# Water-Quality Assessment of Two Slow-Moving Sandy-Bottom Sites on the Saw Mill River, New York.

Barbara E. Warkentine[1, 2,*] and Joseph W. Rachlin[2]

**Abstract** – We selected 2 sites on the Saw Mill River and conducted biological assessments of water quality using macroinvertebrate composition. Assessment metrics used were: Shannon-Weiner diversity, evenness, species richness, Hilsenhoff biotic index (HBI), Ephemeroptera–Plecoptera–Trichoptera richness (EPT), and non-Chironomidae and Oligochaete (NCO) richness. Water temperature, pH, conductivity, total dissolved solids, dissolved oxygen, and water flow and velocity were not significantly different across sites. Shannon-Weiner diversity values were 2.32 (evenness = 0.20) for Chappaqua and 2.68 (evenness = 0.31) for Hawthorne. Species and NCO richness for Chappaqua were 49 and 22, respectively, and for Hawthorne were 44 and 23, respectively. HBI was 7.99 for Chappaqua and 7.69 for Hawthorne. Both sites had equal EPT values of 5. Based on macroinvertebrate assessment indices, we classified water quality at these sites as non-impacted.

## Introduction

Land-use changes that result from urbanization are considered to be among the major impacts to rivers and streams throughout the world (Davies et al. 2010, Paul and Meyer 2001, Tran et al. 2010). Meyer et al. (2005) coined the phrase "urban stream syndrome" to describe the impacts associated with urbanization including decreases in biodiversity, water quality, and populations of sensitive organisms. This term also associates urbanization of streams with changes in hydrology and habitats (Walsh et al. 2005). While urbanization can lead to increased nutrient-loading, erosion of stream banks and substrates, and a reduction in overall water quality, it also impacts macroinvertebrate-community composition (Moore and Palmer 2005). In Moore and Palmer's (2005) study of agricultural and urban headwater streams in Maryland, macroinvertebrate diversity was greater at agricultural sites than at urban sites. In their study of 3 North Carolina Piedmont streams, Lenat and Crawford (1994) also found decreases in both the diversity and abundance of macroinvertebrate communities associated with particular land-use practices. Taxa richness and biotic-index values indicated poor water quality at their urban sites as compared to fair water quality at their agricultural sites. They also observed that chironomid groups were dominant in the agricultural streams but oligochaete groups were dominant in urban streams. Davies et al.'s (2010) study on streams in southeastern Australia found that urban streams had much lower

[1]SUNY Maritime College, Science Department, 6 Pennyfield Avenue, Bronx, NY 10465-4198. [2]Laboratory for Marine and Estuarine Research (La MER), Department of Biological Sciences, Lehman College of CUNY, 250 Bedford Park Boulevard West, Bronx, NY 10468-1589. *Corresponding author - bwarkentine@sunymaritime.edu.

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macroinvertebrate family richness and fewer sensitive taxa when compared to naturally vegetated streams.

Analysis of stream-macroinvertebrate faunal compositions in conjunction with established protocols and indices has proven to be an important and useful approach in assessing water quality throughout the world (Davies et al. 2010, Guimaraes et al. 2009, Ndaruga et al. 2004, Tran et al. 2010). Many US government agencies employ these protocols (Barbour et al. 1999, Bode 1993, Jessup et al. 2005, Plotnikoff 1994) for continuous and long-term water-quality monitoring. Beginning in 1972, the New York Department of Environmental Conservation's Stream Monitoring Unit began amassing data on the condition of 17 drainages (Bode et al. 2004) including the Saw Mill River in Westchester County. This river, which was the target of our study, provided drinking water to many Westchester County residents until 1983, and now serves as a backup water supply (Rogers 1984). At river kilometer 5.1 (mile 3.2) of its 36.9-km (23-mile) length, there is a water-treatment plant that has been closed since 1984 (Rogers 1984), and a reservoir is located at river kilometer 20.9 (mile 13). Given the potential use of this river as a potable water source, it is critical to evaluate the water quality at upstream sites close to its source. To determine water quality, we selected 2 sites for macroinvertebrate assessment: 1 at and 1 near the river's source. One of the sites had overhanging riparian vegetation and the other lacked similar vegetation, and we tested the hypothesis that the former would have better water quality than the latter. We also tested the hypothesis that better water quality is associated with less urbanization by comparing data collected during this study with previously published work on the river's highly urbanized terminus.

## Field-site Description

The Saw Mill River is an urban waterway, which originates at a 0.71-ha (1.75-ac) pond in the hamlet of Chappaqua in the town of New Castle, Westchester County, NY, and empties into the Hudson River in the city of Yonkers, Westchester County, NY. The Saw Mill River Parkway crisscrosses and runs parallel to the river along it's 36.9-km length. The river's course was altered in many areas to accommodate the parkway's route. In the 1920s, the river just north of Yonkers was diverted to a newly constructed, more westerly channel during construction of the southern portion of the parkway. In order to control flooding of adjacent properties and the parkway, the river section flowing through central Yonkers was directed through a concrete-lined channel. This design mediated flooding, but also reduced environmental quality. The last 244 m (800 ft) of the river's course to the Hudson River was, until 2011, completely underground. As the parkway was expanded further northward towards Chappaqua in the 1940s, much of the river's course was preserved, which allowed the river to wind its way southward towards Yonkers. However, there are a few northern regions where the river follows a straight channelized path and is periodically dredged for flood control, including the Chappaqua site evaluated in this study (USACE 2008). Land use adjacent to the river consists of a mixture of parkland, residential dwellings, and industrial and commercial enterprises. As a result of the modifications made to the river's course and the close proximity of the parkway to it, flooding of both the parkway and communities adjacent to this river is common during heavy rain events. We evaluated the river's biological composition to estimate water quality at 2 sites—Chappaqua (40º09'17.0"N, 073º46'40.1"W) near the river's northern most point, and Hawthorne (40º05'49.9"N, 073º48'39.0"W) located 6 miles downriver from Chappaqua (Fig. 1) The Chappaqua site is a long, channelized section of the river adjacent to a large parking lot on its east bank and a railroad on the west bank. It has no overhanging woody vegetation but does have herbaceous vegetation on

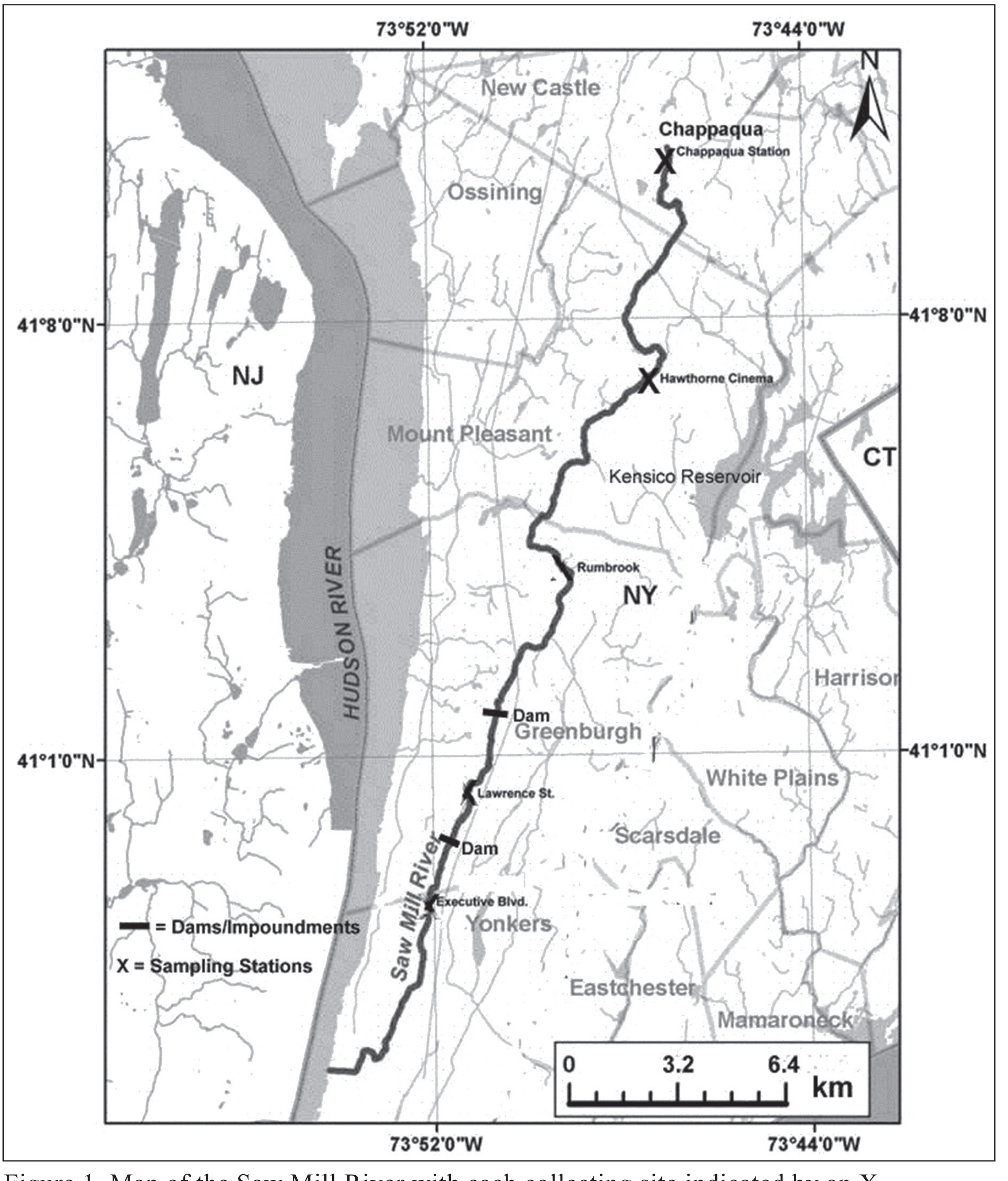


Figure 1. Map of the Saw Mill River with each collecting site indicated by an X.

both banks that separates the channel from the railroad track and the parking area. In comparison, the Hawthorne site has extensive overhanging woody vegetation with very little low-level herbaceous vegetation, and there is a commercial plant nursery on the east bank and a grassy berm and a dense tree line that separate the river and a parking lot on the west bank. The river follows a winding path at this site. We selected these sites because they represent 2 different river configurations and associated fauna which provided us with the opportunity to test the hypothesis that better water quality would be associated with a less disturbed habitat with a woody overstory (Hawthorne) than with a less wooded, more disturbed habitat (Chappaqua). Both sites were shallow non-navigational waterways with sandy bottom substrates and combinations of riffle and pool areas.

## Methods

We sampled macroinvertebrates during June and July 2009 on a weekly basis using a 0.25-$m^2$ Surber net with a 500-µm-mesh bag and cod end. On each sampling date, we collected 2 Surber samples—one associated with submerged vegetation and the other from an open sandy area. We pooled these samples and preserved them in 75% ethanol with 0.025% Rose Bengal. Coincident with each faunal sampling, we measured water temperature, pH, conductivity, and total dissolved solids (TDS) with a Hanna Probe Model 991300 (Hanna Instruments, Inc., Woonsocket, RI). We employed a LaMotte Dissolved Oxygen Kit model EDO code 7414 (LaMotte Company, Chestertown, MD), which uses the classic Winkler titration method to determine dissolved oxygen levels. We measured water flow with a Geopacks Flowmeter MFP51 (Mapmarketing, London, UK) and calculated velocity (m/s) using the relationship 0.000854C + 0.05 where C equals the number of counts/minute recorded by the flowmeter. In the laboratory, we separated macroinvertebrates from substrate material, and when possible, identified organisms to species level. We chose this approach because Resh and Unzicker (1975), in their review of data from a number of biomonitoring studies, found that different tolerance values were assigned to an organism if it was classified at the generic rather than species level. Lenat and Resh (2001) presented a number of examples that supported species-level identifications and proposed that going to this level improved the ability to assign water quality values based on macroinvertebrate presence. We used various keys to identify the organisms (Jokinen 1992, Peckarsky et al. 1990, Pennak 1989, Sompson and Bode 1980). We used chironomid-head capsules to identify these organisms to the species level. We used using a dissecting microscope at 80X magnification followed by closer examination with a compound microscope at 400X magnification to study the configuration of their mandibles and labial plates, which are important characteristics used for chironomid identification (Simpson and Bode 1980). Voucher specimens of each species were preserved for permanent storage.

For our faunal analyses, we excluded Diptera pupae, unidentifiable Coleoptera, and egg masses. Water quality assessment metrics consisted of the Shannon-Weiner diversity index, evenness, species richness (SPP), Hilsenhoff biotic index (HBI;

Hilsenhoff 1982), Ephemeroptera–Plecoptera–Trichoptera richness (EPT), and non-Chironomidae and Oligochaete (NCO) richness. These are the standard indices used for net sampling in shallow, sandy substrate streams (Bode et al. 2002). We determined overall water quality for each site by constructing a biological-assessment profile. This method standardizes the index values onto a 0–10 scale, where 0 indicates very poor water quality or severely impacted, and 10 indicates very good water quality or non-impacted (Bode et al. 2002). We used the formulas given in Bode et al. (2002) for slow, sandy bottom streams to convert the raw SPP, HBI, EPT, and NCO values to scale values for use in developing the biological-assessment profile for each site. We used PAST (V2.17) (Hammer et al. 2001) to perform ANOVAs, Student's *t*-test, diversity, evenness, and diversity profiles. The alpha value for all statistical tests was set at 0.05.

## Results

None of the physical variables we measured showed a significant difference between sites (Table 1). Mean values were based on 8 measurements per site. Both sites had high dissolved oxygen levels that, when considering the average water temperatures, placed them at or above the 85% oxygen-saturation level.

We identified a total of 2218 individual organisms distributed among 49 taxa at the Chappaqua site and 909 individuals distributed among 44 taxa at the Hawthorne location (Table 2). Oligochaetes were the most abundant organisms collected at both areas. They accounted for 48.3% and 39.5% of the samples at the Chappaqua and Hawthorne sites, respectively. Tubificid worms were 4.6 times more abundant at Chappaqua ($n = 975$) than at Hawthorne ($n = 213$). The insect family Chironomidae, while accounting for 40.6% of the Chappaqua fauna, only accounted for 22.9% of the organisms taken from Hawthorne. Table 2 shows the variation we observed in the chironomid species taken from each site. The non-biting midge *Tribelos jucundum* was the most abundant chironomid collected from the Chappaqua site. No chironomid species emerged as dominant in our collections from the Hawthorne site. Amphipods and isopods collectively represented 26.3% of the organisms collected at Hawthorne but only 0.1% of the collection taken from Chappaqua. Ephemeroptera, Simuliidae, and the bivalve *Sphaeriun* were relatively common at the Chappaqua site but were not present in the samples from Hawthorne.

Water-quality index values for both locations are presented in Table 3. Shannon-Weiner diversity values were 2.32 and 2.68 for Chappaqua and Hawthorne,

Table 1. Physical and chemical variables (mean ± standard deviation) for the Chappaqua and Hawthorne sites on the Saw Mill River (June–July 2009). ECS = conductivity, TDS = total dissolved solids, and DO = dissolved oxygen.

| Locations | Temp. (°C) | pH | ECS (µS cm) | TDS (mg/L) | DO ( mg/L) | Flow (counts/min) | Velocity (m/s) |
|---|---|---|---|---|---|---|---|
| Chappaqua | 19.1 ± 1.5 | 7.42 ± 0.2 | 652.0 ± 160.0 | 326.1 ± 80.1 | 8.3 ± 0.8 | 128.0 ± 46.4 | 0.16 ± 0.04 |
| Hawthorne | 17.9 ± 1.5 | 7.58 ± 0.2 | 662.6 ± 111.2 | 331.2 ± 55.9 | 8.2 ± 0.3 | 259.4 ± 140.7 | 0.27 ± 0.10 |
| ANOVA | $P = 0.18$ | $P = 0.22$ | $P = 0.90$ | $P = 0.91$ | $P = 0.67$ | $P = 0.06$ | $P = 0.06$ |

Table 2. Total abundance of macroinvertebrate taxa sampled from the Chappaqua and Hawthorne sites on the Saw Mill River (June–July 2009). *Taxomony could not be determined.

| Class | Order | Family | Genus | Species | Total abundance: Chappaqua | Total abundance: Hawthorne |
|---|---|---|---|---|---|---|
| Oligochaeta | Tubificida | Tubificidae | *Tubifex* | | 975 | 213 |
| | Haplotaxida | Naididae | *Nais* | | 77 | 116 |
| | Lumbriculida | Lumbriculidae | *Lumbriculus* | *variegatus* (Muller) | 19 | 30 |
| Malacostraca | Amphipoda | Gammaridae | *Gammarus* | | 2 | 89 |
| | Isopoda | Asellidae | *Caecidotea* | | | 150 |
| | | Unidentifiable | | | 1 | |
| Gastropoda | Basommatophora | Physidae | *Physella* | | 23 | 24 |
| | | Planorbidae | *Gyraulus* | *parvus* (Say) | 1 | 6 |
| | | Ancylidae | *Ferrissia* | *rivularis* (Say) | 2 | 18 |
| Bivalvia | Veneroida | Pisidiidae | *Sphaerium* | | 58 | |
| | Collembola | Isotomidae | *Isotomurus* | | 1 | |
| | Trichoptera | Lepidostomatidae | *Lepidostoma* | | | 1 |
| | | Hydropsychidae | *Cheumatopsyche* | | 10 | 1 |
| | | Hydropsychidae | *Hydropsyche* | | | 1 |
| | | Philopotamidae | *Wormaldia* | | 1 | 2 |
| | | Hydroptilidae | *Hydroptila* | | 2 | 1 |
| | | Hydroptilidae | *Orthotrichia* | | 2 | |
| | Ephemeroptera | Baetidae | *Centroptilum* | | 57 | |
| | Coleoptera | Elmidae | *Stenelmis* | | 7 | 6 |
| | | Dytiscidae | *Copelatus* | | | 1 |
| | | Unidentifiable | | | | 2 |
| | Odonata | Gomphidae | | | | 2 |
| | Diptera | Chironomidae | *Polypedilum* | *fallax* (Johannsen) | 53 | 5 |
| | | Chironomidae | *Polypedilum* | *illinoense* (Malloch) | 127 | 30 |
| | | Chironomidae | *Polypedilum* | *scalaenum* (Schrank) | 1 | |
| | | Chironomidae | *Polypedilum* | *convictum* (Walker) | 1 | |
| | | Chironomidae | *Dicrotendipes* | *neomodestus* (Malloch) | 14 | 5 |
| | | Chironomidae | *Dicrotendipes* | *nervosus* (Staeger) | 15 | 5 |
| | | Chironomidae | *Nanocladius* | *rectinervis* (Kieffer) | 90 | 40 |
| | | Chironomidae | *Tribelos* | *jucundum* (Walker) | 255 | 30 |
| | | Chironomidae | *Cryptochironomus* | *fulvus* Johannsen | 45 | 12 |
| | | Chironomidae | *Nilotanypus* | *fimbriatus* (Walker) | 28 | 6 |
| | | Chironomidae | *Nilothauma* | *babiyi* (Rempel) | 52 | 5 |
| | | Chironomidae | *Phaenopsectra* | *dyari* (Townes) | 14 | 17 |

Table 2, continued.

| Class | Order | Family | Genus | Species | Total abundance | |
|---|---|---|---|---|---|---|
| | | | | | Chappaqua | Hawthorne |
| | | Chironomidae | *Parakiefferiella* | | 95 | 26 |
| | | Chironomidae | *Eukiefferiella* | *discoloripes* (Goetghebuer) | 27 | 12 |
| | | Chironomidae | *Rheotanytarsus* | *exiguous* (Johannsen) | | 1 |
| | | Chironomidae | *Brillia* | *flavifrons* (Johannsen) | | 4 |
| | | Chironomidae | *Dicrotendipes* | *caelum* Townes | 66 | 2 |
| | | Chironomidae | *Chironomus* | *decorus* (Johannsen) | 8 | |
| | | Chironomidae | *Sublettea* | *coffmani* (Roback) | 6 | |
| | | Chironomidae | *Goeldichironomus* | | 4 | |
| | | Chironomidae | *Cricotopus* | *bicinctus* (Meigen) | 2 | |
| | | Chironomidae | *Paratanytarsus* | | 1 | |
| | | Chironomidae | *Camptocladius* | | 1 | |
| | | Chironomidae | *Thienemanniella* | | 1 | |
| | | Chironomidae | *Sympotthastia* | | 2 | |
| | | Ceratopogonidae | *Serromyia* | | 2 | |
| | | | Pupae | | 33 | 8 |
| | | Tipulidae | *Antocha* | | 1 | 1 |
| | | | *Tipula* | | 1 | |
| | | Simuliidae | *Simulium* | | 21 | |
| | | | Pupae | | | 1 |
| | | Empididae | *Hemerodromia* | | 4 | 5 |
| | | Syrphidae | *Chrysogaster* | | | 1 |
| Hirudinea | Arhynchobdellida | Erpobdellidae | *Mooreobdella* | *fervida* (Verill) | | 2 |
| | | Glossiphoniidae | *Gloiobdella* | *elongata* (Castle) | 1 | |
| | Rhynchobdellida | | *Helobdella* | *stagnalis* (L.) | 8 | |
| Trepaxonemata | Neoophora | Planariidae | | | 1 | 5 |
| Hydrozoa | Anthoathecatae | Hydridae | | | | 3 |
| | Neuroptera | Sisyridae | *Climacia* | | | 1 |
| Arachnida | Sarcoptiformes | | | | 1 | 1 |
| Maxillopoda | Calanoida | | | | | 4 |
| Nematodes* | | | | | 6 | 3 |
| Sponge* | | | | | | 1 |
| Fish larvae* | | | | | 1 | |
| Egg mass* | | | | | | 2 |
| | | | | TOTAL | 2218 | 909 |

respectively. A Student's *t*-test to compare the variances associated with these diversity values (Hutchenson 1970, Zar 1974) indicated a significant difference ($t$ = 6.2637, df = 2000.7, $P < 0.001$). Figure 2 graphically depicts the diversity profiles for both sites. The 2 curves cross far to the left ($\alpha \approx 0.2$), which confirms the non-comparable nature of the diversities for these two sites as indicated by the Student's *t*-test. The sharp decline in diversity in the area of $0.0 \leq \alpha \leq 1.0$ indicates that the diversity for each site was influenced by the presence of rare species (Leinster and Cobbold 2012). At $\alpha = 1$, our analysis indicated an effective number of species (ENS) of 10 at Chappaqua and 14.5 at Hawthorne. These values were further confirmed mathematically by using the relations ENS = $e^{H'}$ (Jost 2006). Species diversity for both sites leveled off at approximately $\alpha = 2.5$.

Evenness was 50% higher (E = 0.31) at the Hawthorne site than at the Chappaqua site (E = 0.20) (Table 3). However, because evenness values were closer to 0 than to 1, uniform species representation was not evident. Species richness was 49 at Chappaqua and 44 at Hawthorne, as shown by density at $\alpha = 0$ (Fig. 2). HBI was slightly higher at Chappaqua (7.99) than at Hawthorne (7.69) (Table 3). Both sites had EPT values of 5. Neither site had any Plecoptera, Hawthorne only had

Table 3. Water-quality index values for the Chappaqua and Hawthorne sites on the Saw Mill River (June–July 2009). SPP = species richness, HBI = Hilsenhoff biotic index, EPT = Ephemeroptera–Plecoptera–Trichoptera richness, and NCO = non-Chironomidae and Oligochaete richness.

| Location | Shannon diversity | Evenness | SPP | HBI | EPT | NCO |
|---|---|---|---|---|---|---|
| Chappaqua | 2.32 | 0.20 | 49 | 7.99 | 5 | 22 |
| Hawthorne | 2.68 | 0.31 | 44 | 7.69 | 5 | 23 |

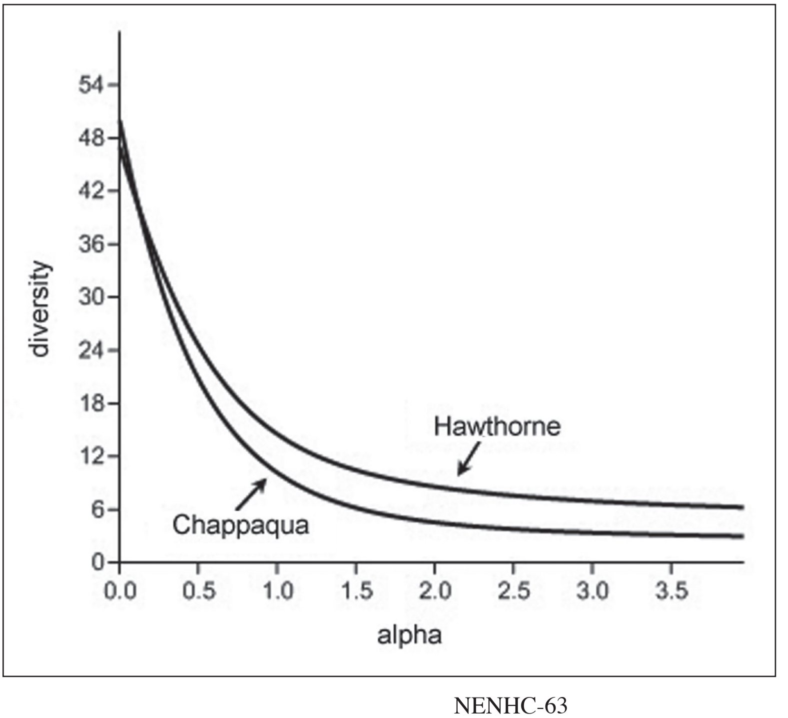


Figure 2. Shannon-Weiner diversity profiles for Chappaqua and Hawthorne sites on the Saw Mill River (June–July 2009).

Trichoptera, and Chappaqua had both Trichoptera and Ephemeroptera (Table 2). There was virtually no difference in NCO richness values between the sites.

The scaled water-quality index values for SPP, HBI, EPT, and NCO richness and the mean value for these 4 indices are presented in Table 4. Figures 3 and 4 show the biological-assessment profiles of these scaled values for Chappaqua and Hawthorne. Species richness and NCO index values for Chappaqua indicate non-impacted water quality (Fig. 3). However, HBI indicated that this section of the river was moderately impacted, and EPT classified the site between moderate and slightly impacted. The average of these index values (7.59) produced an overall result of non-impacted water quality (Fig. 3). The biological-assessment profile for Hawthorne was very similar to the profile for Chappaqua (Fig. 4). Despite a slightly higher average index value (7.71), we also assessed water quality at Chappaqua as non-impacted.

Table 4. Scaled water-quality index values and mean values for the Chappaqua and Hawthorne sites on the Saw Mill River (June–July 2009). SPP = species richness, HBI = Hilsenhoff biotic index, EPT = Ephemeropter–Plecoptera–Trichoptera richness, and NCO = non-Chironomidae and Oligochaete richness.

| Location | SPP | HBI | EPT | NCO | Mean |
|---|---|---|---|---|---|
| Chappaqua | 10 | 3.35 | 7 | 10 | 7.59 |
| Hawthorne | 10 | 3.85 | 7 | 10 | 7.71 |

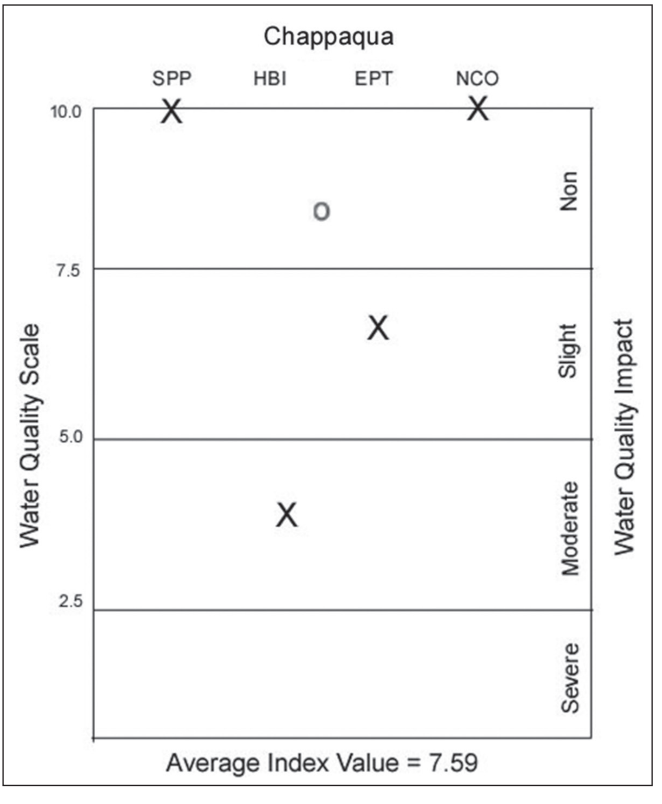


Figure 3. Water–quality impact profile for Chappaqua. X = scaled values for each index value. o = average of scaled index values.

## Discussion

Modifications to the river's course and continuous development along its banks are factors that could impact water quality, species composition, and stream flow (Wall et al. 1998). Impervious surfaces adjacent to river systems increase nutrient-rich and chemical-containing runoff to the system (Paul and Meyer 2001). Studies conducted by Rogers (1984), Phillips and Hanchar (1996), Riva-Murray et al. (2002), and Bode et al. (2004) found high concentrations of metal and organic compounds in samples taken from the highly urbanized southern Yonkers portion of the Saw Mill River. Rogers (1984) reported that heavy metal, nutrient, and synthetic organic-compound concentrations were significantly higher at Yonkers sites than in samples taken from northern areas in Chappaqua and Hawthorne. Bode et al. (2004) reported that this section of the river was classified as severely impacted in 1992. Data gathered in 1993 indicated that water quality in this section of the river showed slight improvement, and it was reclassified as being moderately impacted (Riva-Murray et al. 2002). Follow-up studies conducted from 1997 to 1999 showed no further improvement (Bode et al. 2004, Riva-Murray et al. 2002). Bode et al. (2004) reported that assays of crayfish tissue detected high concentrations of lead and 5 organic compounds that exceeded current thresholds for concern. All 4 studies attributed the decline in water quality over the river's course to runoff input from commercial and industrial properties along the river.

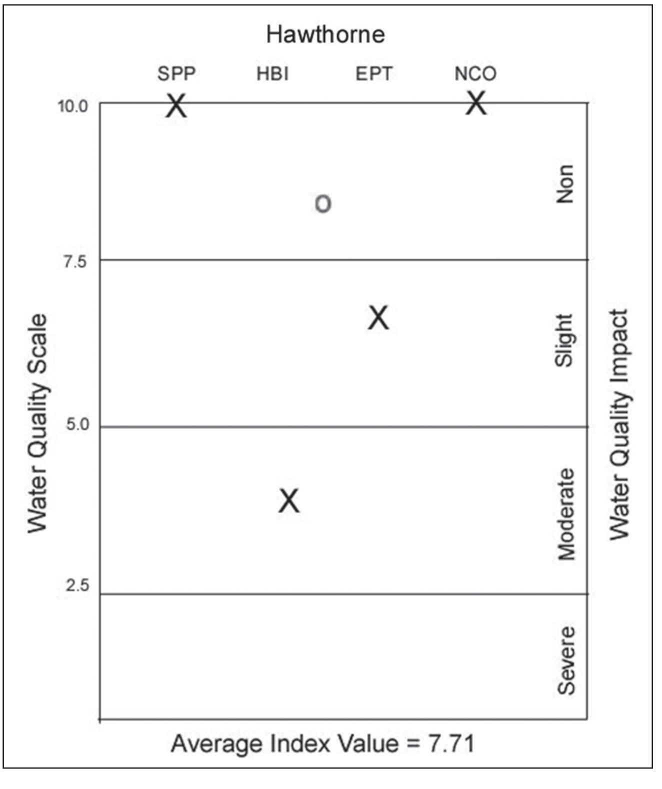


Figure 4. Water-quality impact profile for Hawthorne. X = scaled values for each index value. o = average of scaled index values.

In this study, we found that the water-quality conditions in the river's northern section at Chappaqua and Hawthorne were similar. These 2 sites had overall water-quality values that indicated they were non-impacted (Figs. 3, 4). The channelized configuration associated with the Chappaqua site didn't influence water quality when compared to the Hawthorne site. This result supports the findings of Riva-Murray et al. (2002), which showed that channel shape was not a driving factor in the water quality at sites in Yonkers.

Our classification of water quality at the less-urbanized sites in Chappaqua and Hawthorne as non-impacted supports the hypothesis that water quality would be better at these northern sites than at the highly urbanized Yonkers locations, which were never shown to be above the level of moderately impacted by the aforementioned investigators.

The dominance of tubificid worms and chironomids at both northern sites (Table 2) may be indicative of high concentrations of heavy metals in the sediment. Riva-Murray et al. (2002) found an abundance of these taxa in the Yonkers section of the Saw Mill River, which was found to have heavy loads of metals and hydrocarbons. Further studies are needed in order to determine if any of these pollutants affect the composition of the benthic fauna at the 2 northern sites.

The presence of the chironomid *Tribelos jucundum* in large numbers at the Chappaqua site and as the second-most abundant chironomid at the Hawthorne site (Table 2) is representative of the lentic nature of these 2 areas. Both sites had very low flow (Table 1). *Polypedilum illinoense*, which was the second-most abundant chironomid at Chappaqua and was well represented at Hawthorne, is tolerant to a wide variety of environmental conditions (Simpson and Bode 1980, Rae 1989). Many of the other chironomids collected from both sites are also considered tolerant or facultative species (Bode et al. 2002).

The number of amphipods and isopods at the Hawthorne site was in complete contrast to the virtual lack of these at the Chappaqua site. The gammarid amphipod, *Gammarus*, is a facultative species; however, the isopod, *Caecidotea*, is commonly associated with waters containing high concentrations of organic chemicals and low dissolved oxygen levels (Heitzman et al. 2011). While we didn't detect low oxygen levels at our sites, there may be an influx of organic compounds from the commercial plant nursery that is adjacent to the west bank of the river at this site.

Both sites had a mixture of facultative, sensitive, and pollution-indicator species albeit not necessarily the same species. Our results indicated that the overall diversities at the 2 sites were significantly different. When we converted each diversity-index value to effective number of species (ENS; Jost 2006), we found that the Hawthorne site had a species count of 14.5 compared to 10 for Chappaqua at an $\alpha$ value of 1 (Fig. 2). This result shows that there is a 31% decrease in species diversity at the latter site. This large difference in diversity indicates that fewer but highly dominant species are present in the community (Jost 1966).

Despite the difference in species composition and diversity between the 2 northern sites, our finding of no difference in water quality between sites refutes the hypothesis that the disturbed Chappaqua site would have poorer water quality than the more intact, wooded Hawthorne site. The channelized configuration and adjacent railroad and parking facility abutting the rivers banks in Chappaqua seemed to have little impact on water quality at the Chappaqua site when compared to the findings for the Hawthorne site. The lack of overhanging riparian vegetation apparently didn't influence water quality. Thus our hypothesis that water quality at Chappaqua and Hawthorne would be influenced by overhanging riparian vegetation was refuted.

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