ASSESSING THE EFFECTS OF SMALL BARRIERS IN THE HUDSON RIVER WATERSHED

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EXECUTIVE SUMMARY

During 2018, we studied the ecological conditions (instream habitat, riparian conditions, and benthic macroinvertebrates) surrounding three potential barrier removal sites and continued monitoring at two post-removal sites in the Hudson Estuary watershed. This year's data represents the third year of pre-removal monitoring at Brown's Pond dam (Otterkill), Maiden Lane Dam (Furnace Brook), and Annandale Dam (Sawkill). We collected Year-3 of post-removal data on the Troy Gate (Wynantskill) and Year-2 post-removal data on Shapp Pond Dam (E. Branch Wappingers Creek). Our existing data to date demonstrates that small dam removal efforts lead to rapid recovery of impacted stream segments. We observed significant improvements in stream substrate and habitat quality, macroinvertebrate community composition, and water quality designations. Comparatively, the same ecological metrics at our non-removed sites did not differ significantly across years of pre-removal data collection.

INTRODUCTION

Damming of tributaries to the Hudson River represents some of the earliest manipulation of streams within the watershed and has taken place for hundreds of years. European colonization during the late 18th century needed water-power to drive sawmills, tanneries, and expand industrial aspirations. Dam construction accelerated during the early 20th century when increasing demands arose for water supply, navigation, flood control, and hydropower (Swaney et al., 2006; Figure 1). There are roughly 1,500 state-documented dams and likely several hundred more un-documented dams in the HR watershed according to the New York State Department of Environmental Conservation's Division of Dam Safety's New York State Inventory of Dams. With an average age of 77 years (you can cite the NYS dam Inventory database) and a useful service life of 50 years, many dams in the basin, particularly mill and low-head dams (<7m), have outlived their original purposes and their structural stability continues to decline with age and lack of maintenance.

Nationally, over 1,200 dams have been removed since 1970 and the rate of removal has increased markedly in the last 20 years (Bellmore et al., 2017). Within the Hudson River watershed, the need to remove relic dams to mitigate flood risk and improve riverine habitat continuity and migratory fish access is becoming more widely recognized. In addition to increased flood risk and habitat fragmentation, dams can negatively alter natural riverine flow and sediment transport patterns, physiochemical processes, and biological attributes of impounded stream segments. Despite the increased recognition of the negative effects of relic dams, only four dams have been removed in the Hudson River Estuary watershed in recent years according to the American River's dam-removal database (www.americanrivers.org/DamRemovalDatabase). In addition, only one peer-reviewed study has documented the ecological, hydrologic and geomorphic effects of dam removal in NY (Bellmore et la., 2017).

This study aims to address this lack of information by assessing the local impact and recovery of planned and ongoing barrier removal efforts within the Hudson River watershed. Quality pre-removal datasets are scarce, as most efforts only collect one year of pre-removal data (Foley et al., 2017). Collecting multiple years of pre-removal data facilitates quantification of local temporal variability for both substrate composition and macroinvertebrate community assemblages. Such seasonal and annual variation is to be expected in any dynamic natural system. It is important to document 'baseline' site characteristics so that post-removal recovery trends can be assessed within the appropriate site-level context.

Similarly, documenting post-removal recovery is a multi-year process as well. The magnitudes of change within targeted ecological metrics can be compared to variability within the pre-removal data. Changes that significantly deviate from the pre-removal baseline variation can be attributed to the removal effort within the context of local site characteristics and landscape context variables (e.g. contiguous forest, road miles, percent impervious surface). The rate of these changes and how soon key ecological metrics recover will provide important information to resource managers as to the success of their efforts and lead to predictive tools to help guide future removal efforts.

STUDY SITES

Between 2016 – 2018 we sampled at five Hudson River tributary sites which possessed historic barriers: (1) Tidal Gate on the Wynantskill at Troy, NY in Rensselaer County (2) Annandale dam on the Sawkill at Bard College in Dutchess County, (3) Shapp Pond dam on the East Branch Wappingers Creek at Salt Point, NY in Dutchess County, (4) Browns Pond dam on the Otterkill near Maybrook, NY in Orange County, and (5) Maiden Ln dam on Furnace Brook at Croton-on-Hudson, NY, in Westchester County (Figure 2).

In May of 2016 the barrier on the Wynantskill was removed, providing this study three years of post-removal data at that site. In October 2016, the Shapp Pond dam was completely removed, providing this study 1 year of pre-removal data and 2 years of post-removal data. Barriers at Brown's Pond dam, Maiden Ln dam, and Annandale dam remained intact, providing this study with three years of pre-removal data.

At each site, we identified six sampling locations: three downstream locations at riffle habitats below the barrier, and three locations upstream of the barrier (Table 1). At each downstream location, we established a permanent transect spanning the bankfull width of the stream, monumented with pins. At each upstream location, we attempted to center the sampling point within the original stream channel.

METHODS

Physical Attributes

At each site location, we measured various physical parameters at the time of sampling. At the Wynantskill and at downstream locations at all other sites, we measured water/air temperature, stream depth, and stream wetted-width. We also recorded canopy cover, water clarity, stage height, weather conditions, macrophyte growth, animal observations (mammals, birds, amphibians), and human evidence (trash, fishing remnants). During 2016 and 2017 a site schematic was hand-drawn, identifying the locations of the sampling transects, local landmarks, and key trees for spatial orientation. In 2018 a schematic was drawn only if there were changes at the site. We measured stream-flow with a Marsh-McBirney FlowMate 2000 digital flow meter at each site within the downstream location sampling area.

At upstream locations with intact barriers, we recorded water temperature, water depth, water clarity, macrophyte growth, and animal observations at time of sampling. A site-schematic was drawn at each sampling location.

Macroinvertebrate Collections

At each site, we collected aquatic macroinvertebrate samples. Collection methodologies followed those outlined in the New York State Department of Environmental Conservation's *Standard Operating Procedure: Biological Monitoring of Surface Waters in New York State* (Smith et al., 2014). Kick samples entail disturbing the substrate with one's feet upstream of a D-style net to capture invertebrates in lotic environments. Ponar grab samples involve collecting a benthic sample using a grab-type sampler in lentic environments, and multiplate samplers involve deploying a series of hardboard plates acting as colonization surfaces suspended below a float in lentic waters. Kick and Ponar samples were collected during the first 10-days in June of each year. Multiplate samples were deployed in early June while collecting Ponar samples and set for 5-weeks, being retrieved in mid-July.

At the Wynantskill, we collected a kick-sample at each of the three upstream transect locations, for a total of three kick-samples at that site for each year 2016-2018. At Shapp Pond, we collected a kick samples at each downstream transect location for a total of three downstream samples for each year 2016-2018. At the Shapp Pond upstream locations, we collected three Ponar grab samples at each transect location in 2016 prior to barrier removal. During post-removal years 2017 and 2018, we collected a kick-sample at each upstream transect, as the habitat changed from lentic (2016) to lotic.

At our three intact sites: Browns Pond Dam, Annandale/Sawkill Dam, and Maiden Ln/Furnace Brook Dam, our sampling procedure was identical for collecting pre-removal data from 2016-2018. At each downstream transect location we collected a kick sample for a total of three kick samples at each site 2016-2018. At each upstream location we collected a Ponar grab sample and a multiplate sample.

Following collection, all macroinvertebrate samples were preserved in 70% ethanol and archived in the laboratory until sorting and identification commenced. From each sample, a random 100-ct subsample was extract for each of the kick and Ponar samples. A random 250-ct subsample was extracted from the multiplate samples. Subsampling procedures follow the methodologies in Smith, 2014. All organisms subsampled were enumerated and identified to the lowest practicable taxonomic level, often genus. Due to project timing and budgets, Chironomidae were kept at the family level, and not individually slide mounted for generic ID confirmations. All subsampled organisms were archived for quality control or verification purposes.

Substrate Composition / Stream Channel Morphology

At lotic habitat locations, we assessed substrate composition with three estimates. The first was a qualitative visual estimate of the percent substrate classes of bedrock, boulder, cobble, gravel, sand, and fines/silt across the wetted-width of the transect. The second estimate was a general qualitative measurement of substrate class using the point-touch method (Wollman, 1954) at 11 points along the transect: at each 10% increment of wetted-width and both edges (0%, 110% representing the edges). At each recording point, a blind touch was made to the substrate, and the substrate class was recorded.

The third estimate, beginning in 2017, was a comprehensive transect survey, where substrate was estimated at every 30cm increments from river-left bankfull pin to river-right bankfull pin. Since the transect beginning and end points are static, each 30cm recording position is also static across years, and the total number of measurements is consistent year-to-year. Since the 30-cm bankfull estimate provides the most comprehensive substrate measurements, we used those data to estimate the mean substrate particle size for each replicate across locations where the survey was performed.

At lentic habitat locations, we used the sediment captured in the Ponar grab during macroinvertebrate collections to estimate the substrate classes. We sieved sediment through a 500µm sieve. Substrate which passed through the sieve was identified as silt/clay/fines, while substrate remaining was visually tabulated into percent contributions.

In addition to the substrate assessments, we collected detailed stream channel morphology data across the bankfull-to-bankfull transects. We established a local benchmark unique to each transect so elevational data would be indexed to an identical reference elevation across years. Using standard rod and level surveying techniques, we measured the stream bottom elevation across the transect during 2017 and 2018 at the same 30cm interval substrate recording points. These data allow us to observe the changes in channel morphology post-barrier removal and identify areas of bed erosion or deposition. The elevational information at intact sites affords insight into annual sediment migration and shifting which occurred during the prior hydraulic year.

Biometrics and Water Quality

We assessed the ecological health and water quality of each site by combining data on aquatic invertebrate populations, substrate habitat attributes, and established biometric index valuations. Regarding macroinvertebrate assemblages, we calculated relative abundance, at the family taxonomic level, for each sampling method at each location. Regarding habitat metrics, we calculated the mean particle size – based on substrate class – for each transect (downstream) or sampling point (upstream). Particle size classifications were taken from Bevenger and King, 1995; and Harrelson et al., 1994.

We calculated six bio-index water quality values for each of our aquatic invertebrate collections: (1) Species Richness, (2) Hilsenhoff Biotic Index (HBI), (3) Ephemeroptera-Plecoptera-Trichoptera (EPT) Richness, (4) Percent Model Affinity (PMA), (5) Species Diversity, and (6) % Top-3 Dominance. Using the results from each of the seven bio-indices, a cumulative Biological Assessment Profile (BAP) score and water quality impact designations were calculated. Specific methodologies and calculation formulae for the bio-indices and BAP scores can be referenced in Smith et al., 2014.

Analyses

Pre and Post dam removal data from Shapp Pond was analyzed with a Two-Way ANOVA, testing the response of Species Richness, HBI, EPT, PMA, BAP, % Ephemeroptera, % Chironomidae, % Trichoptera, % Oligochaeta, and substrate particle size. To conform to assumptions of normality, the percent - 'Taxa' data was arc-sine transformed and the substrate particle size was Log-10 transformed. We tested each of the variables against location and year. If year was significant for a location, we then ran a One-Way ANOVA for that location to determine if the pre-removal year (2016) was different from either of the post-removal years (2017, 2018).

Data from the remaining sites were analyzed with a One-Way ANOVA testing the nine variables against Year since there was only pre-removal data at Browns Pond, Furnace Brook, and Sawkill; and Wynantskill only possessed post-removal data.

RESULTS

SHAPP POND

We found initial, pre-removal conditions at Shapp Pond to vary significantly between downstream and upstream locations. The impounded upstream location possessed significantly lower Species Richness ($P \leq 0.001$), EPT Richness ($P=0.0004$), and PMA ($P=0.0002$). We observed significantly higher HBI Index ($P=0.0015$) (Table 2, Figure 3). Water Quality BAP Score was significantly lower upstream than downstream ($P \leq 0.001$) (Figure 4). As to the invertebrate community composition, we observed significantly lower percent Ephemeroptera ($P=0.0336$), percent Trichoptera ($P=0.005$), and significantly higher percent Oligochaeta ($P=0.0005$) upstream; percent midges did not differ significantly between locations during 2016. Substrate particle size was significantly lower upstream than downstream ($P=0.0077$).

Following removal, we observed recovery patterns to differ by location. At the downstream location post-removal, we did not detect significant changes in Species Richness, HBI, EPT, PMA, BAP, percent Ephemeroptera, percent Chironomidae, or percent Trichoptera. The mean substrate particle size decreased downstream due to the post-removal flushing effect, but not significantly. We observed a significant decrease in percent Oligochaeta (P=0.0042) (Figure 5).

Upstream, the beneficial post-removal changes were more pronounced. Compared to our initial pre-removal data, we observed significant improvements during years of recovery for Species Richness (P≤0.001), EPT Richness (P≤0.001), PMA (P=0.0006), and HBI Index (P=0.0011). Water Quality BAP Score increased significantly post-removal (P≤0.001). Regarding the aquatic invertebrate community, we observed significant increases in percent Ephemeroptera (P=0.0056) and percent Trichoptera (P=0.0183), and significant decreases in Oligochaeta (P=0.0012) (Figure 6). Substrate particle size significantly increased during post recovery years (P=0.0053), such that we no longer detected a between upstream and downstream during 2017 and 2018 (Figure 7).

After a one-year period post-removal, we no longer observed significant differences between downstream and upstream locations for seven of our nine variables: Species Richness, HBI Index, EPT Richness, PMA, percent Ephemeroptera, percent Trichoptera, percent Oligochaeta, or substrate particle size, suggesting ecological and biological conditions are homogenizing between the two locations at Shapp Pond. We observed the Water Quality BAP score to remain significantly lower upstream than downstream in 2017 (P=0.0133) and 2018 (P=0.0056), but the differences between the two locations are shrinking over time. We observed significantly more percent Chironomidae upstream than downstream for both 2017 (P=0.0220) and 2018 (P=0.0244), due to opposing populations trends at each location. Downstream Chironomidae populations slightly declined post-removal, while upstream populations slightly increased (Figures 5 & 6).

<u>WYNANTSKILL</u>

During the three-year post-removal period, we observed mixed responses among the biometrics. For example, we observed significantly lower Species Richness (P=0.03) and EPT Richness (P=0.0309) over time (Figure 8). We also observed a significant increase in PMA (P=0.0375) and a non-significant increase in HBI Index (Table 3). Regarding the aquatic invertebrate community, we observed a significant decrease in percent Oligochaeta (P=0.0279) and a significant increase in percent Trichoptera (P=0.0007) (Figure 9). We observed a decline in Water Quality BAP score, but those trends were not significant. (Figure 10). Ephemeroptera abundance increased dramatically from 2016-2017, but then declined from 2017-2018 returning to approximate 2016 levels. Chironomidae populations remained relatively similar over time. Substrate particle size steadily increased over time, but the differences were not significant (Figure 11).

<u>BROWNS POND</u>

We observed clear differences in ecological and biological metrics between downstream and upstream locations at Browns Pond. However, within each location over the three-year pre-removal study period our metrics remained similar (Table 4). Browns Pond remains a moderately to severely impacted system (Figure 12). We observed the presence of mayflies at the downstream location for the first time in 2018, however, other taxa did not show any differences over time. Upstream, we observed significant declines in Chironomidae populations (P=0.0253) while Oligochaeta populations significantly increased (P=0.0289) (Figure 13). Within each location, substrate particle size did not differ significantly over time; downstream was dominated by large substrate classes and upstream is uniformly silt (Figure 14).

SAWKILL

We observed clear differences between downstream and upstream locations at Annandale Dam. However, we did not observe any significant differences in our metrics over the three-year pre-removal study period (Table 5). The downstream reach at Annandale dam possesses decent water quality with only a semi-impacted designation while the upstream impoundment degrades to a moderately impacted system (Figure 15). Within each location, substrate particle size did not differ significantly over time; downstream consists of a mostly cobble substrate and upstream is uniformly silt (Figure 16).

FURNACE BROOK

We observed clear differences in ecological and biological metrics between downstream and upstream locations at Maiden Ln Dam. However, within each location over the three-year pre-removal study period our metrics remained similar (Table 6). Furnace Brook remains a moderately to severely impacted system (Figure 17). Upstream, we observed Oligochaeta populations increase significantly (P=0.0142) from 2016 – 2018, (Figure 18). Within each location, substrate particle size did not differ significantly over time; downstream is dominated by coarse substrates and upstream is uniformly silt (Figure 19).

DISCUSSION

Following small dam removal at Shapp Pond, which was partially breached by prior storms, we observed rapid, ‘across-the-board’ recovery of stream biotic communities and an improvement in water quality designation. Available benthic lotic habitat increased significantly over a one-year period, enabling colonization of favorable invertebrate taxa such as Ephemeroptera, Trichoptera, and Plecoptera. Prior to removal, stream bottom habitat was dominated by sand, which represented ~73% of the substrate, while gravel only represented ~10%. These proportions nearly doubled, in opposing directions, during each of the post-removal recovery years according to visual estimates (Figure 20). Proportion of sand decreased from 73.3% to 36.7% during Year-1 post-removal, then from 36.7% to 15% during Year-2 post-removal. Gravel proportions increased from 10% pre-removal to 35% Year-1 post-removal and then 35% to 60% during Year-2. We also observed cobble substrates increasing from 0% pre-removal to ~13% post-removal.

These habitat changes benefit aquatic invertebrate tax by providing more interstitial spaces within which invertebrates reside. This habitat availability shifts the aquatic invertebrate community towards emergent invertebrate taxa (Figure 21), increasing food supplies for resident fishes (McIntosh, 2000). Furthermore, successful colonization and long-term presence of impact-sensitive taxa, such as Ephemeroptera and Plecoptera, provide evidence of improved water quality. Additionally, the change in substrate class towards gravel and cobble increase spawning habitat for resident fish species, such as trout (Workman et al., 2004).

Our study location at the Wynantskill, which comprised of only post-removal data, illustrates a stable, semi-impacted stream reach, composed of a gravel/cobble substrate. While we lacked pre-removal data at Wynantskill in this study, we can infer a rapid recovery from data at our intact barrier sites. The three-year post-removal mean Water Quality BAP score at Wynantskill is 5.37, indicating a semi-impacted water quality designation. The mean upstream Water Quality BAP score across all our intact barrier sites is 3.24, or moderately impacted. We observe this water quality designation to be consistent over time, regardless of geographic location of the impoundment (Figure 22). Consequently, we can infer that the removal effort at Wynantskill likely improved the water quality by up to ~165%. This assumption can be validated by our invertebrate community data showing soft-sediment favoring taxa Oligochaeta and Ostracoda declining and being replaced by Ephemeroptera, Trichoptera, Coleoptera and Diptera (Figure 23).

Overall, we observed generally stable systems within each of our intact barrier sites at Browns Pond, Sawkill-Annandale Dam, and Furnace Brook-Maiden Ln Dam. Substrate composition trends are relatively flat over time for both upstream and downstream locations. Water Quality designations hover around the moderate/severely impacted designation threshold across upstream locations. While we observe increasing Oligochaeta populations at both Browns Pond and Furnace Brook, the co-dominant upstream taxa at these sites continues to be Chironomidae and Oligochaeta.

At downstream intact barrier locations, substrates and aquatic invertebrate communities remain consistent for the duration of our study period. Sawkill maintains a semi-impacted designation, while Furnace Brook and Browns Pond remain moderately-impacted. The downstream Water Quality designation discrepancy (compared to more similar upstream designations), suggests a broader landscape context influence.

CONCLUSIONS AND RECOMMENDATIONS

We observed rapid recovery of a historically impounded stream reach on the East Branch Wappingers Creek (Shapp Pond), such that biological and ecological characteristics between adjacent impounded and free-flowing segments were barely distinguishable two years after barrier removal. Water quality designations improved from moderately-impacted to semi-impacted, doubling the water quality index score. Ephemeroptera populations increased by five-fold during the first-year post-removal. Oligochaeta populations declined 97% during the second-year post-removal. Species richness doubled over the two years post-removal, and EPT richness of impact-sensitive taxa increased 9-fold during the first-year post-removal. All these results suggest a successful stream restoration effort. Post-removal data at Wynantskill represented a more muted recovery than observed at Shapp Pond, perhaps due to the differing structure characteristics (i.e. tidal gate vs run-of-river dam), but certain trends continued in a positive direction. Trichoptera populations increased 8-fold and Oligochaeta populations decreased 75% over the three years post-removal. The natural flushing of smaller sediments accumulated behind the barrier is evidenced by the mean substrate particle size increasing 2.5-fold during the post-removal study period. The water quality designation is wholly contained within the semi-impacted category, a likely significant improvement from its formerly impounded state.

Our intact-barrier sites displayed ecological and biological stability over our three-year study period, despite widely varying hydrologic conditions – 2016 being very dry and 2017 being very wet. The impounded upstream segment of the intact sites remained moderately to severely impacted during the whole of the study. Segments immediately downstream of the barrier varied between semi and moderately impacted. Our data provides valuable insight into the behavior of stream systems manipulated by long-term barriers. A key complementary question to be addressed by future studies is how broader landscape-level forces govern biological and ecological attributes at a local scale surrounding barriers, and how such external characteristics influence recovery patterns during removal restoration efforts. Our current data suggest a landscape-context effect.

Differences in current water quality metrics at our existing barrier sites and the behavior of post-recovery trajectories at Wynantskill and Shapp Pond support this hypothesis. However, to validate these observations, a more formal, robust landscape-context component must be added to the existing research scope. If landscape context variables can provide strong correlations with post-removal recovery trajectories, important predictive and prioritization models can be developed to optimize future barrier removal and stream restoration efforts.

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Table 1: Collection locations at our five Hudson River tributary study sites in 2017.

Site	Location	Replicate	Northing (m)	Easting (m)	Sample Type
Wynantskill Gate	Upstream	#2	4728925	606493	Kick
		#3	4728939	606444	
		#4	4728943	606421	
Sawkill-Annandale Dam	Upstream	#1	4651474	590369	Ponar Multiplate
		#2	4651573	590357	
		#3	4651612	590383	
	Downstream	#1	4651725	590362	Kick
		#2	4651778	590371	
		#3	4651767	590365	
Shapp Pond Dam	Upstream	#1	4629673	603078	Ponar
		#2	4629687	603074	Kick
		#3	4629714	603081	
	Downstream	#1	4629765	603078	Kick
		#2	4629810	603090	
		#3	4629843	603108	
Browns Pond Dam	Upstream	#1	4589834	565903	Ponar Multiplate
		#2	4589807	565958	
		#3	4589772	566015	
	Downstream	#1	4589730	566065	Kick
		#2	4589721	566082	
		#3	4589740	566071	
Furnace Bk.-Maiden Ln Dam	Upstream	#1	4564941	590924	Ponar Multiplate
		#2	4564967	590901	
		#3	4564965	590876	
	Downstream	#1	4564962	590824	Kick
		#2	4564940	590808	
		#3	4564896	590768	

Table 2: Mean values at Shapp Pond for Species Richness (SPP), Ephemeroptera-Trichoptera-Plecoptera Richness (EPT), Hilsenhoff Biotic Index (HBI), Water Quality (BAP) score, taxa abundance percentages, and substrate particle size.

Location	Year	SPP	EPT	HBI	PMA	BAP	Ephemeroptera %	Chironomidae %	Trichoptera %	Oligochaeta %	Particle Size (cm)
Downstream	2016	19.33	12.0	4.69	80.8	7.55	28.6	29.9	10.3	6.8	51.397
	2017	18.33	10.0	4.55	74.2	7.48	25.5	18.9	13.7	2.1	29.351
	2018	20.33	12.0	4.78	87.6	7.50	37.3	20.5	13.8	0.8	22.300
Upstream	2016	7.67	1.0	6.20	39.2	3.16	7.3	20.1	0.8	30.3	0.131
	2017	17.33	10.0	5.60	69.9	6.36	24.4	38.2	5.2	13.6	2.083
	2018	15.33	8.7	4.97	79.6	6.29	44.2	34.0	10.1	0.9	7.954

Table 3: Mean values at Wynantskill for Species Richness (SPP), Ephemeroptera-Trichoptera-Plecoptera Richness (EPT), Hilsenhoff Biotic Index (HBI), Water Quality (BAP) score, taxa abundance percentages, and substrate particle size.

Location	Year	SPP	EPT	HBI	PMA	BAP	Ephemeroptera %	Chironomidae %	Trichoptera %	Oligochaeta %	Particle Size (cm)
Upstream	2016	15.5	7.5	6.06	67.2	5.70	17.4	27.2	2.3	27.7	0.454
	2017	12.0	6.5	6.19	74.5	4.91	49.0	17.5	8.2	19.6	1.218
	2018	12.0	5.5	5.40	79.5	5.44	32.1	27.5	20.7	6.9	1.044

Table 4: Mean values at Browns Pond for Species Richness (SPP), Ephemeroptera-Trichoptera-Plecoptera Richness (EPT), Hilsenhoff Biotic Index (HBI), Water Quality (BAP) score, taxa abundance percentages, and substrate particle size.

Location	Year	SPP	EPT	HBI	PMA	BAP	Ephemeroptera %	Chironomidae %	Trichoptera %	Oligochaeta %	Particle Size (cm)
Downstream	2016	11.0	1.7	6.15	31.3	3.36	0	23.3	3.7	2.3	14.145
	2017	12.3	1.3	6.38	38.8	4.16	0	22.8	8.5	7.7	23.045
	2018	10.0	2.7	5.84	38.3	3.84	4.5	22.0	7.8	1.3	27.232
Upstream	2016	4.6	0	6.57	31.7	1.56	0	65.0	0	19.2	0.002
	2017	3.8	0.33	6.80	32.8	1.67	0	45.3	1.3	41.1	0.002
	2018	3.8	0	7.21	27.4	1.34	0	35.9	0	51.9	0.002

Table 5: Mean values at Sawkill-Annandale Dam for Species Richness (SPP), Ephemeroptera-Trichoptera-Plecoptera Richness (EPT), Hilsenhoff Biotic Index (HBI), Water Quality (BAP) score, taxa abundance percentages, and substrate particle size.

Location	Year	SPP	EPT	HBI	PMA	BAP	Ephemeroptera %	Chironomidae %	Trichoptera %	Oligochaeta %	Particle Size (cm)
Downstream	2016	19.7	10.0	5.51	68.6	6.53	25.0	36.4	11.5	0.3	3.731
	2017	21.0	11.0	4.80	67.3	7.34	26.5	24.4	23.1	4.5	3.172
	2018	21.7	11.7	4.75	72.9	7.27	33.9	22.7	16.3	2.3	2.607
Upstream	2016	12.5	2.3	6.54	38.9	3.18	1.8	62.0	1.9	14.7	0.002
	2017	10.0	2.7	6.16	42.5	3.46	5.1	61.2	10.1	4.9	0.002
	2018	13.8	3.5	6.81	44.5	4.04	6.5	40.4	1.8	10.4	0.002

Table 6: Mean values at Furnace Brook-Maiden Ln Dam for Species Richness (SPP), Ephemeroptera-Trichoptera-Plecoptera Richness (EPT), Hilsenhoff Biotic Index (HBI), Water Quality (BAP) score, taxa abundance percentages, and substrate particle size.

Location	Year	SPP	EPT	HBI	PMA	BAP	Ephemeroptera %	Chironomidae %	Trichoptera %	Oligochaeta %	Particle Size (cm)
Downstream	2016	13.7	4.7	5.65	45.9	4.92	1.2	44.0	17.7	0.0	7.916
	2017	11.3	3.7	5.92	44.0	3.68	2.0	71.8	10.7	0.9	5.329
	2018	13.0	4.3	5.67	47.3	5.19	3.8	29.3	19.9	0.6	6.412
Upstream	2016	8.5	0.3	7.22	26.5	1.89	2.2	9.8	0	58.2	0.002
	2017	8.7	0.8	7.06	33.4	2.27	0.9	30.8	0.1	48.1	0.002
	2018	6.2	0.7	7.71	18.4	1.10	2.0	6.9	0.1	86.1	0.002

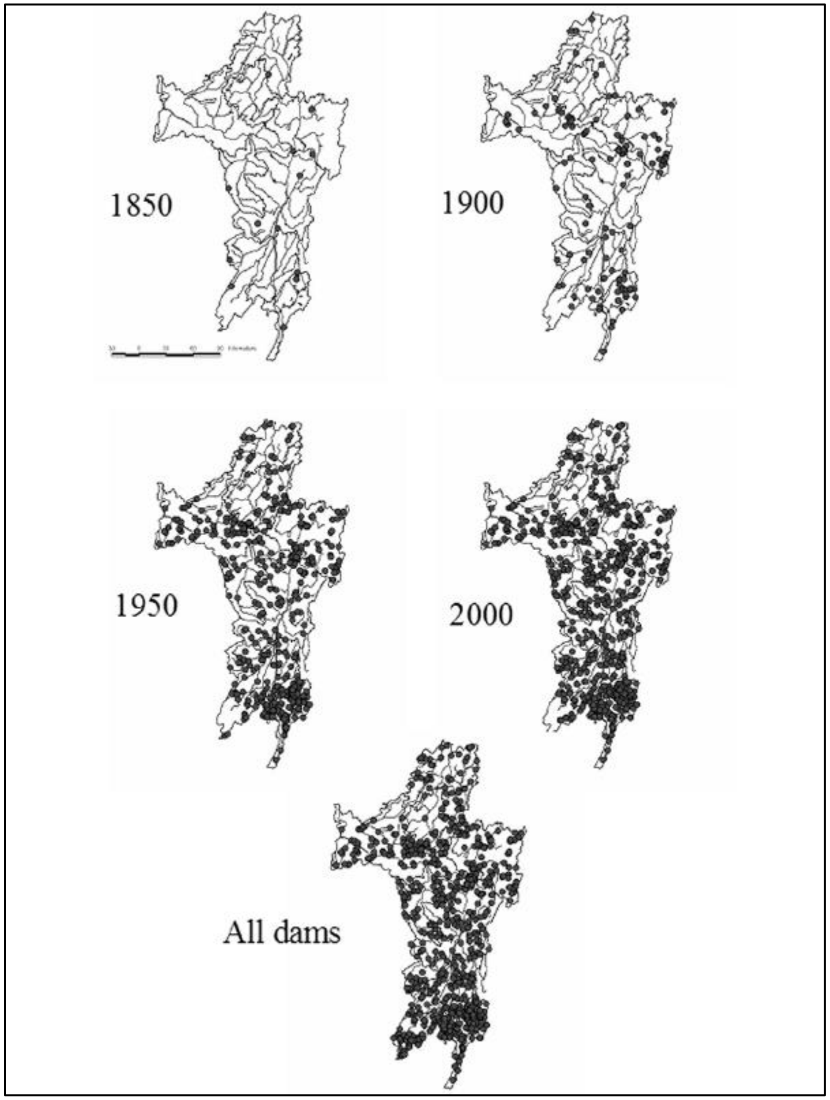

Figure 1: Accumulation of dams over time in the Hudson watershed. Top four maps show the cumulative change in numbers of dams, by dates of completion, every 50 years from 1850 onwards. Bottom map shows all dams, inclusive of those lacking dates. From: Swaney et al. (2006). Data source: BASINS (USEPA 2001).

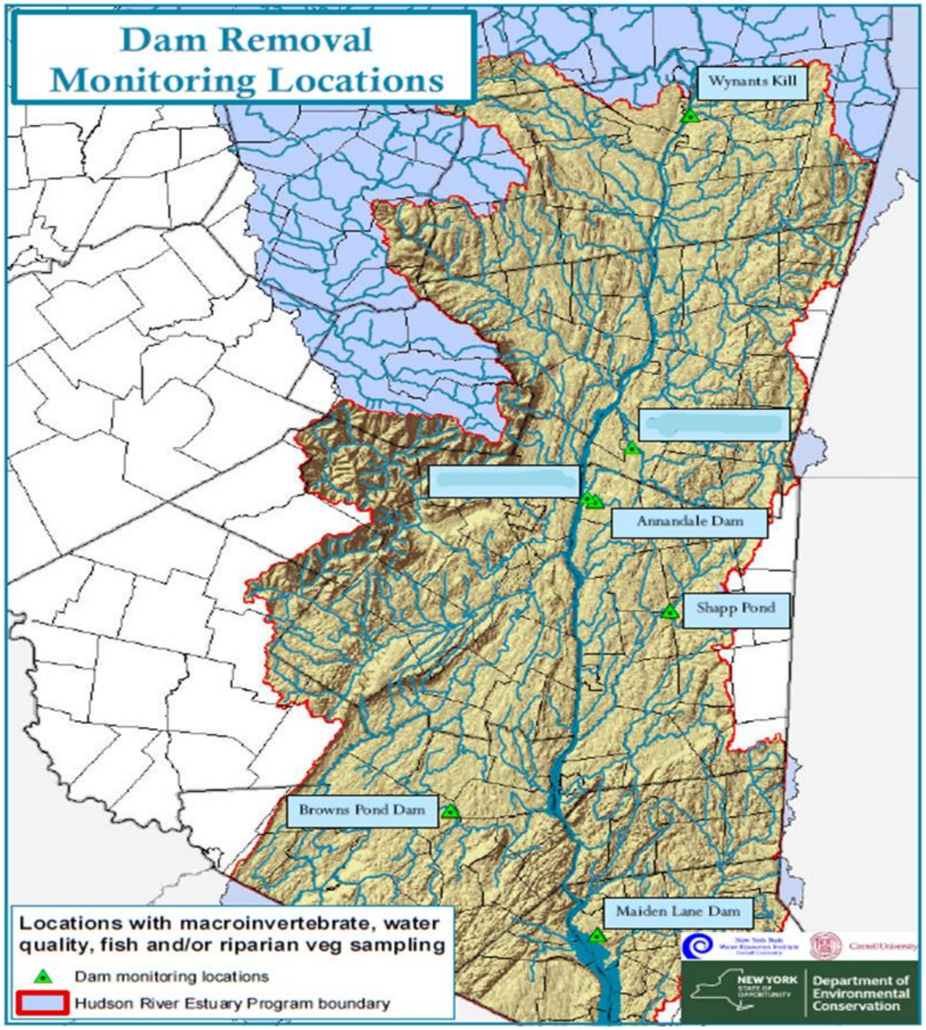

Figure 2: Study site locations within the Hudson River watershed for the period 2016 – 2018.

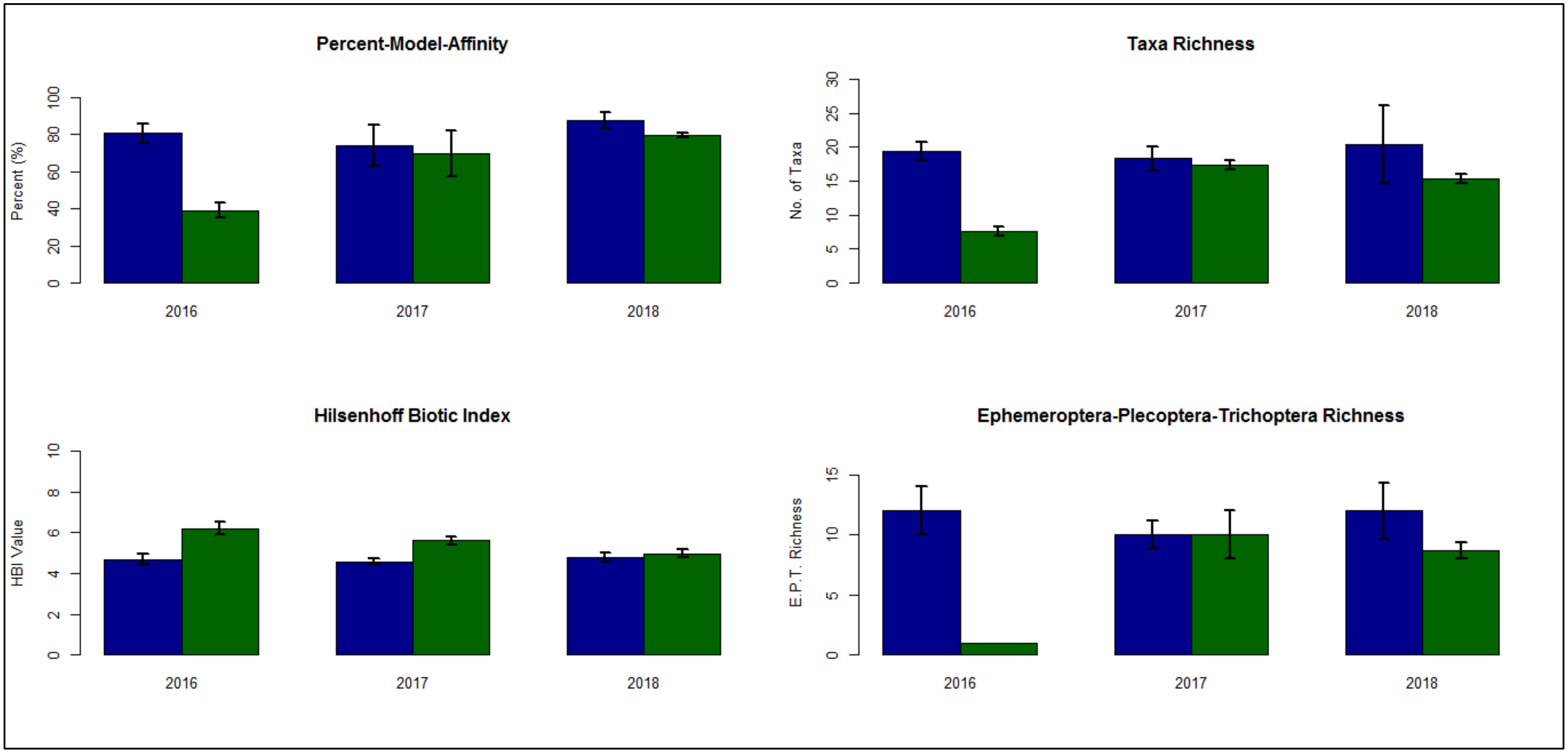

Figure 3: Comparison of Bio-Index metrics at Shapp Pond between upstream (green) and downstream (blue) locations during the study period 2016-2018. Upstream indices display significant improvements across all metrics following dam removal in 2016.

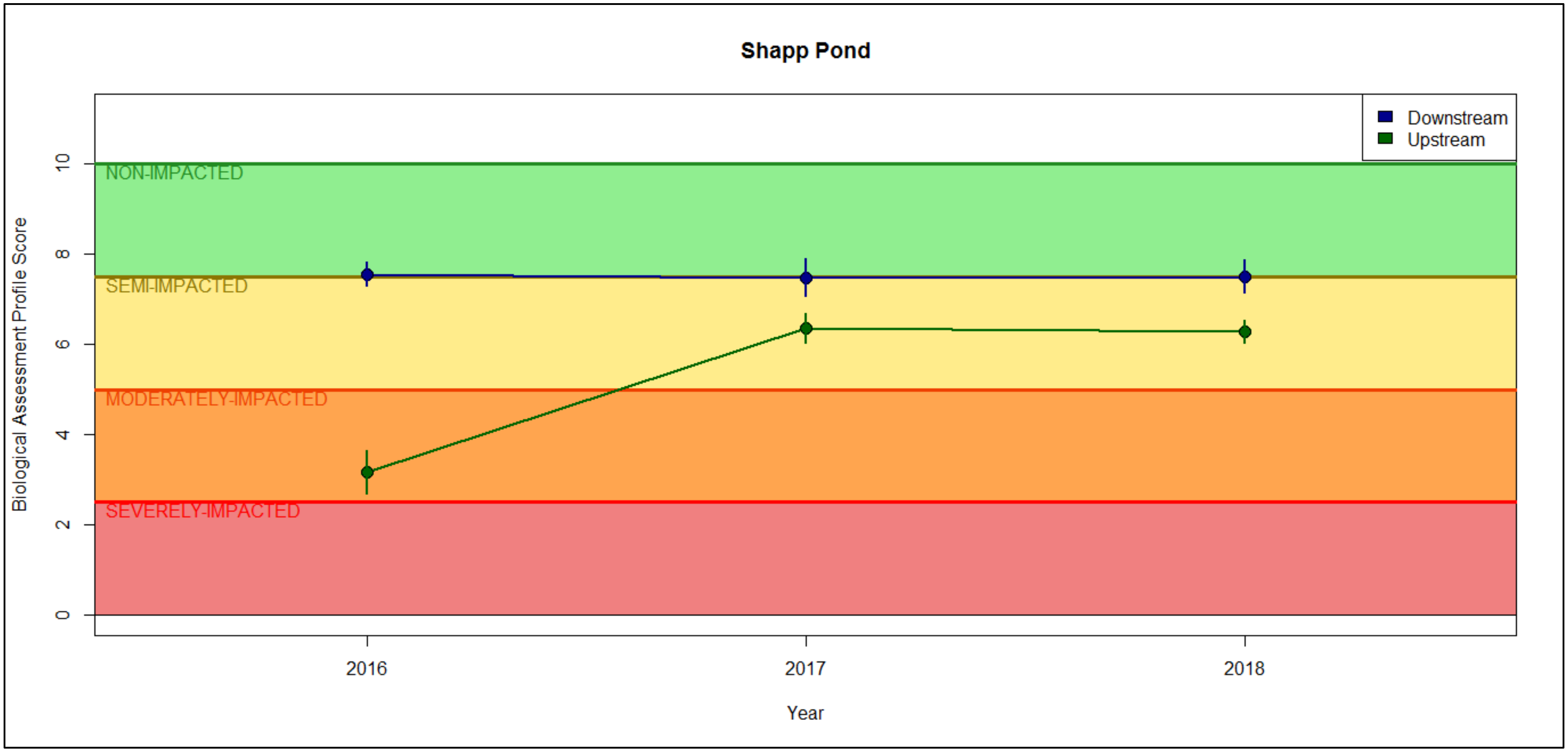

Figure 4: Biological Assessment Profile Score and water quality impact designation trends at Shapp Pond for upstream and downstream locations between 2016-2018. BAP score improved from moderate-impact to semi-impact following dam removal in October 2016.

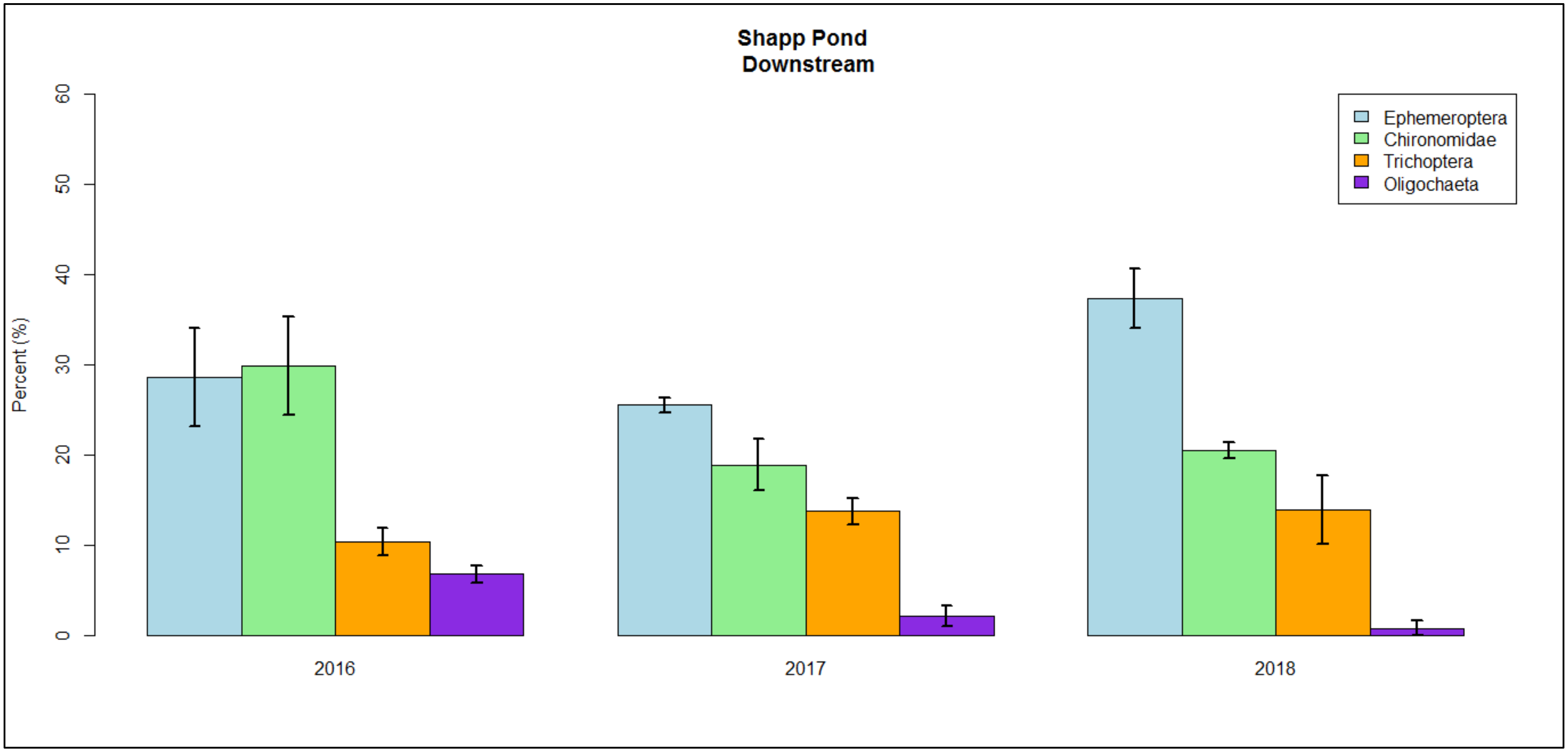

Figure 5: Relative abundance trends for key macroinvertebrate taxa at Shapp Pond, downstream location, over the study period 2016-2018. Oligochaeta decrease significantly during post-removal years (2017,2018).

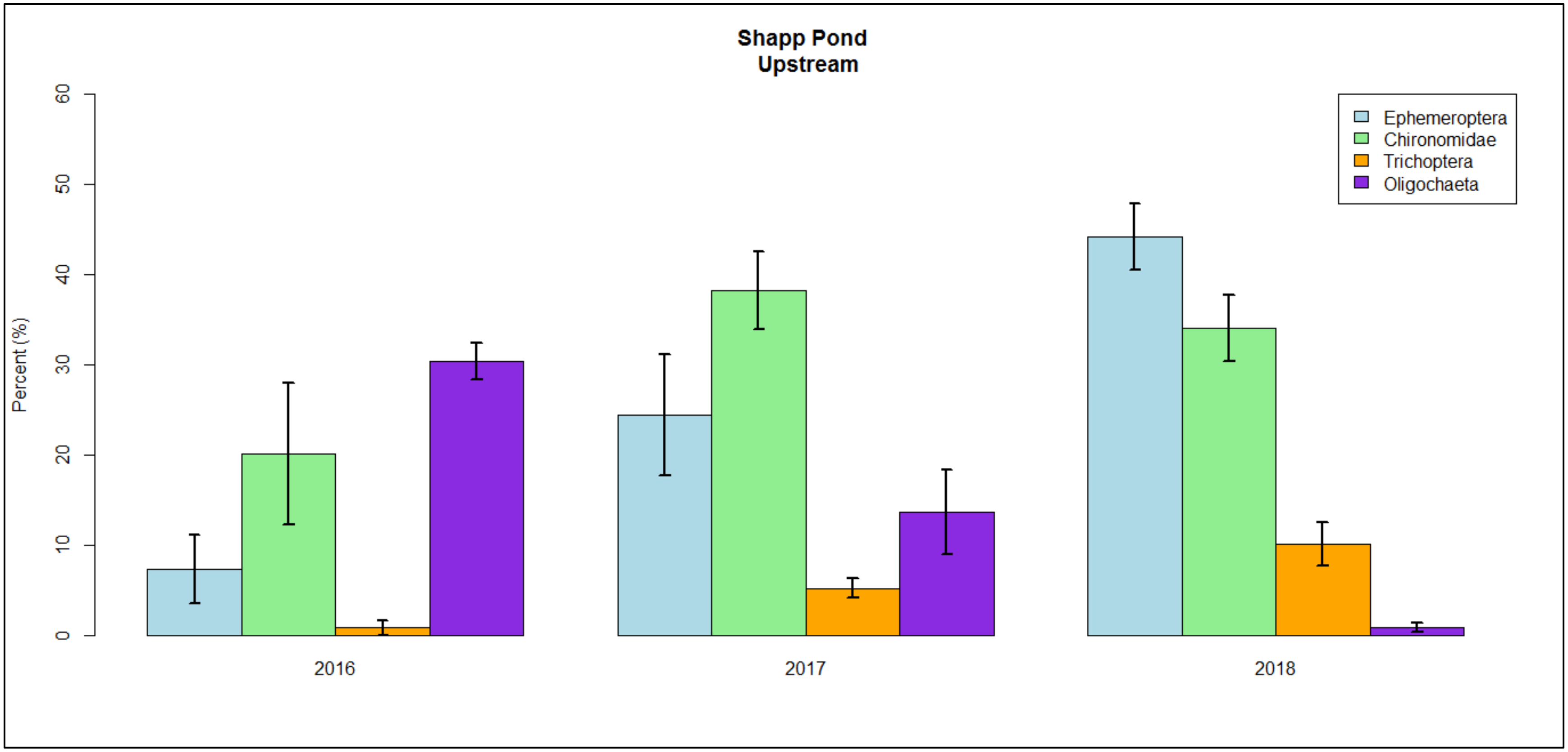

Figure 6: Relative abundance trends for key macroinvertebrate taxa at Shapp Pond, upstream location, over the study period 2016-2018. Significant increases are observed for Ephemeroptera and Trichoptera from pre-removal year (2016) through post-removal years (2017,2018). Oligochaeta experience opposing, significant decreasing trends.

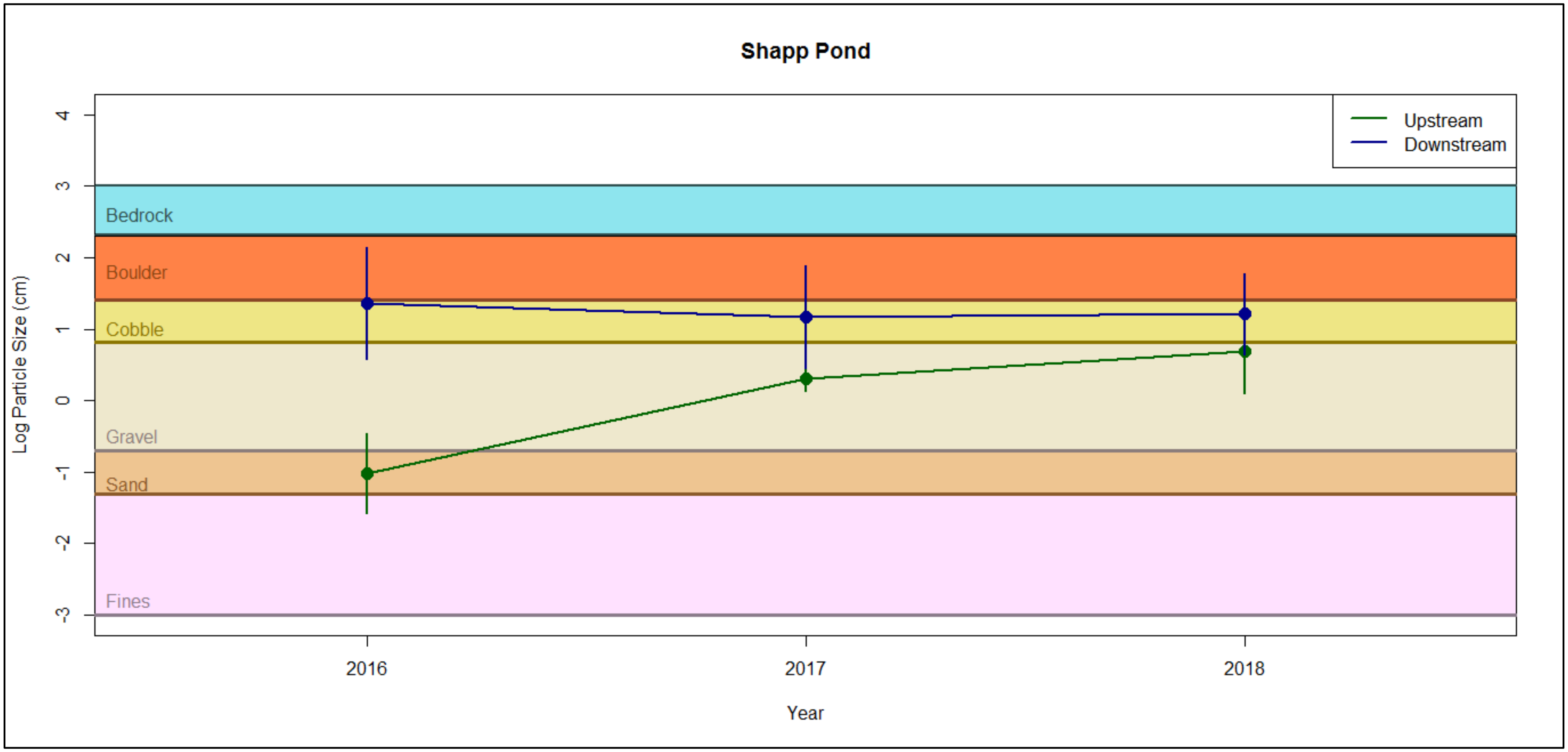

Figure 7: Substrate particle size trends at Shapp Pond for both upstream and downstream locations. Particle size was significantly lower upstream pre-removal year (2016) but differences were lost during post-removal years (2017-2018).

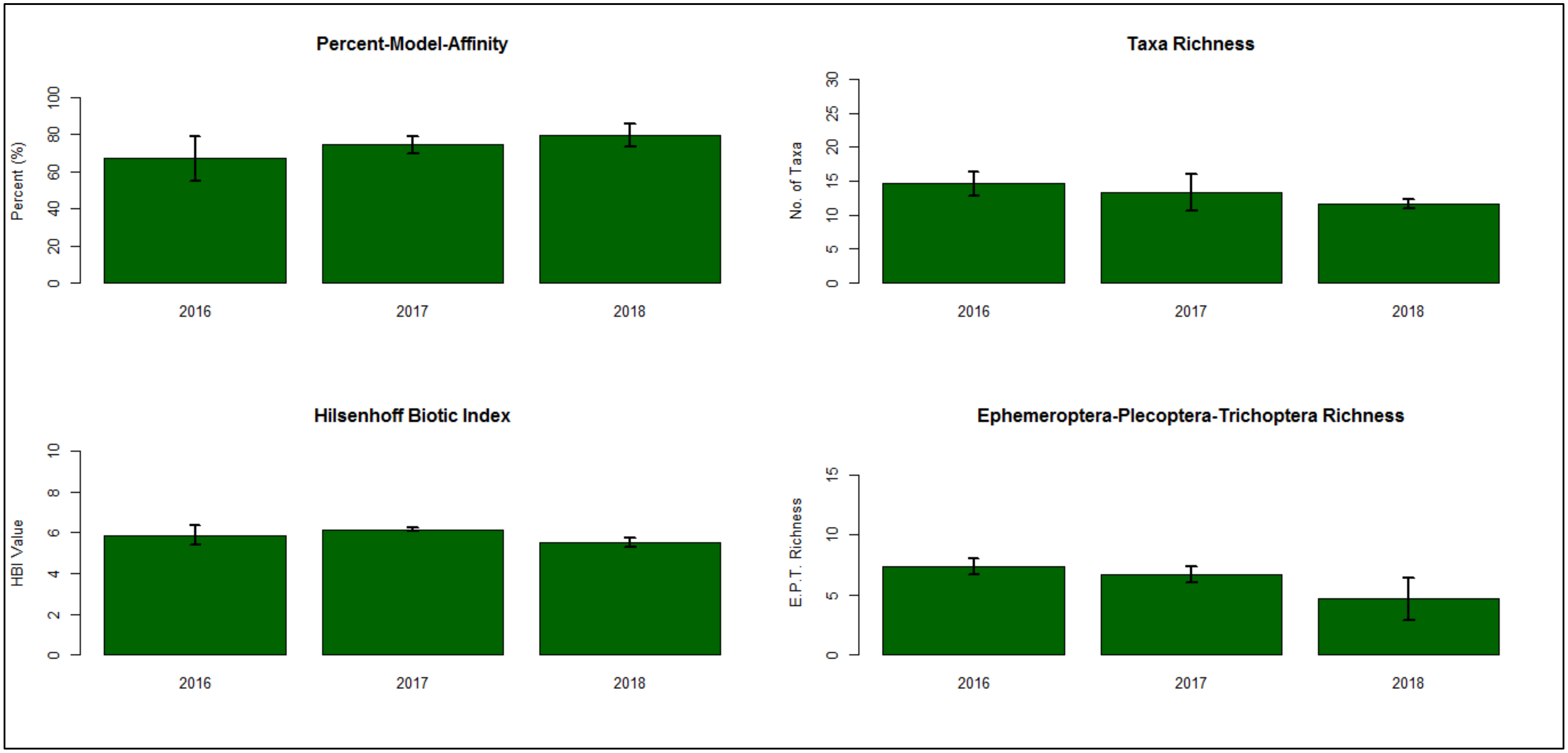

Figure 8: Bio-Index metrics on the Wynantskill during the study period 2016-2018. Upstream indices display significant trends for Percent Model Affinity, Taxa Richness, and E-P-T Richness.

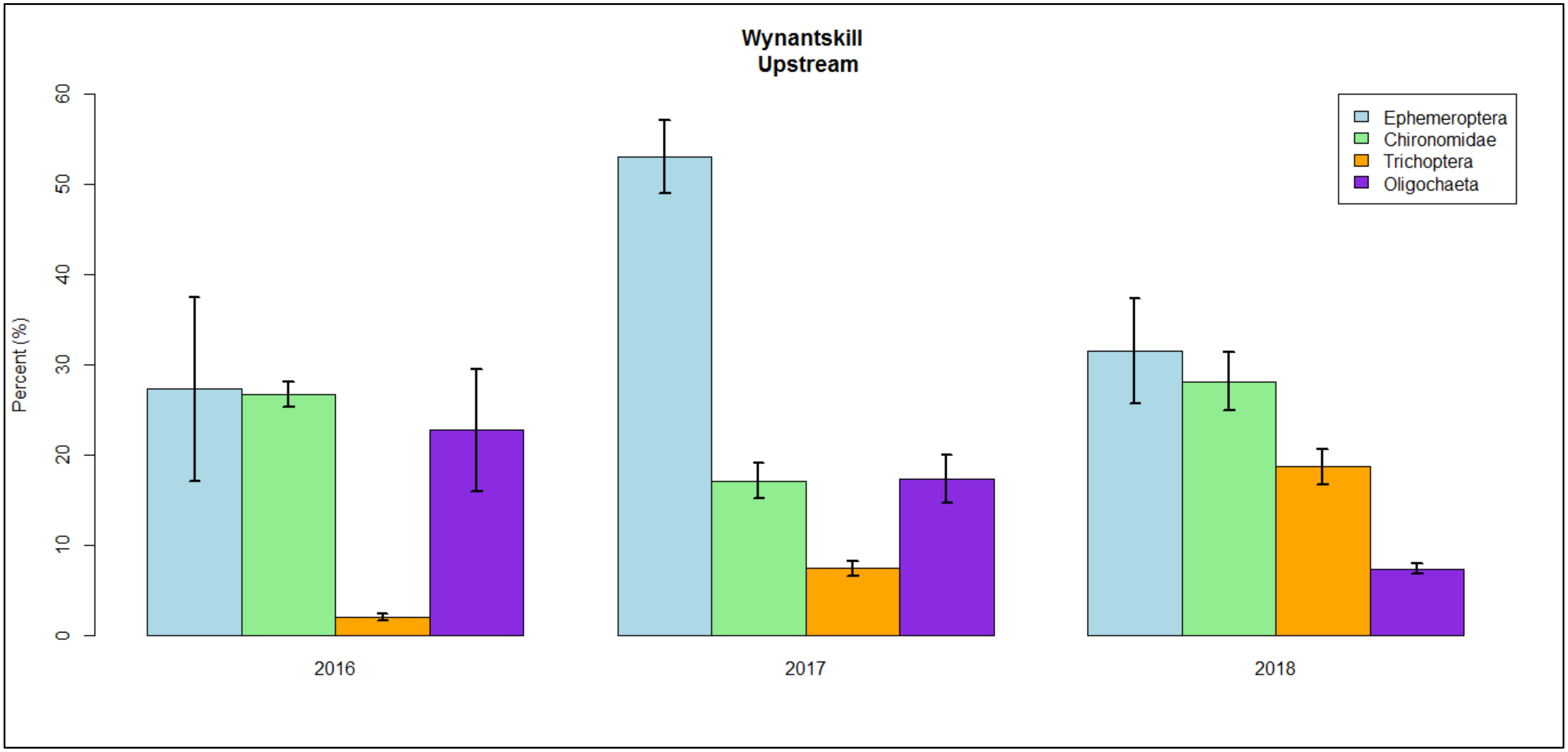

Figure 9: Post-removal relative abundance trends for key macroinvertebrate taxa at Wynantskill, upstream location, between 2016-2018. Trichoptera populations significantly increase, while Oligochaeta populations significantly decrease.

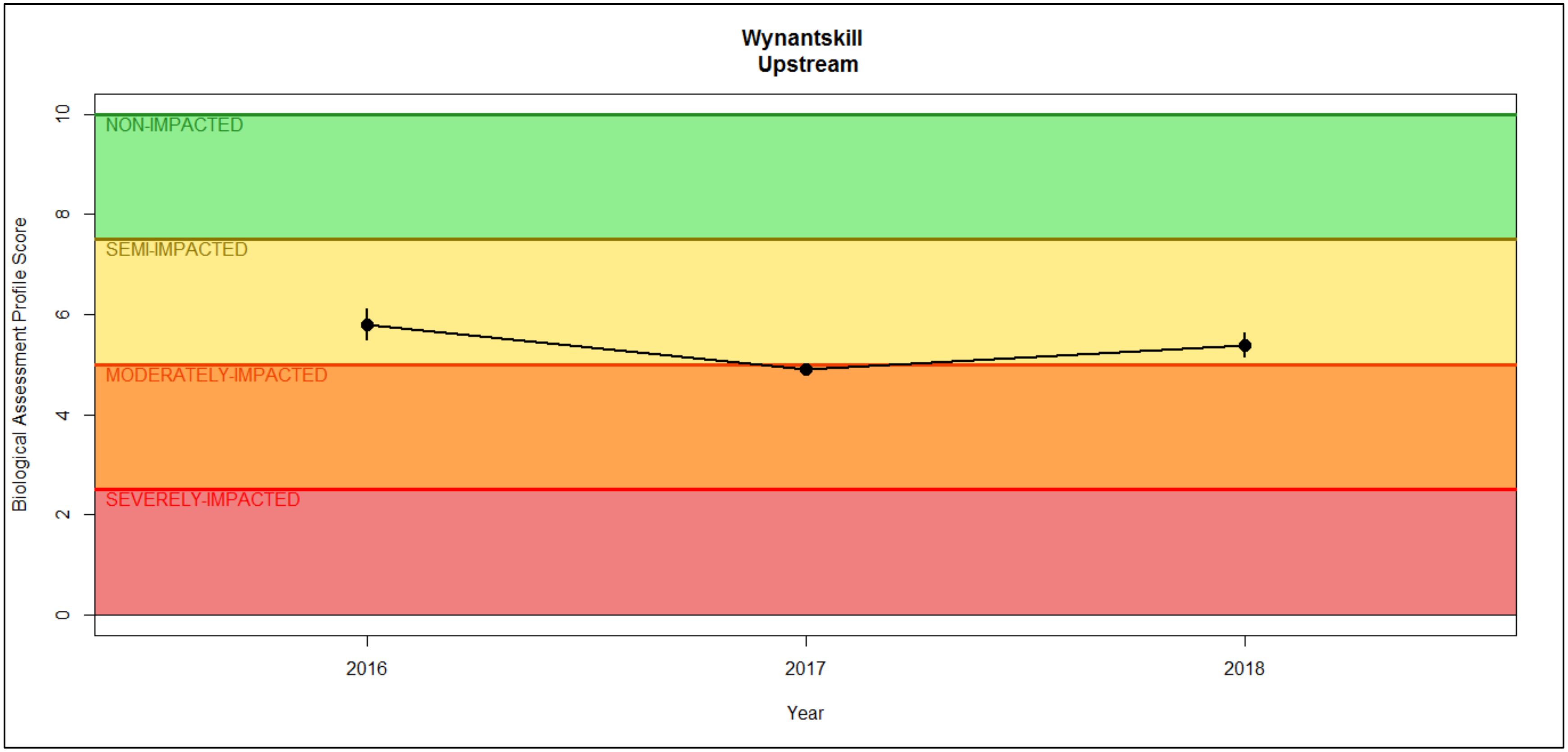

Figure 10: Post-removal Biological Assessment Profile Score and water quality impact designation trend at Wynantskill, upstream location, between 2016-2018.

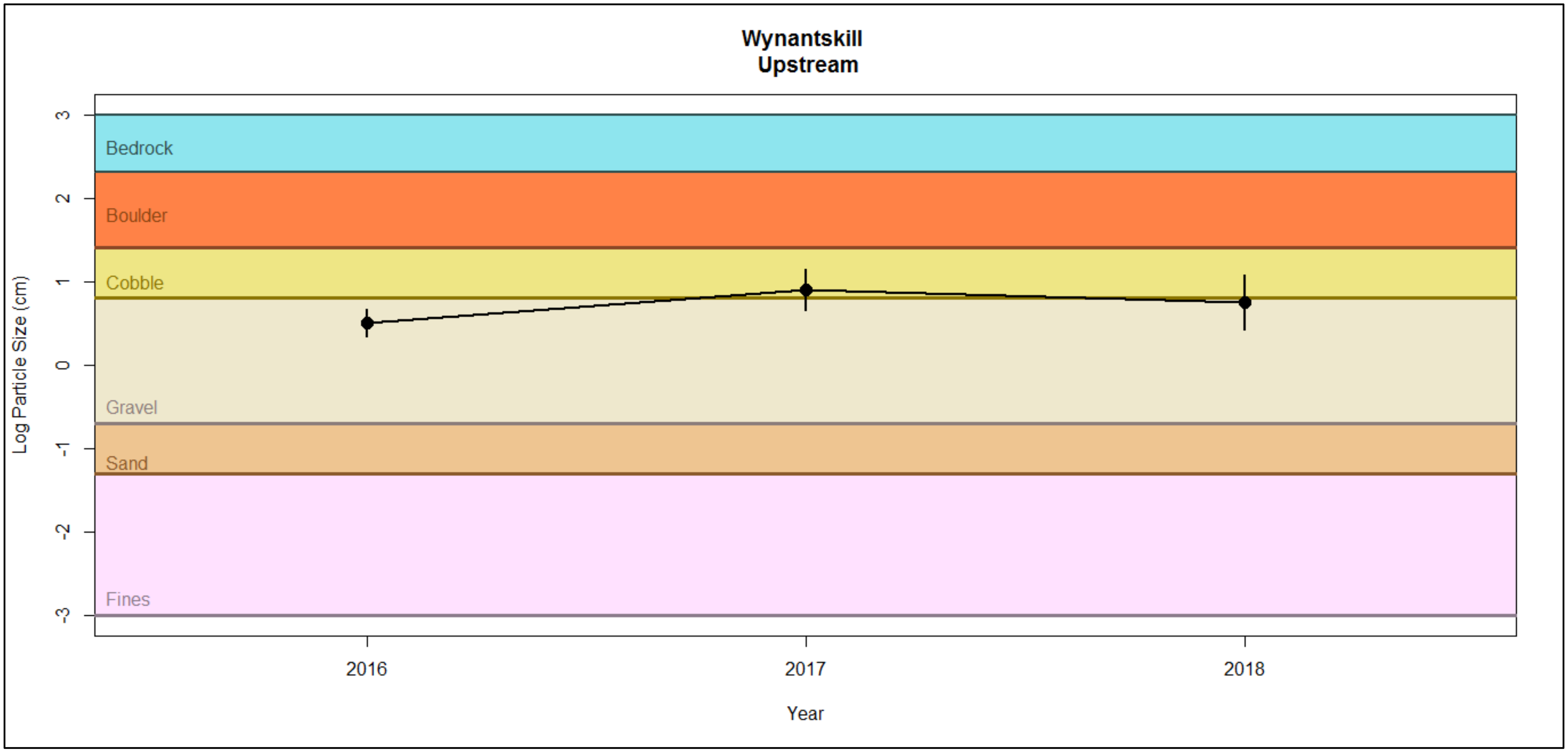

Figure 11: Post-removal substrate particle size trend at Wynantskill, upstream location, between 2016-2018.

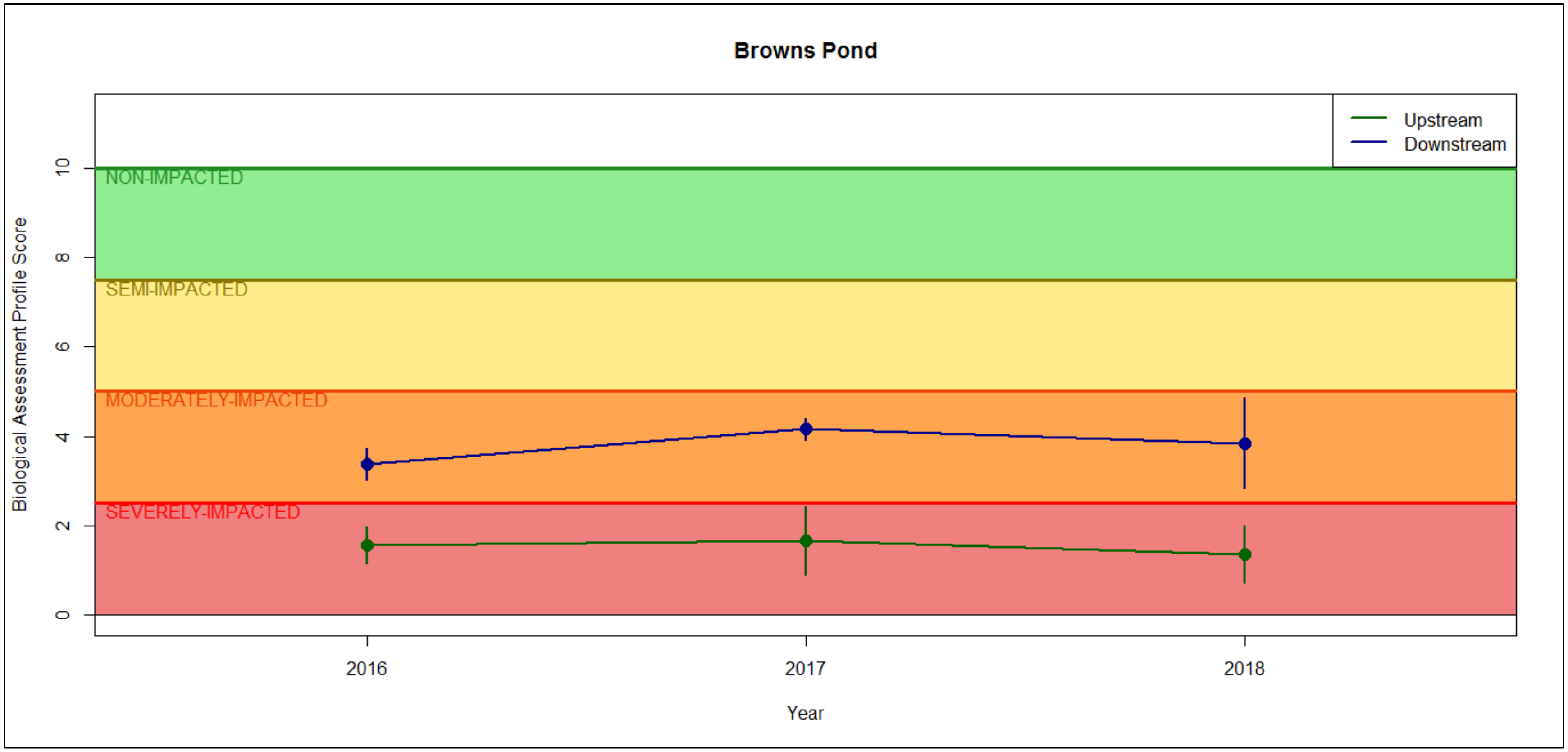

Figure 12: Pre-removal Biological Assessment Profile Score and water quality impact designation trends at Browns Pond for upstream and downstream locations between 2016-2018.

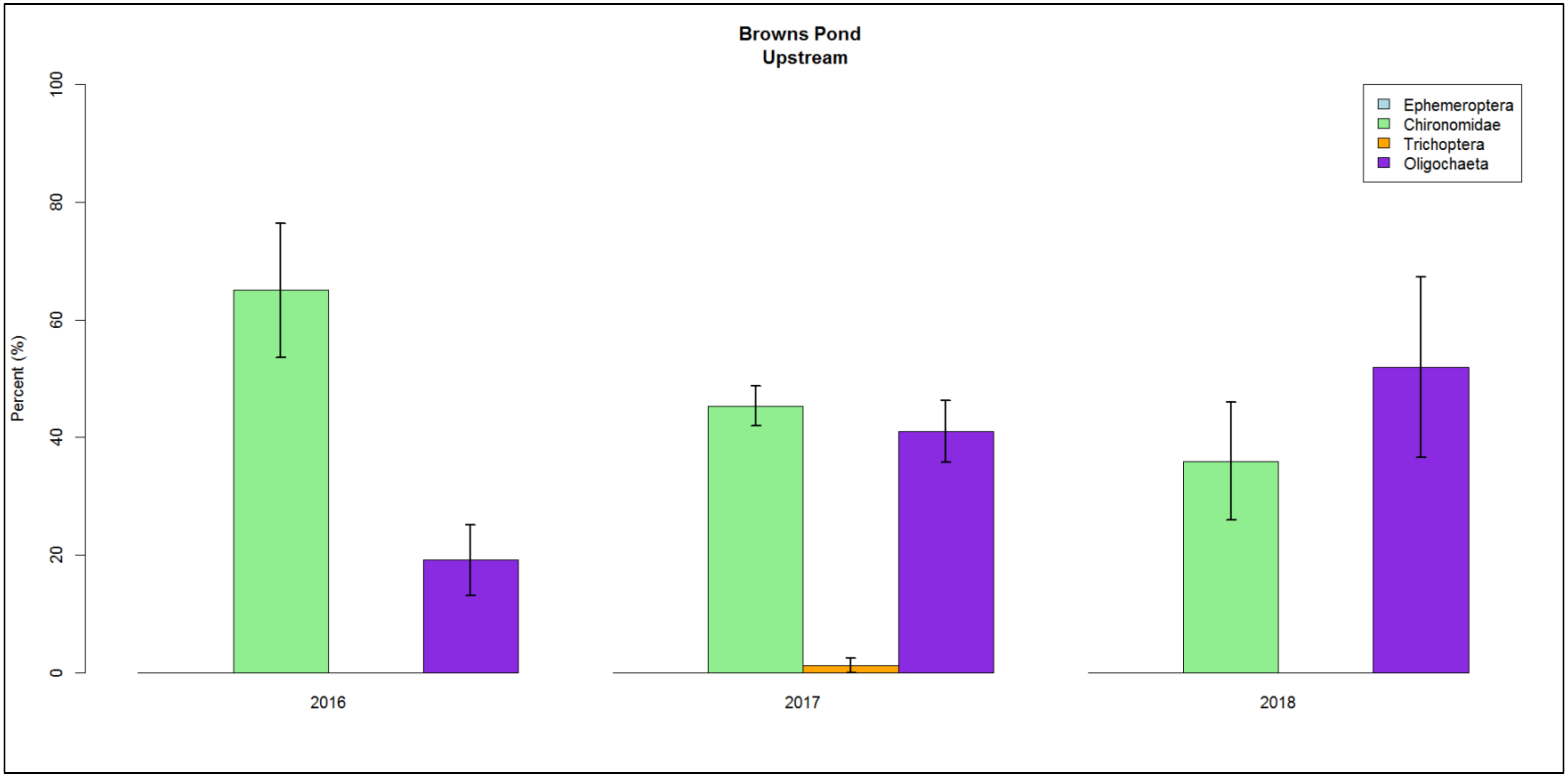

Figure 13: Pre-removal relative abundance variability for key macroinvertebrate taxa at Browns Pond, upstream location, between 2016-2018. Chironomidae populations significantly decrease, while Oligochaeta significantly increase between 2016-2018.

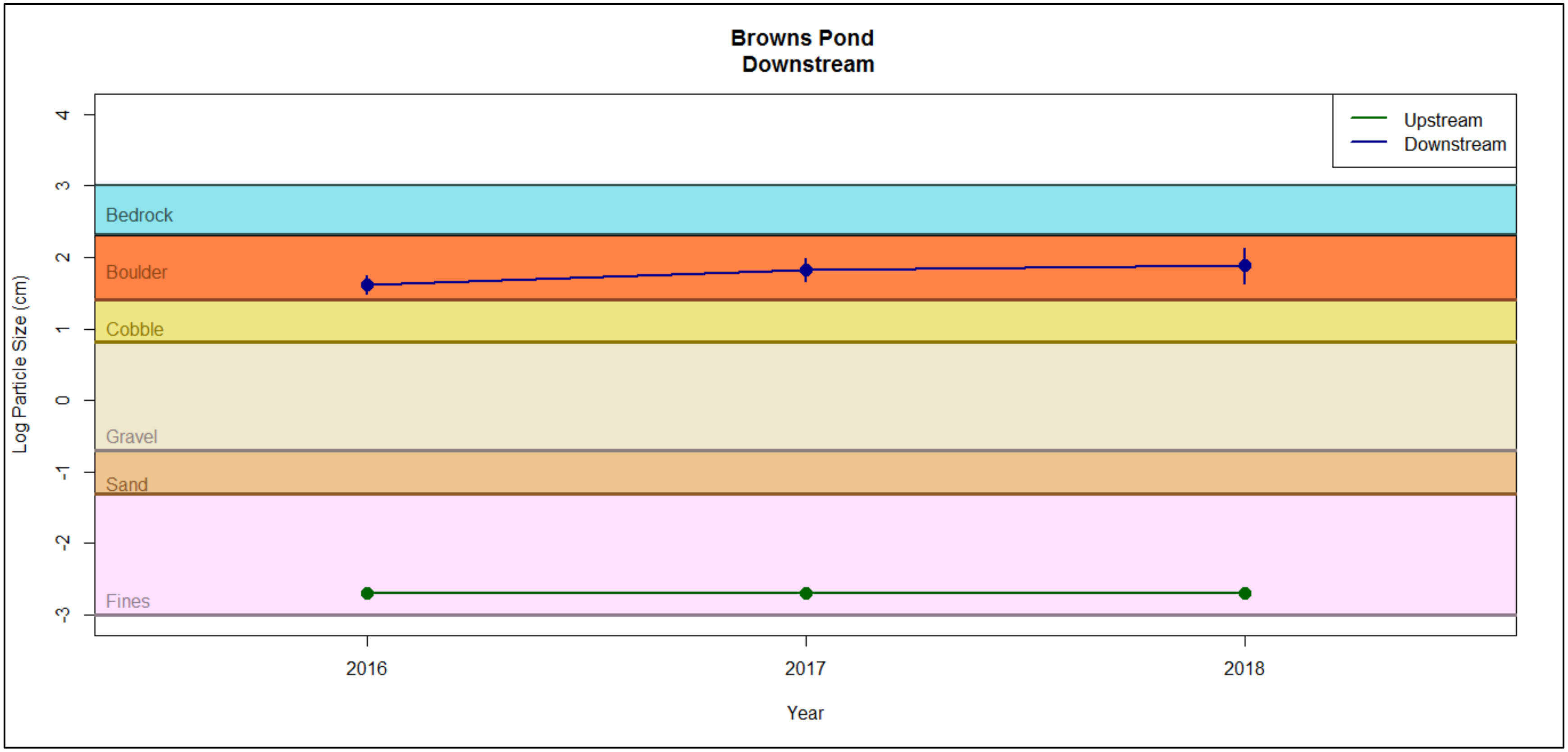

Figure 14: Pre-removal substrate particle size trends at Browns Pond for both upstream and downstream locations between 2016-2018.

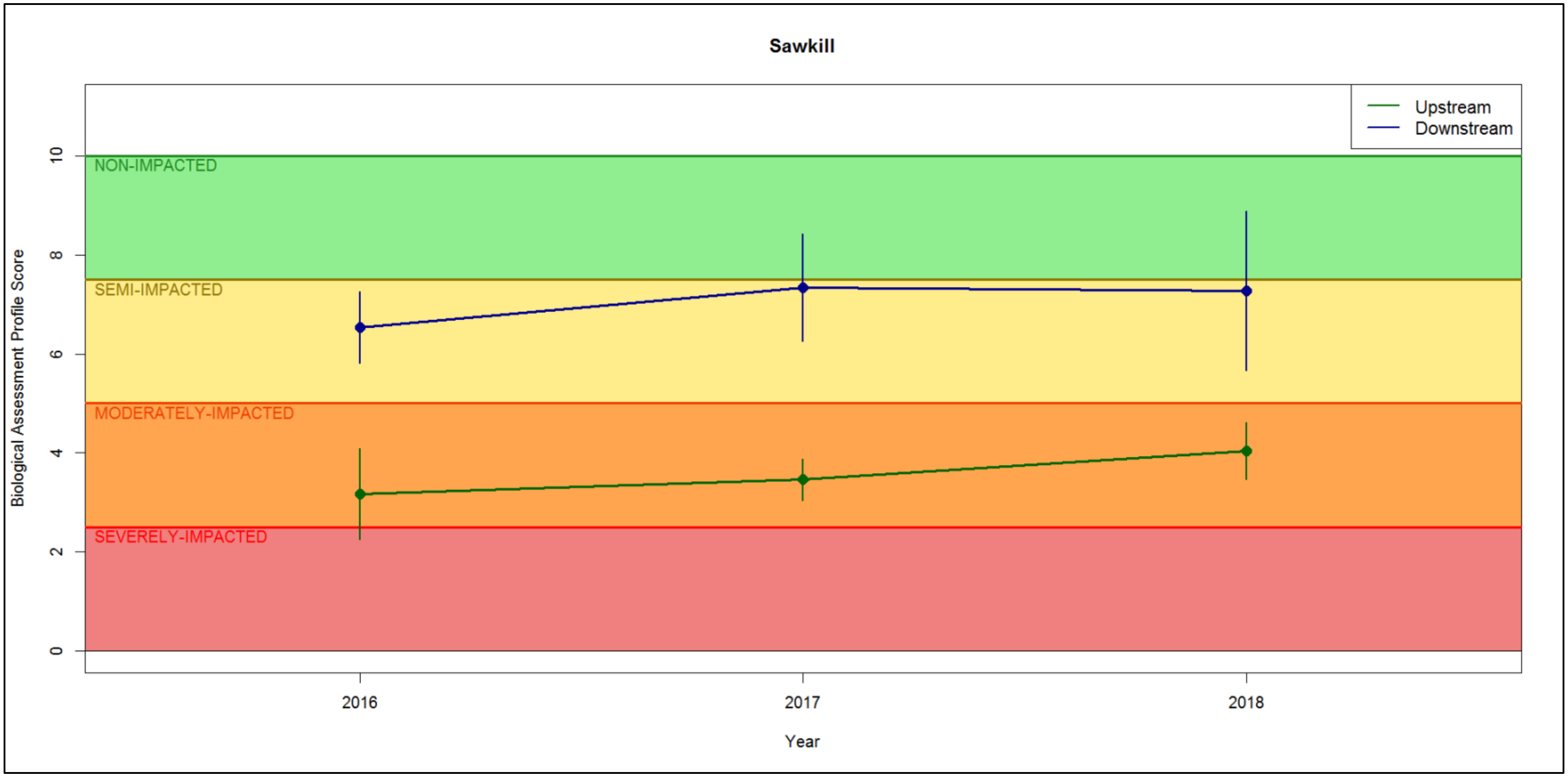

Figure 15: Pre-removal Biological Assessment Profile Score and water quality impact designation trends at Sawkill-Annandale Dam for upstream and downstream locations between 2016-2018.

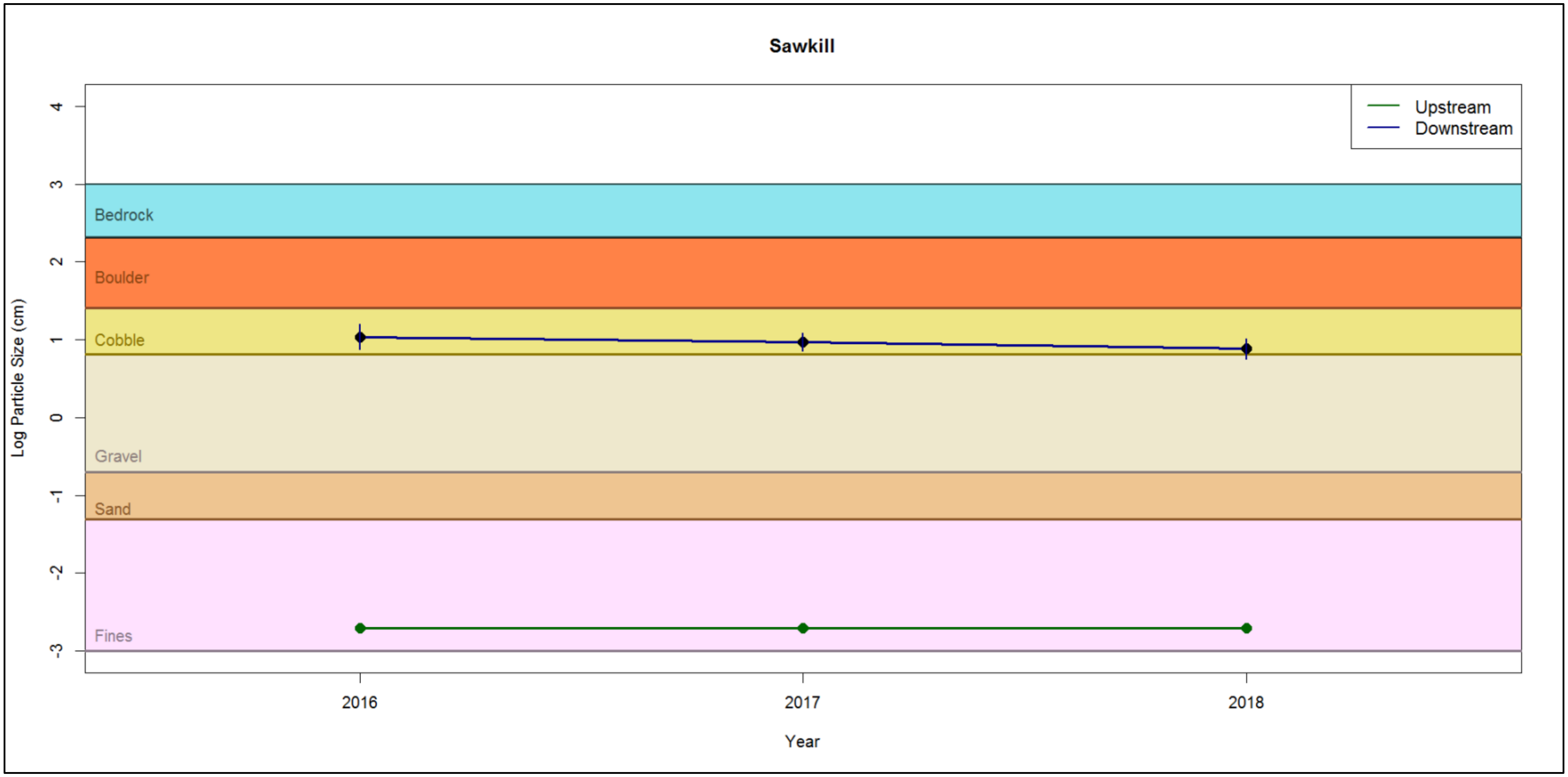

Figure 16: Pre-removal substrate particle size trends at Sawkill-Maiden Ln Dam for both upstream and downstream locations between 2016-2018.

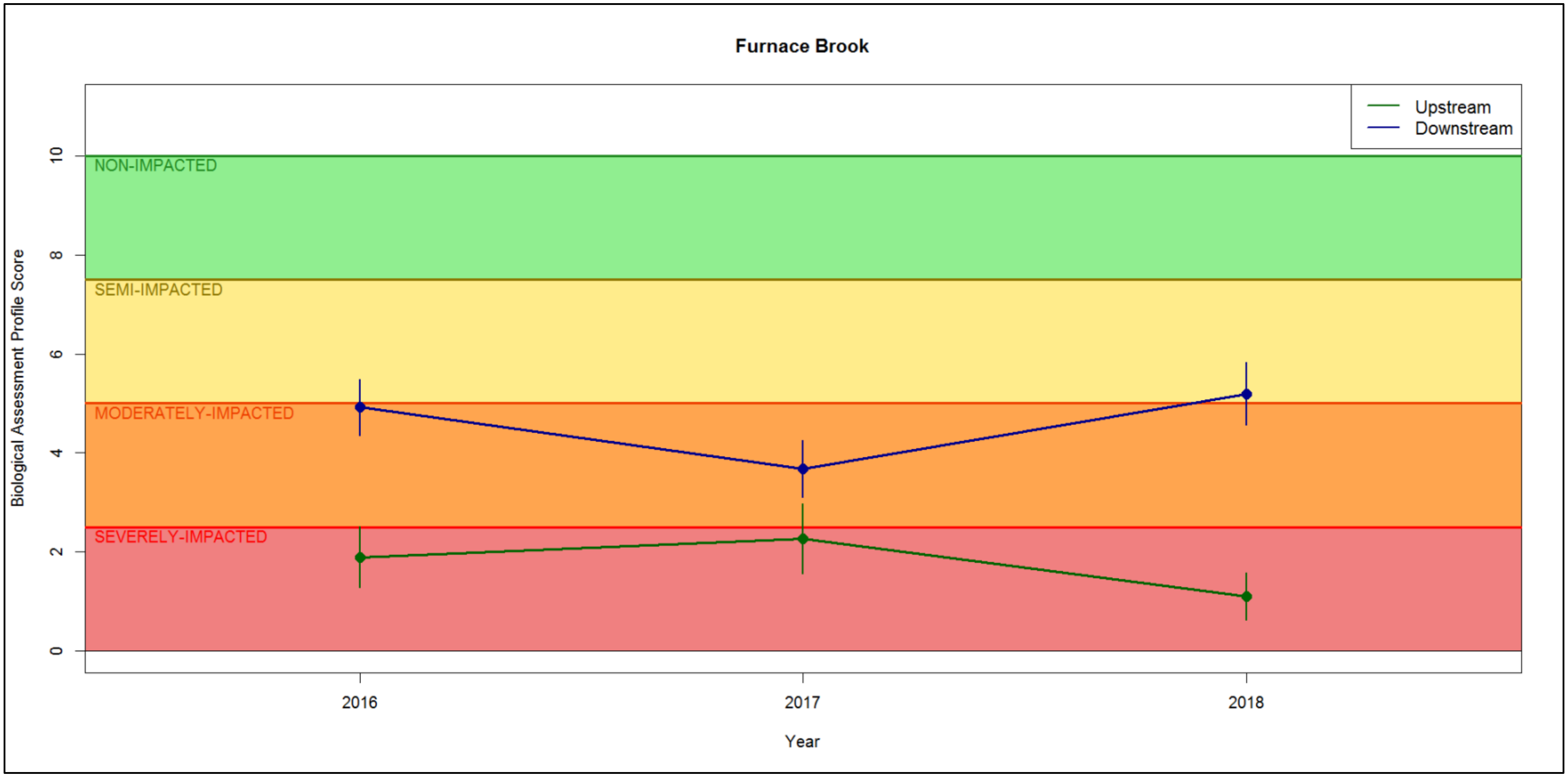

Figure 17: Pre-removal Biological Assessment Profile Score and water quality impact designation trends at Furnace Brook-Maiden Ln Dam for upstream and downstream locations between 2016-2018.

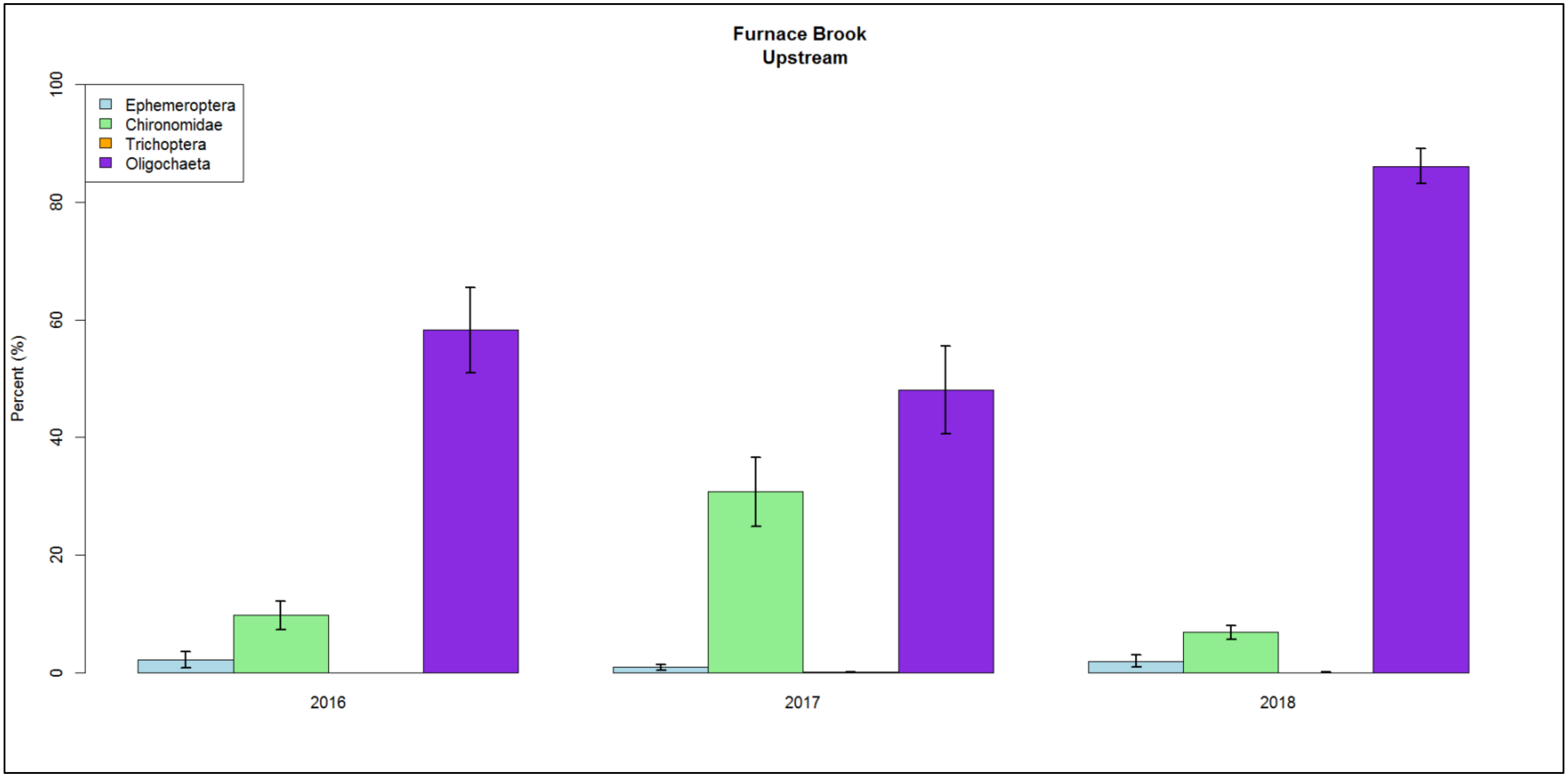

Figure 18: Pre-removal relative abundance variability for key macroinvertebrate taxa at Furnace Brook – Maiden Ln Dam, upstream location. Oligochaeta populations increase significantly between 2016-2018

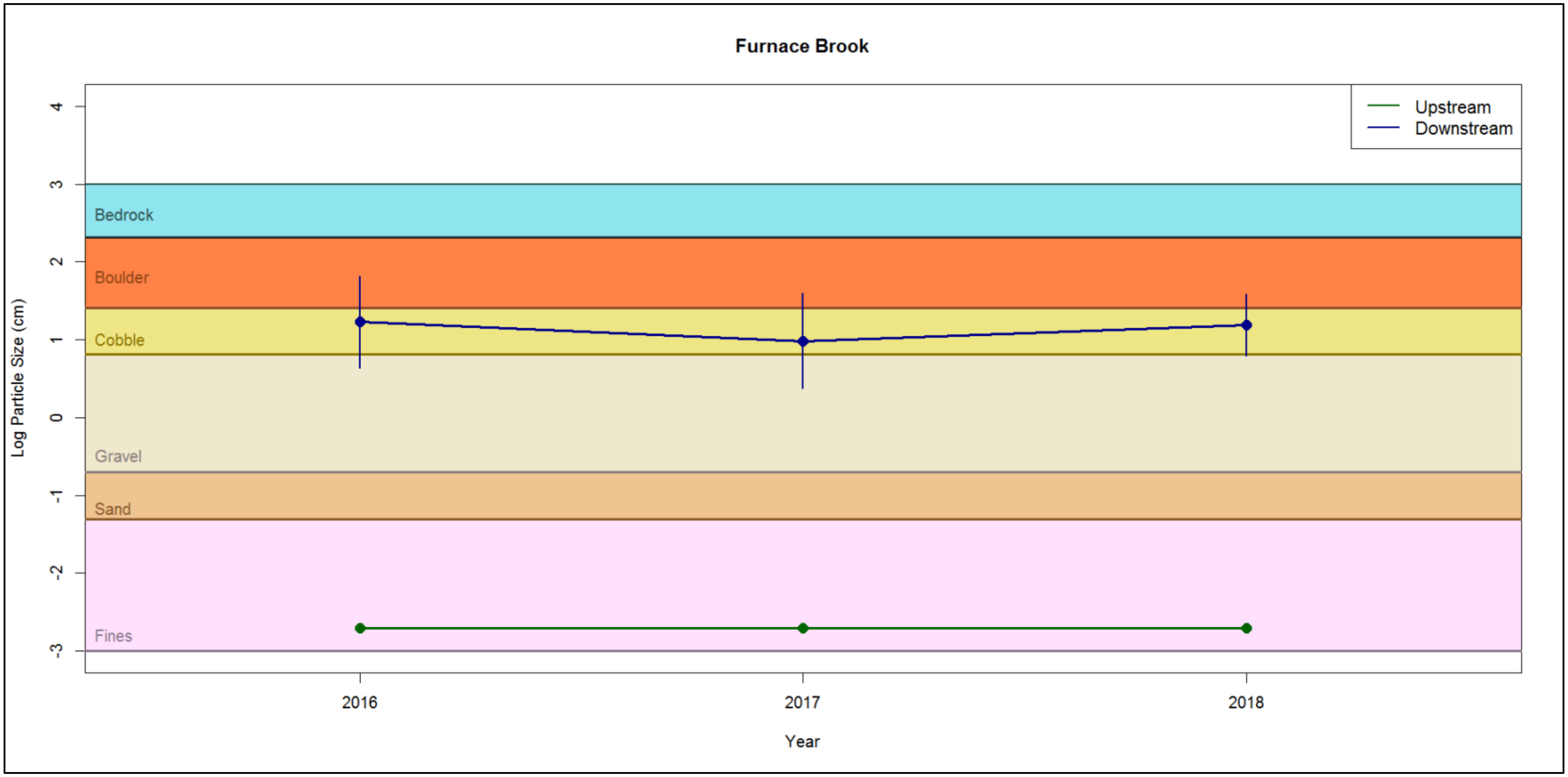

Figure 19: Pre-removal substrate particle size trends at Furnace Brook-Maiden Ln Dam for both upstream and downstream locations between 2016-2018.

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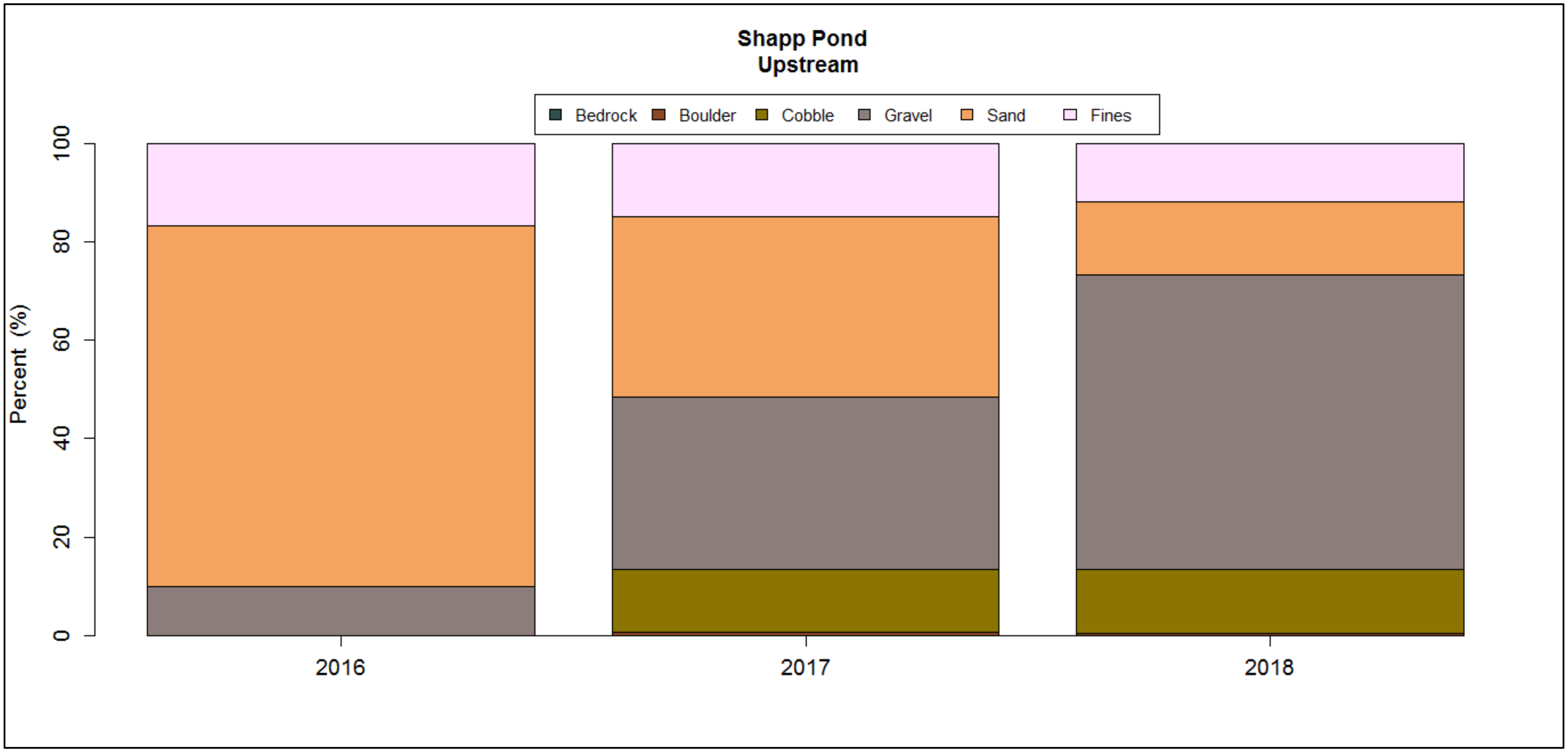

Figure 20: Substrate composition at Shapp Pond over the period 2016 – 2018. Pre-removal data in 2016 illustrates the stream bottom is dominated by sand and silt. During the post removal years (2017, 2018) sand is flushed out of the system and erosive forces expose coarser substrates like gravel and cobble. By 2018, two years post-removal, gravel substrates dominate.

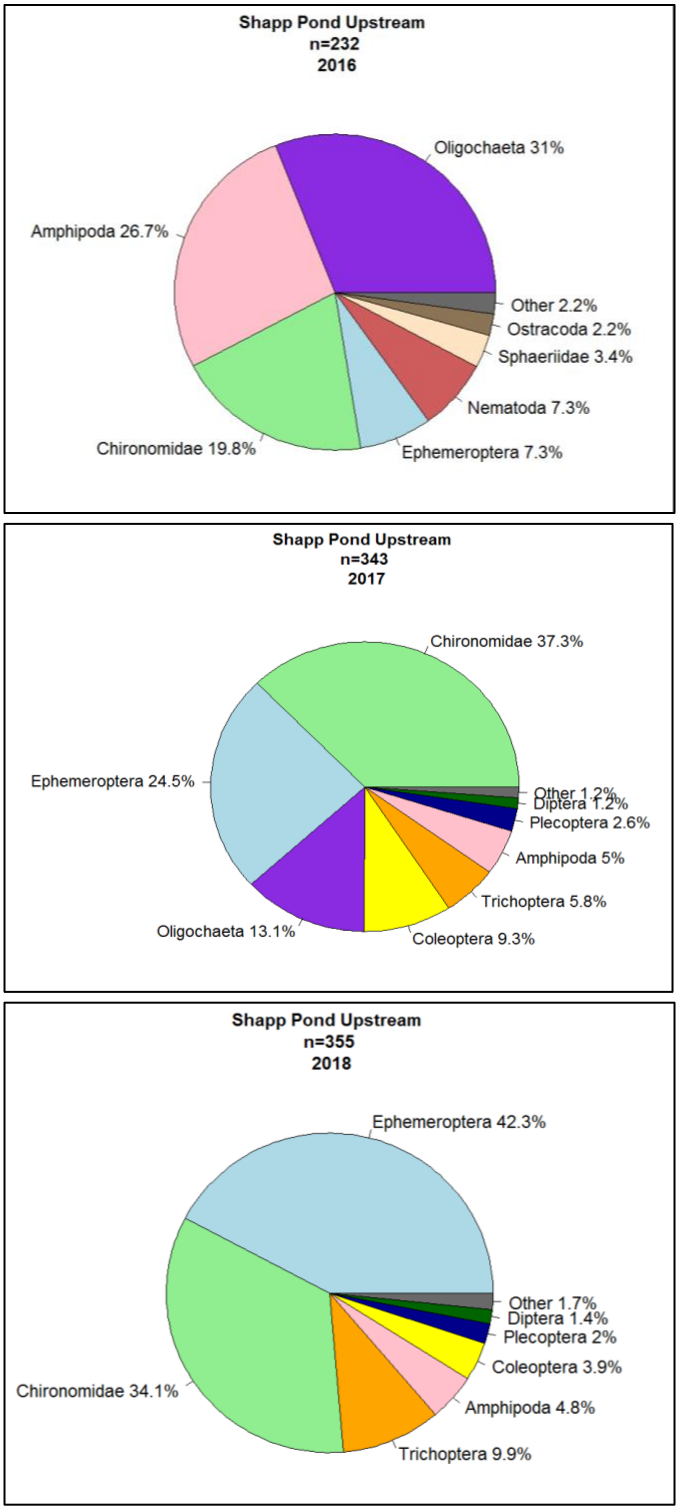

Figure 21: Aquatic invertebrate community composition at Shapp Pond, upstream location. Dam removal recovery trend shows benthic taxa (Oligochaeta, Nematoda, Ostracoda, and Amphipoda) being replaced by emergent taxa: Ephemeroptera, Trichoptera, Plecoptera, and Diptera.

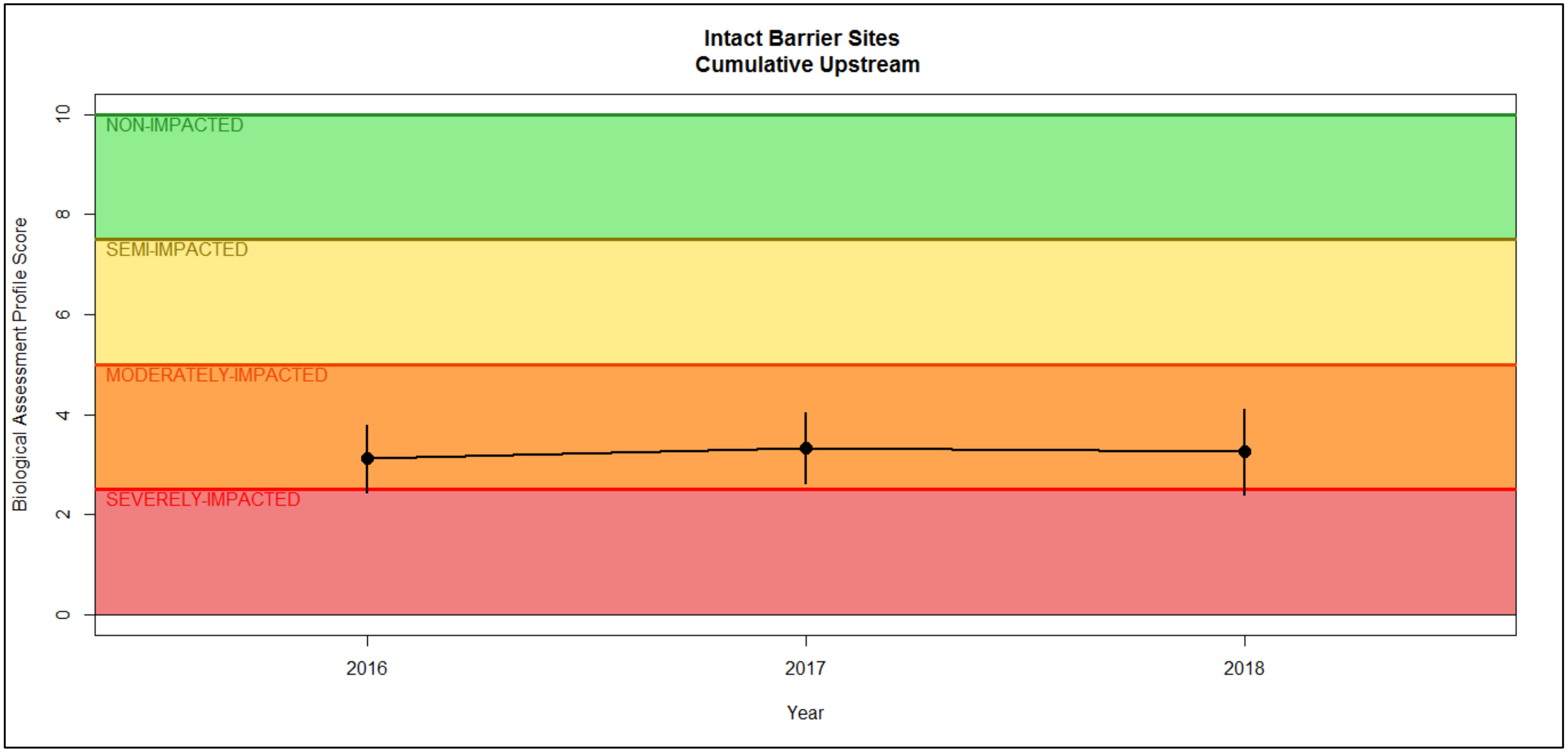

Figure 22: Cumulative Water Quality BAP data representing upstream impounded sites Browns Pond, Furnace Brook-Maiden Ln Dam, and Sawkill-Annandale Dam. Our data shows impounded systems within the Hudson Watershed are strongly impacted, regardless of geographic location.

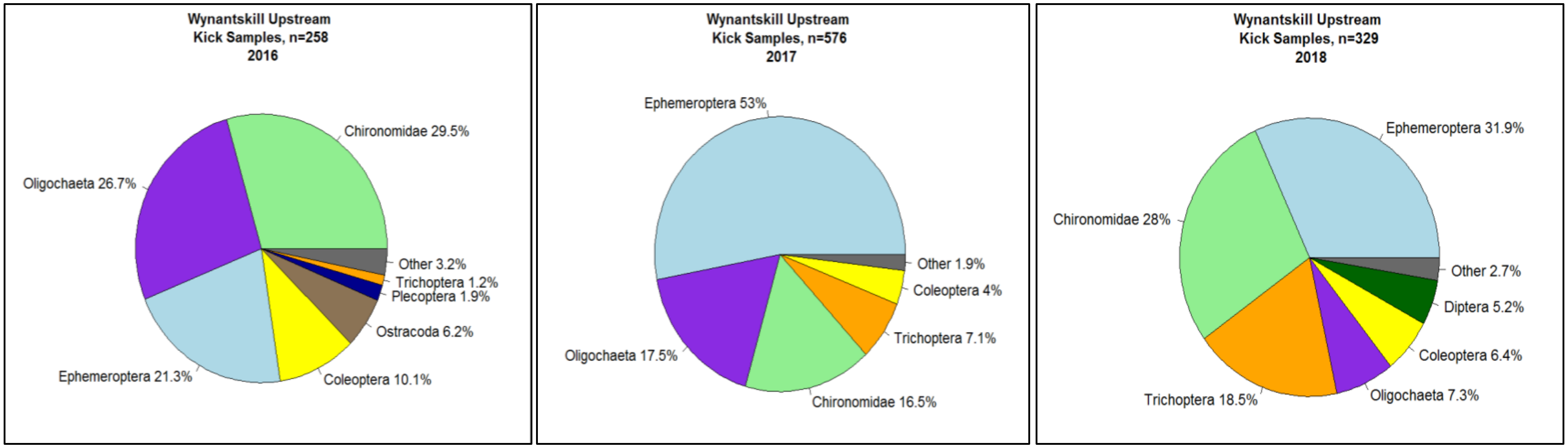

Figure 23: Aquatic invertebrate community composition at Wynantskill, upstream location. Barrier removal recovery trend shows benthic taxa: Oligochaeta and Ostracoda being replaced by more common lotic taxa: Ephemeroptera, Trichoptera, Coleoptera, and Diptera.