example of how we could design a sampling program in the future. In part one of the two part study, researchers used qualitative surveys to map major mussel habitat types within the river (Cummins 2012). Cummins (2012) documented six species (N=61) near the Potomac River's Dam #5 in 2010, including two endangered species (*Alasmidonta varicosa* and *Lasmigona subviridis*). In the second part of the study, targeted quantitative surveys were performed in July and August and consisted of randomly selected sites within each reach of the river, after which canoes were used to reach the location. Timed visual and excavation mussel searches were performed in 0.25 m^2 quadrats down to 15cm. Cummins (2012) did not find any mussels at the Paw Paw location and only two species and very few individuals at Mason Island (*Elliptio complanata* and *Lampsilis sp.*). While the Paw Paw reach had high quality mussel habitat, low water quality during the middle and late 20th century led to the extirpation of the freshwater mussels, and they have not returned (Cummins 2012). Current assessments in this portion of the Potomac River are further indications that the diversity and abundance of these important species is low. Further surveys and monitoring are required to determine if mussel populations show any signs of positive change.

Conclusion

Because of the ecological importance of freshwater mussels, high priorities have been set to conserve Virginia's native freshwater mussel species. Continued monitoring and assessments of the distribution of freshwater mussel species are needed to establish baseline data on the population structure, abundance, and species diversity. Any future efforts to conserve freshwater mussel populations could be compared to these baseline values to document positive change. We have outlined the current and extirpated species in the tidal freshwater Potomac River and highlighted the biology and conservation status of the various species. We have also reviewed various factors impacting freshwater mussels and outlined sampling methods for mussels in the tidal freshwater Potomac River.

Recommendations

We have collected a few mussels (<5) each year in the regular Gunston Cove sampling which included 20-30 ponar grabs per year. Given the tedious nature and limited scope of ponar grab sampling, we recommend that brail sampling be tested. We propose to construct a brail sampler following literature descriptions and spend 2-3 days on the tidal Potomac testing it in the coming year to determine its effectiveness. Based on those results we will recommend future sampling efforts.

Literature Cited

Allen, D.C., Vaughn, C.C. 2010. "Complex hydraulic and substrate variables limit freshwater mussel species richness and abundance." *Journal of the North American Benthological Society* 29(2): 383-394.

Arey, L.B. 1932. "The formation and structure of the glochidial cyst." *Biological Bulletin* 62(2): 212-221.

Baueg, G., Wachtler, K. 2004. *Ecology and evolution of the freshwater mussels Unionida.* Berlin: Springer-Verlag.

Bloodsworth, K.H., Bosman, B.R., Sietman, B.E., Hove, M.C. 2013. "Host fishes and conservation status of Alasmidonta marginata (Bivalvia: Unionidae) in Minesota." *Northeastern Naturalist* 20: 49-68.

Bogan, A.E. 1993. "Freshwater bivalve extinctions (Mollusca: Unionida): A search for causes." *American Zoology* 33: 599-609.

Bogan, A.E. 2008. "Global diversity of freshwater mussels (Mollusca: Bivalvia) in freshwater." *Hydrobiologia* 595: 139-147.

Bogan, A.E., Alderman, J.M. 2004. "Workbook and key to the freshwater bivalves of South Carolina."

Bogan, A.E., Ashton, M. 2016. "A manual of the freshwater bivalves of Maryland." *Maryland Department of Natural Resources Resource Assessment Service.*

Bogan, A.E., Raley, M., Levine, J. 2003. "Determination of the systematic position and relationships of the Lanceolate Elliptio complex (Mollusca: Bivalvia: Unionidae) with distributions in North and South Carolina." *United States FIsh and Wildlife Service.*

Boltovskoy, D, Cataldo, D.H. 1999. "Population dynamics of Limnoperna fortunei, an invasive fouling mollusc in the lower Parana River (Argentina)." *Biofould* 14:255-263.

Chazal, A.C., Roble, S.M. 2011. "Decline of the Freshwater Mussel Fauna of the North Fork and South Fork Shenandoah Rivers, Virginia." *Banisteria* 38: 38-59.

Cummings, K.S., Bogan, A.E. 2006. "Chapter 25: Unionida: Freshwater Mussels." In *The Mollusks: A Guide to Their Study, Collection, and Preservation*, by C.F., Pearce, T.A., Valdes, A. Sturm, 313-325. American Malacological Society.

Cummins, J.D. 2012. *Freshwater Mussel Survey of the Potomac Mainstem 2012 Summary Report for U.S. E.P.A.* The Interstate Commission on the Potomac River Basin.

Darrigran, G. 2002. "Potential impact of filter-feeding invaders on temperate inland freshwater environments." *Biomedical and life sciences* 4)1-2): 145-156.

Doherty, F.G., Cherry, D.S., Carins Jr., J. 1987. "Valve closure responses of the Asiatic clam Corbicula fluminea exposed to cadmium and zinc. ." *Hydrobiologia* 153:159-167.

Dresler, P.V., Cory, R.L. 1980. "The Asiatic Clam, Corbicula fluminea (Müller), in the Tidal Potomac River, Maryland." *Coastal and Estuarine Research Federation* 3(2): 150-151.

Eisler, R. 1988. "Arsenic hazards to fish, wildlife, and invertebrates. A synoptic review." *Department of the Interior, Fish and Wildlife. Biological Report* 85 (1.12) 92 pp.

Foster, G.D., Cui, V. 2008. "PAHs and PCBs deposited in surficial sediments along a rural to urban transect in a Mid-Atlantic coastal river based (USA)." *Journal of environmental science and health Part A: Toxic/Hazardous Substances and Environmental engineering* 43:1333.

Geist, J., Kuehn, R. 2005. "Genetic diversity and differentiation of central European freshwater pearl mussel (Margaritifera margaretifera L.) populations: implcations for conservation and management." *Molecular ecology* 14(2) 425-439.

Haag, W.R. 2012. *North American freshwater mussels natural history, ecology, and conservation.* New York NY: Cambridge University Press.

Haag, W.R. 2013. "The role of fecundity and reproductive effort in defining life-history strategies of North American freshwater mussels." *Biological Reviews* 88: 745-766.

Hartmann, J.T., Beggel, S., Auerswald, K., Stoeckle, B.C., Geist, J. 2015. "Establishing mussel behavior as a biomarker in ecotoxicology." *Aquatic Toxicology* 170: 279-288.

Howard, J.K., Furnish, J.L., Brim Box, J., Jepsen, S. 2015. "The decline of native freshwater mussels (Bivalvia: Unionoida) in California as determined from historical and current surveys." *California Fish and Game* 101(1): 8-23.

Huff, T., Foster, G.D. 2011. "Parts-per-trillion LC-MS(Q) analysis of herbicides and transformation products in water." *Journal of Environmental Science and Health* B46: 1-12.

Hwang, H., Foster, G.D. 2008. "Polychlorinated biphenyls in stormwater runoff entering the tidal Anacostia River, Washington DC, through small urban catchments and combined sewer outfalls." *Journal of Environmental Science and Health, Part A: Toxic/Hazardous Substances and Environmental Engineering* 43:567.

Johnson, R.I. 1970. "The systematics and zoogeographyof the Unionidae (Mollusca: Bivalvia) of the Southern Atlantic slope region." *Bulletin of the Museum of Comp. Zoology.* 140(6): 263-450.

Kreeger, D., Kraeuter, J. 2010. *Appendix N: Ecologically significant bivalve mollusks of the.* Wilmington, Delaware: PDE Report No. 10-01.

LeFevre, G., Curtis, W.C. 1911. "Metamorphosis without parasitism in the Unionidae." *Science* 33: 863-865.

Lopes-Lima, M., Sousa, R., Geist, J., Aldridge, D. C., Araujo, R., Bergengren, J., Bespalaya, Y., Bódis, E., Burlakova, L., Van Damme, D., Douda, K., Froufe, E., Georgiev, D., Gumpinger, C., Karatayev, A., Kebapçi, Ü., Killeen, I., Lajtner, J., Larsen, B. 2016. "Conservation status of freshwater mussels in Europe: state of the art and future challenges." *Biology Reviews* 92: 572-607.

McMahon, R.F. 2000. "Invasive characteristics of the freshwater bivalve Corbicula fluminea in nonindigenous freshwater organisms." *Vectors, Biology, and Impacts* 315-343.

Metcalfe-Smith, J.L., Di Maio, J., Staton, S.K., Mackie, G.L. 2000. "Effect of sampling effort on the efficiency of the timed search method for sampling freshwater mussel communities." *Journal of the North American Benthological Society* 19(4): 725-732.

Michaelson, D.L., Neves, R.J. 1995. "Life history and habitat of the endangered dwarf wedgemussel Alasmidonta heterodon (Bivalvia: Unionidae)." *Journal of the North American Benthological Society* 14: 324-340.

Najjar, R.G. 2015. *Potential impact of climate change on bivalves in the tidal freshwater.* Pennsylvania Sea Grant.

Ortmann, A.E. 1919. "A monograph on the naiades of Pennsylvania. Part 3. Systematic account of the genera and species. ." *Memoirs of the Carnegie Museum* 8(1)xiv, 384 pp.

Price, J.E., Eads, CS.B. 2011. "Brooding patterns in three freshwater musels of the genus Elliptio in the Broad River in South Carolina." *American Malacological Bulletin* 29: 121-126.

Pynnönen, K. 1995. "Effect of pH, hardness and maternal pre-exposure on the toxicity of Cd, Cu and Zn to the glochidial larvae of a freshwater clam Anodonta cygnea." *Water Research* 29: 247-254.

Ragnarsson, S.A., Raffaelli, D. 1999. "Effect of the mussel Mytilus edilus on the invertebrate fauna of sediments." *Journal of Experimental Marine Biology and Ecology* 241: 31-43.

Shala, L., Foster, G.D. 2010. "Surface water concentrations and loading budgets of pharmaceutical and other domestic-use chemicals in an urban watershed (Washington DC, USA)." *Archives of Environmental Contamination and Toxicology* 58: 551-561.

Sousa, R., Antunes, C., Guilhermino, L. 2008. "Ecology of the invasive Asian clam Corbicula fluminea (Muller 1774) in aquatic ecosystems: an overview." *Ann. Limn. Int. J. Lim.* 44(2): 85-94.

Sousa, R., Novais, A., Costa, R. 2013. "Invasive bivalves in fresh waters: impacts from individuals to ecosystems and possible control strategies." *Hydrobiologia.*

Strayer, D.L., Ralley, J. 1993. "Microhabitat use by an assemblage of stream-dwelling unionaceans (Bivalvia), including two rare species of Alasmidonta." *Journal of the North American Benthological Society* 12:247-258.

Swertz, B.I., Nedeau, E. 2007. "Freshwater mussel assessment." *Maine Department of Inalnd Fisheries Wildlife Division. Resource Assessment Section.*

Swertz, B.I., Nedeau, E. n.d. "Freshwater Mussel Assessment." *Maine Department of Inland Fisheries* .

van snik Gray, E., Lellis, W., Cole, J.C., Johnson, C.S. 2002. "Host identification for Strophitus undulatus (Bivalvia: Unionidae), the creeper, in the upper Susquehanna River basin, Pennsylvania." *The American Midland Naturalist* 147: 153-161.

Vaughn, C.C., Spooner, D.E. 2006. "Scale-dependent associations between native freshwater mussels and invasive Corbicula." *Hydrobiologia* 568: 331-339.

Vaughn, C.C., Taylor, C.M. 1999. "Impoundments and the decline of freshwater mussels: a case study of an extinction gradient." *Conservation Biology* 13(4): 912-920.

Vaughn, C.C., Taylor, C.M., Eberhard, K.J. 1997. "A comparison of the effectiveness of timed searches vs. quadrat searches." *Conservation and Management of Freshwater Mussels II: Proceedings of a UMRCC Symposium* 1-11.

Watters, G.T., O'Dee, S.H. 1999. "Glochidia of the freshwater mussel Lampsilis overwintering on fish hosts." *Journal of Molluscan Studies* 65: 453-459.

Table 1. Life history data and habitat preferences of native freshwater mussels and an invasive clam species in the tidal freshwater Potomac River based on a literature review. (1) Bogan and Ashton 2016; (2) Michaelson and Neves 1995; (3) Ortmann 1919; (4) Bloodsworth et al. 2013; (5) Swartz and Nedeau 2007; (6) Johnson 1970; (7) Price and Eads 2011; (8) Bogan and Aldermann 2004; (9) LeFevre and Curtis 1912; (10) Watters 1999; (11) van Snik Grey et al. 2002; (12) Sousa et al. 2008

Species	Common Name	Reproductive Mode	Larval Type	Brooding Period	Glochidia Release	Host Fish	Habitat
Alasmidonta heterodon (Lea, 1829)	dwarf wedge mussel	Bradytictic	glochidia	September to April **(2)**	April to June **(2)**	*Etheostoma nigru, Ethostoma olmstedi, Cottus bairdi, Salmo salar, Salmo trutta, Fundulus diaphanus, Morone saxatalis, Percina peltata* **(1)**	Sand-gravel substrates, found in shallow runs in small to moderate streams **(1)**
Alasmidonta marginata (Say, 1818)	elktoe	Bradytictic	glochidia	August to April **(3,4)**	April to July **(3,4)**	*Fundulus diaphanus, Cottus carolinae, Fundulus olivaceous, Luxilus zonatus, Culea inconstans, Semolitus atromaculatus, Erimyzon oblongus, Moxostoma erythrurum, Notimegnus crysoleucas, Rhinichthys cataractae, Gambusia affinis, Cottus bairdii, Hypentelium nigricans, Fundulus catenatus, Cyprinus carpio, Ambloplites rupestris, Moxostoma macrolepidotum, Moxostoma anisurum, Cottus cognatus, Ictiobus bubalus, Luxilus chrysocephalus, Lepomis gulosus, Catostomus commersoni,* **(1)**	Coarse or fine gravel substrate, found mostly in riffles **(3)**
Alasmidonta undulata (Say, 1817)	triangle floater	Bradytictic	glochidia	July to April **(3,5)**	April to June **(3,5)**	*Percina maculata, Campostoma anomalum, Luxilus cornutus, Semotilus corporalis, Etheostoma flabellare, Micropterus salmoides, Rhinichthys cataracte, Hypentelium nigricans, Lepomis gibossus, Notropis rubellus, Cottus cognatus, Morone americana* **(1)**	Coarse or fine gravel with sand and mud substrates (**3**) in large streams or rivers, but also found in lakes and ponds at low abundance (**5**) Does not favor riffles **(3)**
Alasmidonta varicosa (Lamarck, 1819)	brook floater	Bradytictic	glochidia	August to May **(5)**	April to June **(5)**	*Notemigonus chrysoleucas, Cottus bairdi, Cottus cognatus, Etheostoma flabellare, Etheostoma nigrum, Lepomis auritus, Lepomis gibossus, Lepomis microchirus, Luxilus albeolus, Noturus insignis, Perca flavescens, Percina crassa, Rhinichthys cataracte, Rhinichthys atratulus* **(1)**	Coarse sand and gravel substrate in flowing water habitats from small streams to large rivers **(3)**. Frequently found in streams with low calcium levels and low nutrients **(5)**
Anodonta implicata (Say, 1829)	alewife floater	Bradytictic	glochidia	September to May **(3)**	April to September (5)	*Alosa pseudoharengus, Lepomis gibossus, Morone americana, Catastomus commersoni, Gasterosteus aculeatus, Alosa aestivalis* **(1)**	Silt, sand and gravel substrates in streams, rivers and lakes **(5)**
Elliptio complanata (Lightfoot, 1786)	eastern elliptio	Bradytictic	glochidia	April to July **(3,5)**	June to August **(3,5)**	*Alosa pseudoharengus, Anguilla rostrata, Fundulus diaphanous, Lepomis cyanellus, Micropterus salmoides, Lepomis gibbosus, Lepomis humilis, Lepomis auritus, Micropterus dolomieu, Pomoxis annularis, Morone americana, Perca flavascens* **(1)**	Variety of substrates (clay, mud, sand, gravel, and cobble) in small streams, large rivers, freshwater tidal rivers and ponds and lakes **(5)**
Elliptio lanceolota (Lea, 1828)	yellow lance	Tachytictic	glochidia	May to June **(3)**	unknown	unknown **(1)**	Mud, sand, and gravel substrates and among rocks and mud where current is not too swift **(3)**
Elliptio producta (Conrad, 1836)	Atlantic spike	Tachytictic	glochidia	May to June **(3,6)**	unknown	unknown **(1)**	Mud or clay substrates in runs and pools near the stream bank. Fairly abundant in coastal streams **(1)**
Elliptio fisheriana (Lea, 1838)	northern lance	Tachytictic	glochidia	May to June **(3)**	unknown	*Lepomis macrochirus, Lepomis cyanellus, Etheostoma nigrum, Micropterus salmoides, Luxilus albeolus* **(1)**	Mud or clay substrates in runs and pools near the stream bank. Fairly abundant in coastal streams **(1)**
Elliptio angustata (Lea, 1831)	Carolina lance	Tachytictic	glochidia	March to June **(7)**	March to June **(7)**	unknown **(1)**	Coarse sand or gravel substrates in deep river habitats with a swift current **(8)**
Lampsilis cardium (Rafinesque, 1820)	plain pocketbook	Bradytictic	glochidia	August to Mid-July **(2,3)**	June to July **(2,3)**	*Lepomis macrochirus, Micropterus salmoides, Micropterus dolomieu, Pomoxis annularis, Stizostedion canadense, Stizostedion vitreum, Perca flavescens* **(1)**	Mud, sand, and gravel substrates in lakes, streams, and rivers **(3)**
Lampsilis cariosa (Say, 1817)	yellow lamp mussel	Bradytictic	glochidia	August to April **(9)**	April to June **(9)**	*Fundulus diaphanous, Esox niger, Micropterus salmoides, Micropterus dolomieu, Morone americana, Catostomus commersoni, Perca flavescens* **(1)**	Variety of substrates (silt, sand, gravel, and cobble) in medium to large rivers, including impounded areas, and lakes and ponds **(10)**
Lampsilis radiata (Gmelin, 1791)	eastern lamp mussel	Bradytictic	glochidia	August to April **(5)**	April to June **(5)**	*Fundulus diaphanous, Pomoxis nigromaculatus, Micropterus salmoides, Amblopites rupestris, Lepomis gibbosus, Micropterus dolomieu, Morone americana, Perca flavescens* **(1)**	Variety of substrates but most commonly in sand or gravel in small streams, large rivers, ponds or lakes **(5)**
Lasmigona subviridis (Conrad, 1835)	green floater	Bradytictic; Hermaphroditic	glochidia	August to June **(5)**	June to July **(5)**	unknown **(1)**	Sand or gravel substrates in large rivers, small streams, often few in number. Prefers pools or eddies **(3)**
Leptodea ochracea (Say, 1817)	tidewater mucket	Bradytictic	glochidia	August to April **(1)**	April to June **(1)**	*Fundulus diaphanous, Morone americana* **(1)**	Mud, fine gravel or sand substrates in most tidal waters, such as estuaries, ponds, canals and ditches **(3)**
Ligumia nasuta (Say, 1817)	eastern pond mussel	Bradytictic	glochidia	August to July **(3)**	June to August **(3)**	*Perca flavescens* **(1)**	Sandy bottoms, quiet bodies of water like pools **(3)**
Pyganodon cataracta (Say, 1817)	eastern floater	Bradytictic	glochidia	July to April **(3,5)**	April to July **(3,5)**	*Cyprinus carpio, Lepomis giboosus, Ambloplites rupestris, Gasterosteus aculeatus, Catostomus commersoni* **(1)**	Sand or mud substrates in a variety of habitats including small streams, rivers, ponds or lakes. Prefers slow-moving water. Can tolerate deep silt substrates in deeper water of ponds and lakes **(5)**
Strophitus undulatus (Say, 1817)	creeper	Bradytictic	glochidia	August to April **(11)**	April to June **(11)**	*Etheostoma zonale,Ameriurus melas, Lepomis macrochirus, Pimephales notatus, Campostoma anomalum, Semotilus atromaculataus, Etheostoma flabellare, Micropterus salmoides, Lepomis megalotis, Rhinichthys cataractae, Etheostoma caeruleum, Ambloplites rupestris, Cyrpinella spiloptera, Stizostedion vitreum, Pomoxis annularis, Ameriurus natalis* **(1)**	Sand and gravel substrates in most rivers and large streams and headwaters **(1)**
Utterbackia imbecillis (Say, 1829)	paper pond shell	Bradytictic; Hermaphroditic	glochidia	June to May **(3)**	May to June **(3)**	*Fundulus diaphanous, Semotilus atromaculatus, Ambloplites rupestris, Lepomis macrochirus, Lepomis marginatus, Lepomis cyanellus, Lepomis megalotis, Lepomis gibbosus, Lepomis gulosus, Micropterus salmoides, Gambusia affinis, Perca flavescens* **(1)**	Soft mud or sand in ponds, creeks, and banks of larger rivers **(6)**
Corbicula fluminea (Mueller, 1776)	Asian clam	Hermaphroditic	direct development	Spring to early summer and summer to early autumn **(12)**	NA	not required **(1)**	Sand and mud substrates in ponds, lakes, canals, and reservoirs **(1)**. Intolerant to high salinity and even moderate hypoxia. Prefers areas with high organic matter content **(12)**

Table 2. Conservation status of historical and current freshwater mussels and an invasive clam species in the tidal freshwater Potomac River in Maryland and Virginia(1) Maryland Department of Natural Resources (Bogan and Ashton 2016) (2) Virginia Department of Game and Inland Fisheries (2015 Virginia Wildlife Action Plan)

Species	***Common Name***	***Historical or Current***	***Native or Introduced***	***Conservation Status in MD (1) and VA (2)***
Alasmidonta heterodon (Lea, 1829)	dwarf wedge mussel	Historical	Native	Endangered (**1**) Endangered **(2)**
Alasmidonta marginata (Say, 1818)	elktoe	Unknown	Native	N/A (**1**) Special concern, unsure if extirpated (**2**)
Alasmidonta undulata (Say, 1817)	triangle floater	Current	Native	Special concern, endangered (**1**) Moderate conservation need **(2)**
Alasmidonta varicosa (Lamarck, 1819)	brook floater	Current	Native	Threatened, Endangered (**1**) Critical conservation need **(2)**
Anodonta implicata (Say, 1829)	alewife floater	Current	Native	Currently stable (**1**) Moderate conservation need **(2)**
Elliptio complanata (Lightfoot, 1786)	eastern elliptio	Current	Native	Stable, secure (**1**) Moderate conservation need **(2)**
Elliptio lanceolota (Lea, 1828)	yellow lance	Unknown	Native	Endangered, unknown (**1**) Threatened **(2)**
Elliptio producta (Conrad, 1836)	Atlantic spike	Current	Native	Special concern, in need of conservation (**1**) Moderate conservation need **(2)**
Elliptio fisheriana (Lea, 1838)	northern lance	Current	Native	Special concern, watch list (**1**) Moderate conservation need **(2)**
Elliptio angustata (Lea, 1831)	Carolina lance	Historical	Native	N/A (**1**) Moderate conservation need **(2)**
Lampsilis cardium (Rafinesque, 1820)	plain pocketbook	Current	Native	Lower risk, near threatened (**1**) Moderate conservation need **(2)**
Lampsilis cariosa (Say, 1817)	yellow lamp mussel	Current	Native	Threatened, unknown (**1**) Threatened **(2)**
Lampsilis radiata (Gmelin, 1791)	eastern lamp mussel	Current	Native	Currently stable (**1**) Moderate conservation need **(2)**
Lasmigona subviridis (Conrad, 1835)	green floater	Current	Native	Threatened, endangered (**1**) Threatened **(2)**
Leptodea ochracea (Say, 1817)	tidewater mucket	Current	Native	Special concern, rare (**1**) Moderate conservation need **(2)**
Ligumia nasuta (Say, 1817)	eastern pondmussel	Current	Native	Threatened, endangered (**1**) Moderate conservation need **(2)**
Pyganodon cataracta (Say, 1817)	eastern floater	Current	Native	Currently stable, secure (**1**) N/A (**2**)
Strophitus undulatus (Say, 1817)	creeper	Current	Native	Currently stable, in need of conservation (**1**) Moderate conservation need **(2)**
Utterbackia imbecillis (Say, 1829)	paper pondshell	Current	Native	Currently stable, secure (**1**) N/A (**2**)
Corbicula fluminea (Mueller, 1776)	asian clam	Current	Introduced	Non-native, stable (**1**) **(2)**

Table 3. Synonyms of native freshwater mussels and an invasive clam species in the tidal freshwater Potomac River. Modified from Bogan and Ashton (2016).

Species	Common name	Synonymy
Alasmidonta heterodon (Lea, 1829)	dwarf wedge mussel	*Unio heterodon* (Lea, 1829) *Alasmidonta (Pressodonta) heterodon* (Lea, 1829)
Alasmidonta marginata (Say, 1818)	elktoe	Alasmidonta (Decurambis) marginata Say, 1818
		Mya rugulosa (Wood, 1828) *Alasmidonta (Decurambis) scriptum* (Rafinesque, 1831) *Unio swanaonensis* (Hanley, 1842) *Alasmidonta corrugate* (DeKay, 1843) *Marginata marginata "var. truncata"* (Wright, 1898) *Alasmidonta (Decurambis) marginata susquehannae* (Ortmann, 1919) *Alasmidonta marginata variabilis* (Baker, 1928)
Alasmidonta undulata (Say, 1817)	triangle floater	*Unio undulata* (Say, 1817) "*Unio glabratus?* Lamarck" (Sowerby, 1823) *Unio hians* (Valenciennes, 1827) *Alasmidonta sculptilis* (Say, 1829) *Uniopsis radiata* (Swainson, 1840) *Uniopsis mytiloides* (Swainson, 1840) *Margaritana triangulate* (Lea, 1858) *Unio swainsoni* (Sowerby, 1868)
Alasmidonta varicosa (Lamarck, 1819)	brook floater	*Unio varicosa* (Lamarck, 1819) *Alasmidonta corrugata* (DeKay, 1843) *Alasmidonta (Decurambis) varicosa* (Lamarck, 1819) *Mya rugulosa* (Wood, 1856)
Anodonta implicata (Say, 1829)	alewife floater	*Anodonta (Pyganodon) implicata* (Say, 1829)
		Anodonta newtonensis (Lea, 1836) *Anodonta housatonica* (Linsley, 1845)
Elliptio complanata (Lightfoot, 1786)	eastern elliptio	*Mya complanata Lightfoot, 1786* *Unio violaceus Spengler, 1793* *Unio purpureus Say, 1817* *Unio rarisulcata Lamarck, 1819* *Unio coarctata Lamarck, 1819* *Unio purpurascens Lamarck, 1819* *Unio rhombula Lamarck, 1819* *Unio carinifera Lamarck, 1819* *Unio georgina Lamarck, 1819* *Unio glabrata Lamarck, 1819* *Unio sulcidens Lamarck, 1819* *Unio virginiana Lamarck, 1819*

		Unio aurata Rafinesque, 1820
		Unio fluviatilis Green, 1827
		Mya rigida Wood, 1828
		Unio griffithianus Lea, 1834
		Unio complanatus subinflatus Conrad, 1835
		Unio jejunus Lea, 1838
		Unio fuliginosus Lea, 1845
		Unio cuvierianus Lea, 1852
		Unio errans Lea, 1856
		Unio vicinus Lea, 1856
		Unio geminus Lea, 1856
		Unio abbevillensis Lea, 1857
		Unio percoarctatus Lea, 1857
		Unio wheatleyi Lea, 1857
		Unio catawbensis Lea, 1861
		Unio insulsus Lea, 1857
		Unio spadiceus Lea, 1857
		Unio macer Lea, 1857
		Unio contractus Lea, 1857
		Unio virens Lea, 1857
		Unio savannahensis Lea, 1857
		Unio subflavuslea, 1857
		Unio fumatus Lea, 1857
		Unio subniger Lea, 1857
		Unio neusensis Lea, 1857
		Unio purus Lea, 1858
		Unio exactus Lea, 1858
		Unio pastelIii Lea, 1858
		Unio roswellensis Lea, 1859
		Unio burkensis Lea, 1859
		Unio hallenbeckii Lea, 1859
		Unio baldwinensis Lea, 1859
		Unio salebrosus Lea, 1859
		Unio raeensis Lea, 1859
		Unio latus Lea, 1859
		Unio quadratus Lea, 1859
		Unio squameus Lea, 1861
		Unio rostrum Lea, 1861
		Unio northamptonensis Lea, 1861
		Unio decumbens Lea, 1861
		Unio raleighensis Lea, 1863

		Unio aberrans Lea, 1863
		Unio weldonensis Lea, 1863
		Unio mecklenbergensis Lea, 1863
		Unio chathamensis Lea, 1863
		Unio gastonensis Lea, 1863
		Unio quadrilaterus Lea, 1863
		Unio indefinilus Lea, 1863
		Unio indefinitus (Lea, 1866)
		*Unio mediocris (*Lea, 1863)
		Unio perlucens (Lea, 1863)
		*Unio curatus (*Lea, 1863)
		*Unio protensus (*Lea, 1865)
		Unio lazarus (Sowerby, 1868)
		Unio beaverensis (Lea, 1868)
		Unio nubilus (Lea, 1868)
		Unio datus (Lea, 1868)
		Unio humerosus (Lea, 1868)
		Unio uhareensis (Lea, 1868)
		Unio tortuosus (Sowerby, 1868)
		Unio santeensis (Lea, 1871)
		Unio yadkinensis (Lea, 1872)
		Unio amplus (Lea, 1872)
		Unio ligatus (Lea, 1872)
		Unio differtus (Lea, 1872)
		Unio subparallelus (Lea, 1872)
		Unio oblongus (Lea, 1872)
		Unio curvatus (Lea, 1872)
		Unio irwinensis (Lea, 1872)
		Unio subsquamosus (Lea, 1872)
		Unio infuscus (Lea, 1872)
		Unio ratus (Lea, 1872)
		Unio basalis (Lea, 1872)
		Unio dissimilis (Lea, 1872)
		*Unio cirratus (*Lea, 1874)
		Unio subolivaceus (Lea, 1874)
		Unio infulgens (Lea, 1874)
		Unio corneus (Lea, 1874)
		Unio dooleyensis (Lea, 1874)
		Unio gesnerii (Lea, 1874)
		Unio invenustus (Lea, 1874)
		Unio (Arconaia) provancheriana (Pilsbry, 1890)

		Unio palliatus (Simpson, 1900) *Unio pullatus majusculus* (De Gregorio, 1914) *Unio complanatus mainensis* (Rich, 1915)
Elliptio lanceolota (Lea, 1828)	yellow lance	*Unio lanceolatus Lea, 1828* *Unio duttonianus Lea, 1841* *Unio sagittformis Lea, 1852* *Unio rostraeformis Lea, 1856* *Unio rostriformis Lea, 1856* *Unio emmonsii Lea, 1857* *Unio naviculoides Lea, 1857* *Unio hazelhurstianus Lea, 1858* *Unio viridulus Lea, 1863* *Unio haslehurstianus Sowerby, 1866* *Margaron (Unio) hazlehurstianus Lea, 1859* *Margaron (Unio) sagittaeformis Lea, 1870* *Unio rostreformis de Gregorio, 1914* *Unio arctior var. fisheropsis de Gregorio, 1914*
Elliptio producta (Conrad, 1836)	Atlantic spike	*Unio productus Conrad 1836* *Unio nasutus Conrad 1838* *Unio barrotti Küester 1861* *Unio nasutidus Lea 1863* *Unio nasutulus Lea 1863* *Unio nasutilus Simpson 1900*
Elliptio fisheriana (Lea, 1838)	northern lance	*Unio fisherianus Lea, 1838* *Margarita (Unio) fisherianus (Lea, 1838)*
Elliptio angustata (Lea, 1831)	Carolina lance	None
Lampsilis cardium (Rafinesque, 1820)	plain pocketbook	*Lampsilis ovata ventricosa (Barnes, 1823)* *Unio ventricosus Barnes, 1823* *Unio occidens Call, 1887* *Lampsilis ventricosus (Barnes, 1823)* *Lampsilis ovata ventricosa (Barnes, 1823)* *Unio occidens Lea, 1829* *Unio subovatus Lea, 1831* *Unio Ienis Conrad, 1838* *Unio canadensis Lea, 1857* *Unio latissimus Sowerby, 1868* *Lampsilis ventricosa var. lurida Simpson, 1914* *Lampsilis ventricosa cohongoronta Ortmann, 1912* *Lampsilis ventricosa winnebagoensis Baker, 1928* *Lampsilis ventricosa pergloboas Baker, 1928*

Lampsilis cariosa (Say, 1817)	yellow lampmussel	*Unio cariosus Say, 1817* *Margarita (Unio) cariosus (Say, 1817)* *Lampsilis pallida Rafinesque, 1820* *Unio ovata Valenciennes, 1827* *Unio viridis Ferussac, 1835* *Unio crocatus Lea, 1841* *Unio oratus Conrad, 1849*
Lampsilis radiata (Gmelin, 1791)	eastern lampmussel	*Mya radiata Gmelin, 1791* *Unio luteola Lamarck, 1819* *Unio lineata 'Valenciennes' Bory de St. Vincent, 1827* *Unio tenebrosus Conrad, 1834* *Unio melinus Conrad, 1838* *Unio boydianus Lea, 1840* *Unio rosaceus De Kay, 1843* *Mya oblongata Wood, 1856* *Unio elongata S.G. Goodrich, 1858* *Unio obliquiradiatus Reeve, 1865* *Unio conspicuus Lea, 1872* *Unio virginiana Simpson, 1900* *Lampsilis radiata (Gmelin, 1791)* *Lampsilis radiata oneidensis Baker, 1916* *Unio virginea Frierson, 1927* *Lampsilis radiata var. conspicua (Lea, 1872)* *Lampsilis (Lampsilis) radiata radiata (Gmelin, 1791)*
Lasmigona subviridis (Conrad, 1835)	green floater	*Unio subviridis Conrad, 1835* *Unio tappanianus Lea, 1838* *Unio hyalinus Lea, 1845* *Margaritana quadrata Lea, 1861* *Unio pertenius Lea, 1863*
Leptodea ochracea (Say, 1817)	tidewater mucket	*Mytilus fluviatilis Gmelin 1791 (Nomen dubium)* *Unio ochraceus Say, 1817* *Lampsilis rosea Rafinesque, 1820* *Unio rosaceus Conrad, 1849* *Lampsilis ochracea (Say, 1817)* *Lampsilis (Lampsilis) ochracea (Say, 1817)*
Ligumia nasuta (Say, 1817)	eastern pondmussel	*Unio nasutus Say, 1817* *Obliquaria attenuata Rafinesque, 1820* *Unio rostrata Valenciennes, 1827* *Unio vaughanianus Sowerby, 1868* *Unio fisherianus Kuester, 1860 non Lea, 1838* *Lampsilis nasuta (Say, 1817)*

		Eurynia nasuta (Say, 1817)
Pyganodon cataracta (Say, 1817)	eastern floater	*Anodonta cataracta Say, 1817* *Anodonta marginata Say, 1817* *Anodonta teres Conrad, 1834* *Anodon excurvata De Kay, 1843* *Anodonta virgulata Lea, 1857* *Anodonta lacustris Lea, 1857* *Anodonta hallenbeckii Lea, 1858* *Anodonta gesnerii Lea, 1858* *Anodonta dariensis Lea, 1858* *Anodonta williamsii Lea, 1862* *Anodonta tryoni Lea, 1862* *Anodonta dolearis Lea, 1863* *Anodonta doliaris Lea, 1866* *Anodonta (Pyganodon) cataracta cataracta Say, 1817*
Strophitus undulatus (Say, 1817)	creeper	*Anodonta undulata Say, 1817* *Alasmodonta edentula Say, 1820* *Anodon rugosus Swainson, 1822* *Anodonta edentula (Say, 1817)* *Strophitus edentulus (Say, 1817)* *Strophitus rugosus (Swainson)* *Anodonta pensylvanica [sic] Lamarck, 1819* *Anodon areolatus Swainson, 1829* *Alasmodonta edentula Say, 1829* *Anodonta virgata Conrad, 1836* *Anodonta pavonia Lea, 1836* *Anodonta wardiana Lea, 1838* *Anodon unadilla De Kay, 1843* *Anodonta tetragona Lea, 1845* *Anodonta arkansensis Lea, 1852* *Anodonta shaefferiana Lea, 1852* *Alasmodon rhombica Anthony, 1865* *Anodon papyracea Anthony, 1865* *Anodon annulatus Sowerby, 1867* *Anodon quadriplicatus Sowerby, 1867* *Anodonta salmonia Clessin, 1873* *Strophitus undulatus ovatus Frierson, 1927* *Strophitus rugosus pepinensis Baker, 1928* *Strophitus rugosus winnebagoeinsis Baker, 1928* *Strophitus rugosus lacustris Baker, 1928* *Strophitus edentulus (Say, 1817)*

Utterbackia imbecillis (Say, 1829)	paper pondshell	*Anodonta imbecillis Say, 1829* *Anodonta imbecilis [sic] Say, 1829* *Anodonta incert Lea, 1834* *Anodon horda Gould, 1855* *Anodonta henryana Lea, 1857* *Utterbackia imbecillis fusca Baker, 1927* *Anodonta ohiensis Rafinesque, 1820 [in part)* *Anodonta (Utterbackia) imbecilis* (Say, 1829)
Corbicula fluminea (Mueller, 1776)	Asian clam	None