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Toxicity of sediments in eight urban stormwater management ponds: bioassessment by oligochaete community metrics used in the sediment quality triad

Guillaume Tixier[a,*], Quintin Rochfort[b], Lee Grapentine[b], Jiri Marsalek[c], Michel Lafont[d] and Régis Vivien[e]

[a] Independent Consultant, 360 Torrance St, Unit 810, Burlington, ON L7R 2R9, Canada
[b] Water Science and Technology Branch, Environment and Climate Change Canada, 867 Lakeshore Road, Burlington, ON L7S 1A1, Canada
[c] Department of Civil, Environmental and Natural Resources Engineering, Luleå University of Technology, SE-971 87 Luleå, Sweden
[d] Univ Lyon, Université Claude Bernard Lyon 1, CNRS, ENTPE, UMR 5023 LEHNA, F-69622, Villeurbanne, France
[e] Swiss Centre for Applied Ecotoxicology (Ecotox Centre), EPFL ENAC IIE-GE, 1015 Lausanne, Switzerland
*Corresponding author. E-mail: gtvoi@hotmail.com

ABSTRACT

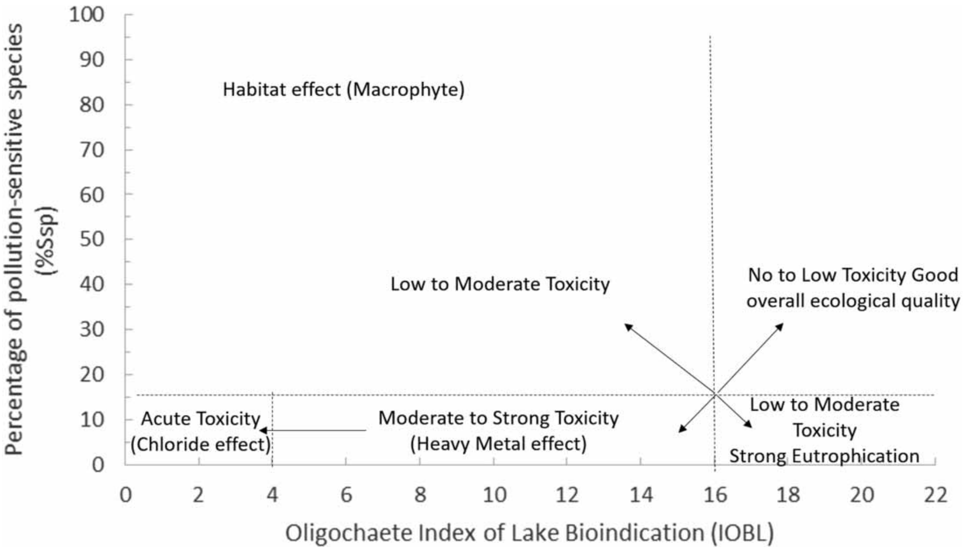
Implemented for decades as part of the 'best management practices (BMPs)' for controlling urban runoff impacts on receiving waters, stormwater management ponds (SMPs) have been increasingly viewed as potential habitats for urban wildlife. However, since SMPs are subject to a lot of environmental constraints, research toward assessing their ecological quality and their actual benefits as habitats for biota is needed. In this study, the sediment toxicity of eight SMPs located in Southern Ontario, Canada was assessed using the sediment quality triad (SQT) approach. Sediment samples were collected for chemical, ecotoxicological and biological analyses. An oligochaete-based index approach (Oligochaete Index of Lake Bioindication and percentage of pollution-sensitive species) was used as the biological endpoint and integrated into a weight-of-evidence approach to assessing the general sediment quality of the ponds. Our results showed that (i) heavy metals in the sediment and (ii) chloride concentrations in the sediment interstitial water caused detrimental effects on the ecological quality of the sediments in the ponds studied. The oligochaete indices applied in this study showed value as biological endpoints to be integrated into the SQT and used for setting up sediment ecological quality goals.

Key words: benthos, Oligochaete, sediment quality, sediment toxicity, stormwater pond management

HIGHLIGHTS

- An oligochaete-based index approach was used as a line of evidence in the SQT to assess the ecological quality of eight stormwater ponds in Southern Ontario.
- High concentrations of chloride in pore water and of heavy metals in the sediment matrix were linked to detrimental effects.
- The oligochaete-based index approach was used to set up ecological quality goals for stormwater pond sediment.

GRAPHICAL ABSTRACT



1. INTRODUCTION

Stormwater management ponds (SMPs) have been introduced into stormwater management and widely implemented in many countries since the late 1960s as one of the best management practices (BMPs) for controlling urban runoff peaks and reducing the risk of flooding in downstream areas (Chocat *et al.* 2001; Marsalek *et al.* 2005). The early ponds provided stormwater storage needed for runoff peak 'shaving' and reduced the cost of runoff conveyance, but environmental concerns, including stormwater pollution control, were considered low priorities (APHA 1981). Further experience with, and research into, SMPs led to the realization that the ponds should be designed as multipurpose facilities generally serving to control both the stormwater's quantity and quality (Whipple 1979). The main process for enhancing stormwater quality was identified as stormwater settling, which 'removed' (immobilized) not just stormwater sediment but also the adsorbed metals, hydrocarbons and nutrients (U.S. EPA 1983).

In the following decades, the design of SMPs has further evolved and the list of SMPs benefits in contemporary guidance documents includes such aspects as flood control, prevention of excessive erosion and undesirable changes in stream morphology, protection of water quality by stormwater treatment, provision of ecosystem services (carbon sequestration and biodiversity), cultural services (recreation, education and visual amenities) and provision of opportunities for stormwater reuse (MOE 2003; Taguchi *et al.* 2020). With the introduction of low impact development (LID), it was also acknowledged that SMPs have inherent limitations, which need to be considered when developing stormwater management plans at the catchment level: (i) SMPs modify (balance) the runoff discharges entering ponds but do not attenuate significantly the runoff volume; (ii) SMPs contribute to stormwater heating by exposing the pond water surface to solar radiation (Van Buren *et al.* 2000a, 2000b); and (iii) in regions with seasonal ice, snow and use of road salts in winter road maintenance, ponds tend to accumulate chloride at concentrations exceeding the acute toxicity threshold (Marsalek 2003) and furthermore, chloride changes the partitioning of heavy metals in benthic sediments by increasing the bioavailable dissolved fraction (Warren & Zimmerman 1994; Bäckström *et al.* 2004; Reinosdotter & Viklander 2007).

In general, SMPs can be designed as elements of green infrastructure, but their performance may be compromised by the risk of 'unintended consequences' of pond operation, as reported by Taguchi *et al.* (2020). There are two competing interests: planning SMPs as quasi-natural water bodies and the reality of ponds serving as treatment facilities accumulating contaminated sediments. In the former case, SMPs are viewed as new urban aquatic habitats, which are, however, severely constrained by multiple environmental stressors imposed by the urban catchment, including the highly modified hydrological regime (flow discharges), increased temperatures of runoff (Van Buren *et al.* 2000a) and elevated concentrations of suspended solids, nutrients, chloride (Dugan *et al.* 2017), heavy metals (Marsalek *et al.* 2006), bacterial contaminants (Crawford *et al.* 2010) and many legacy and emerging organic pollutants (Flanagan *et al.* 2021). As a consequence, the benefit of SMPs and constructed wetlands, which share most characteristics with ponds, as wildlife habitats have been largely questioned (Bishop *et al.* 2000; Bäckström *et al.* 2004; Roy *et al.* 2008; Clevenot *et al.* 2018; Hale *et al.* 2019). Studies showed that the biota in SMPs was not significantly different from the one in natural ponds, therefore suggesting that SMPs could act as important habitats, helping to support biodiversity in urban environments (Le Viol *et al.* 2009; Stephansen *et al.* 2016). Moreover, some studies focusing on the impact of common pollutants such as heavy metals and road salts on benthic communities in SMPs have shown no impact on taxonomic richness or diversity, and only contradicting results so far on benthic community structure (Stephansen *et al.* 2016; Sun *et al.* 2019). Most of these *in situ* studies investigating the ecological potential of SMPs have focused on near-shore sampling in shallow and/or vegetated habitats. Though fine sediments accumulating at the bottom of SMPs which mirror the quality of runoff and sediments entering the ponds (Pitt 2003) can represent 'hot spots' of inorganic and organic contaminants (Marsalek *et al.* 2006; Flanagan *et al.* 2021). These fine sediments can be potentially harmful to biota depending on their bioavailability and ease of potential remobilization under specific environmental conditions (Semadeni-Davies 2006). As a consequence, there were concerns voiced that SMPs had the potential to act as ecological traps (Robertson & Hutto 2006; Hale *et al.* 2015). Such concerns created a need to assess the ecological risk of SMP sediments and provide some ecological quality guidance for their management, particularly because of the dearth of data on the ecological quality of SMPs (Snodgrass *et al.* 2008; Tixier *et al.* 2011a; Clevenot *et al.* 2018). In assessing the quality of fine sediments, the methods using multiple lines of assessment, like the sediment quality triad (SQT) employing *in situ* biological indicators (e.g. the benthic macroinvertebrate community structure) have inherent advantages in comparison with the approaches relying solely on chemical analyses in the determination of the bioavailability of pollutants in sediments and the associated risk of ecosystem damages (Chapman *et al.* 1991).

Traditional benthic macroinvertebrate community structure approaches used within the SQT typically require comparison to reference sites. This limits the applicability of the method to urban SMPs because of the difficulty of finding suitable reference sites for these constructed habitats. Furthermore, the largely ubiquitous nature of the benthic macroinvertebrates in SMPs can result in a lack of bioassessment power (Bishop *et al.* 2000; Rochfort *et al.* 2000; Grapentine *et al.* 2008; Wik *et al.* 2008; Tixier *et al.* 2011a; Hassall 2014). The analysis of oligochaete community structure is well adapted for assessing the biological quality of SMP sediments. Though infrequently used in benthic studies (Chapman 2001; Tixier *et al.* 2011a), oligochaetes are among the most diverse and abundant taxa found in sediments. In preliminary research using the oligochaete species diversity and species-specific pollution sensitivities, some metrics have shown that examination of the oligochaete community can produce substantive information in the ecological risk assessment of fine sediments in SMPs (Lafont *et al.* 2007; Tixier *et al.* 2011b). For example, when integrated into the SQT, as a biological line of evidence, an oligochaete-based index of the sediment biological quality, provided strong evidence of the seasonal toxicity of chloride (deicing agent) in the sediment pore water of an SMP facility receiving heavily polluted runoff from one of the busiest highways in North America (annual average daily traffic > 450,000 vehicles) (Tixier *et al.* 2012). The main objective of the present study was to extend the application of the oligochaete index-based methodology integrated into the SQT to SMPs receiving runoff from a wider range of pollution, including a mix of various residential and rural watersheds. The specific objectives were to (i) analyze the response of the oligochaete indicators to a range of pollution perturbations; (ii) identify the substances potentially harmful to aquatic biota in the studied facilities and (iii) assess the ecological risk of the sediment in the SMPs.

2. MATERIALS AND METHODS

2.1. Study sites

We selected eight SMPs in Southern Ontario, Canada. Six of them were located within the Greater Toronto Area (GTA), the most populated region in Canada and two were in Peterborough (ON) just east of the GTA. The climate in this Southern Ontario region is humid continental, with mean annual temperatures in Toronto and Peterborough of 9.0 and 7.5 °C, annual precipitation of 831 and 870 and annual mean snowfall of 122 and 138 cm, respectively. All the SMPs studied were constructed wet ponds, with varying characteristics of pond surface, volume and drainage area (Table 1). Ponds also varied in age, with the mill pond being the oldest, built in the 1800s, but adopted as a stormwater control facility in the 1960s. The most recent pond was Pond H, constructed in 2005 to control runoff from a newly developed residential area. Land uses in pond catchments varied, ranging from a busy motorway corridor for Sites A and B, to mostly residential areas in the cases of Ponds C, D, E and F and mixed residential and rural lands at Sites G and H, with Pond G also receiving runoff from a nearby gas station.

Table 1 | General characteristics of the ponds studied

Pond	Terra view	Willowfield	RH19-9	Mill	W33-01	W46-01	P2	P16
Name	A	B	C	D	E	F	G	H
Location	Toronto	Toronto	Richmond Hill	Richmond Hill	Whitby	Whitby	Peterborough	Peterborough
Surface (m^2)	5,500	3,500	1,570	20,000	6,150	8,400	3,370	5,170
Max vol. (m^3)	9,800	5,000	4,200	30,000	7,500	14,000	5,500	6,890
Max depth (m)	2.5	2.5	1.5	3.0	1.5	3.0	1.8	3.5
Year built	1999	1999	2001	1800s (1960s)	1996	2002	1977	2005
Inlets	2	2	1	2	1	1	1	2
Outlets	1	1	1	1	1	1	1	1
Drainage (ha)	9 (highway) 11 (residential)	7+ discharge from A	24.6	552	49	59.6	46.8	45.2
Imperv. (%)	95 (highway) 40 (residential)	40	41	40	45	45	44	52
Land use	Highway + residential	Residential	Residential	Residential	Residential	Residential	Residential + rural	Residential + rural

vol., volume; Imperv., imperviousness.

2.2. Sampling procedures

Four sampling sites were selected along the main path of flow through each stormwater pond, with the exception of Pond C, where only three sites were selected because of its smaller size. Altogether, 31 sampling sites were selected and sampled once during each of the three seasons, corresponding to spring, summer and fall for a total of 93 samples. Not all the ponds were sampled during the same calendar year and the sampling period extended from 2008 to 2010. Before each sample collection, temperature, dissolved oxygen (DO), electrical conductivity, turbidity and pH were recorded at 10 cm above the bottom sediment surface using a YSI multi-probe. Multiple water samples were then collected at 10 cm above the sediment surface using a peristaltic pump for further water chemistry analyses. On each sampling occasion, three replicate sediment samples were collected for oligochaete analysis using a push-corer (65 mm diameter). The top 10 cm of each sediment core were individually extruded into a plastic jar and immediately preserved with 100 mL of 10% formalin. Three additional sediment samples were collected using a Petite Ponar Grab sampler (15 cm × 15 cm) and placed in a large bucket. These samples were then homogenized in the bucket before being distributed into glass jars for sediment chemistry analysis, a large plastic bag for pore water chemistry, and three small replicate bags for ecotoxicological tests. Pore water was separated from the sediment in the laboratory by centrifugation at 3,000 rpm for 15 min using a Sorvall Legend™ T/RT centrifuge. Pore water was not analyzed during the summer campaign in Ponds A, B and G.

2.3. Physico-chemical analyses

Water and sediment samples were analyzed for the 16 USEPA Priority polycyclic aromatic hydrocarbons (PAHs) by liquid–liquid extraction and gas chromatography/mass spectrometry analysis. Total and dissolved (in water and pore water samples) metals Cd, Cr, Cu, Fe, Ni, Pb and Zn were analyzed by inductively coupled plasma mass spectrometry equipped with a mass-selective detector. Metals in sediment were analyzed using inductively coupled argon plasma collision/reaction mass spectrometry equipped with a mass-selective detector (CRC-ICP-MS) after a hotblock digestion with concentrated nitric acid and hydrochloric acid (*aqua-regia*) for 5 h at 85 °C. Total Kjeldahl nitrogen (TKN) and total phosphorus (TP) concentrations in water and sediment were determined by a spectrophotometer after digesting samples with hot, concentrated sulfuric acid. Sediment samples were also analyzed for moisture, particle size distribution, total organic carbon (TOC) and organic content. Particle size distributions were determined on freeze-dried samples by weighing fractions separated on a sieve tower. All sediment less than 63 μm in diameter was considered 'fines' (silt + clay fractions) and the weight of fines was expressed as a percentage of the total dry sediment weight. Sediment moisture levels were determined by loss of weight on drying sediment samples; weights were determined by using a Denver Electronic balance XP-300. TOC was determined by combustion using a Leco TOC Analyzer. Organic content was measured as ash-free dry mass (AFDM), calculated as the loss of mass on ignition from dry sediment samples burned at 550 °C in a muffle furnace for 1 h. TOC concentrations in water were determined by infrared combustion and CO_2 detection on a Shimadzu TOC analyser. Chloride concentrations in water were determined by ion chromatography using a Dionex ICS2000 system equipped with an IonPac AS15 column.

2.4. Ecotoxicity tests

Sediment toxicity was measured in the laboratory using survival and growth bioassays with the amphipod *Hyalella azteca* (28 days) and the mayfly *Hexagenia* spp. (21 days). Sediment toxicity tests were run only on samples collected during the spring sampling campaigns. The tests were conducted under environmentally controlled conditions in replicates of three for pond samples and five replicates for the laboratory control sediment (Long Point, Lake Erie). The *H. azteca* test was conducted starting with the random addition of 15 organisms (aged 2–10 days) per beaker. On day 28, the contents of each beaker were sieved through a 250 mm screen and the surviving amphipods were counted. Amphipods were then dried at 60 °C for 24 h and dry weights were measured. Initial weights were considered negligible. The *Hexagenia* spp. test was conducted starting with the random addition of 10 pre-weighed nymphs (5–8 mg wet weight/nymph). On day 21, the contents of each jar were wet sieved through a 500 mm screen and surviving mayfly nymphs were counted. Nymphs were then dried at 60 °C for 24 h and dry weights were measured. Initial dry weights were calculated using the following previously determined relationship: Initial dry weight = (wet weight + 1.15)/7.35. Final growth was determined as final dry weight minus initial dry weight. Detailed descriptions of the test methods are given in Milani *et al.* (2013).

2.5. Oligochaete identification

Each replicate core sample (in total, $93 \times 3 = 279$ core samples) was individually sieved and rinsed through a 250 μm mesh screen. Each sample was transferred to and homogenized in a Marchant Box. Cells from a 10×10 grid were randomly chosen until a maximum of one hundred oligochaete specimens per sample were sorted using a stereomicroscope. Oligochaetes were then mounted on a microscope slide in a mixture of glycerine and lactic acid and identified to the lowest possible taxonomic level under a compound microscope, according to Kathman & Brinkhurst (1998).

2.6. Metrics and statistics

Taxonomic richness and abundance of oligochaetes determined for each sample were used to calculate the Oligochaete Index of Lake Bioindication (IOBL), assessing sediment metabolic potential (Lafont *et al.* 2012) and calculated as follows:

$$\mathrm{IOBL} = S + (3x)\log_{10}(D + 1)$$

where S is the number of oligochaete taxa and D is the oligochaete density per 0.1 m^2. The IOBL value per sampling site and per season ($n = 93$) was calculated by averaging the IOBL values of the three replicate core samples. The index varies from 0 to >20 and is regarded as an indicator of the metabolic potential of lake sediments (or the capacity of sediments to mineralize organic matter). The higher the IOBL, the greater the metabolic potential (see Table 2).

Using the list of oligochaete pollution-sensitive species available in Lafont *et al.* (2012), we calculated an abundance ratio of oligochaete pollution-sensitive species to total oligochaete from the three replicate core samples. This ratio was expressed as

Table 2 | Interpretation of IOBL values in five class-criteria model of the metabolic potential of lake sediments according to Lafont *et al.* (2012)

IOBL value	Metabolic potential
>15	Very high
10–15	High
6.1–9.9	Medium
3.1–6	Low
≤3	Very low

the percentage of pollution-sensitive species (%Ssp) per sampling site and per season ($n = 93$). The %Ssp has been used as an index of biological quality of the sediment (see Table 3).

Sediment toxicity endpoints for each sampling site were expressed as average growth rates (mg/21 days for *Hexagenia* spp. and mg/28 days for *H. azteca*) and average survival rates after 21 days for *Hexagenia* spp. and 28 days for *H. azteca*.

Chemical variables, i.e. PAH compound concentrations in the sediment and heavy metal compound concentrations in the overlying water, the pore water and the sediment were analyzed by performing separate principal component analyses (PCAs) to identify the major axes of variation in the contamination patterns. PCA scores extracted from the axes representing the most significant variance were used as surrogate variables to represent pollution levels. PCAs were also conducted on heavy metal and PAH concentrations in sediment normalized by TOC, calcium (Ca), AFDM and fines to account for key sediment geochemical components (potential ligands). All environmental data were $\log_{10} + 1$ transformed prior to PCA.

Spearman rank correlation analyses were used to examine relationships between biological variables: %Ssp and IOBL; and environmental variables: DO, TOC, Ca, AFDM, fines, chloride and surrogate variables from the PCA on heavy metals and PAHs. The Spearman rank correlation test was preferred to the Pearson correlation test to account for monotonic relationships between environmental variables and biological variables. A Bonferroni correction was applied to the standard significance threshold so that correlations were marked significant at $p < 0.0018$.

Contour plots (x, y and z) were used to examine the main relationships found between the biological variables used as predictor variables (x, y) and environmental variables (z). Data were smoothed to a fitted surface using the distance-weighted least squares procedure with a 0.25 stiffness coefficient (McLain 1974). A non-parametric Kruskal–Wallis ANOVA was used to test for differences in environmental variables of interest among samples grouped according to their IOBL and %Ssp values. All statistical analyses and contour plots were carried out using Statsoft Statistica 8.0 and ordinations were made with PC-ORD 5.0.

A weight-of-evidence approach combining the lines of evidence of the triad, including the oligochaete metrics, was used to assess the general quality of the water and sediment in the ponds. Except for chloride, integration of the results for the chemistry and toxicity tests was made according to a five class-criteria model: Bad, Poor, Moderate, Good and Very Good quality (see details in Supplementary Annex 1) The integration of the data from the heavy metal and PAH concentrations in the sediment was made by comparisons with the Ontario sediment quality assessment guidelines (Fletcher *et al.* 2008) in the form of a sediment quality index (Grapentine *et al.* 2002b). The integration of the data from the heavy metal concentrations in water was made by comparison with the provincial water quality objective (PWQO) values for protection of aquatic life (MOEE 1999). The integration of the data from the toxicity tests was based on the 20–50% endpoint reduction benchmarks, giving more weight to the survival tests as acute endpoints (Grapentine *et al.* 2002a; see Supplementary Annex 1). The integration of the oligochaete metrics was made accordingly to the five class-criteria model of the IOBL and the %Ssp for the determination of *in situ* sediment metabolic potential and biological quality (Lafont *et al.* 2012; see Tables 2 and 3). In the interpretation of the weight-of-evidence approach, we gave more weight to the *in situ* biological responses reflected in this study by the oligochaete metrics (Chapman & Anderson 2005).

3. RESULTS

3.1. Physico-chemical parameters

The physico-chemical parameters measured in the overlying water, the pore water and the sediment of the ponds are summarized in Tables 4–6. The ponds showed a significant inter-pond variability of contamination by PAHs and heavy metals

Table 3 | Interpretation of the percentage of oligochaete pollution-sensitive species in five class-criteria model for the biological quality of lake sediments according to Lafont *et al.* (2012)

% Sensitive species	Biological quality of the sediment
>50%	Very good
>20 to ≤50%	Good
>10 to ≤20%	Medium
>5 to ≤10%	Poor
≤5%	Bad

Table 4 | Minimum–maximum values of the physical and chemical parameters measured in the sediment of the ponds studied (four sites per pond × three campaigns, $n = 12$; except Pond C: three sites × three campaigns, $n = 9$; total number of observations $n = 93$)

	Units	LEL	A	B	C	D	E	F	G	H
Fines	%	–	0–96.6	32.2–98.8	77.9–96.3	93.4–99.4	0–99.3	62.4–98.1	0–93.1	46.6–100
TOC	mg C/g	10	18.1–106	13.9–90.9	4.4–20.6	26.1–88.3	0.1–20.21	11.0–34.5	0.8–34.3	0.7–11.6
AFDM	mg/cm^3	–	1.5–16.0	0.2–5.5	0.5–1.7	0.2–1.3	0.2–8.1	0.1–4.8	1.2–6.4	0.1–1.0
TP	μg/g P	600	378–1,040	579–1,310	658–862	821–986	368–943	601–949	289–1,050	695–859
TKN	μg/g N	550	317–3,070	952–4,340	1,460–2,920	4,030–5,710	246–2,500	1,560–3,690	126–3,150	788–1,520
Cd	μg/g	0.6	0–3.5	0–2.9	0–0.6	0.4–0.8	0–3.9	0–0.5	0.1–2.1	0–0.4
Cr	μg/g	26	57.6–168	26.8–99.3	28.8–40.5	22.3–34.1	12.3–42.7	17.4–31.4	4.4–37.2	17.5–28.7
Cu	μg/g	16	117–295	37.1–203	20.8–42.1	28.5–40.2	7.8–47.0	15.2–32.5	4.5–48.1	12.5–22.0
Fe	mg/g	20	19.9–34.7	16.7–32.5	20.5–27.2	18.1–26.2	5.07–30.9	12.2–24.5	5.1–15	16.4–24.3
Ni	μg/g	16	22.8–30.4	13.3–30.9	17.4–22.7	14.0–18.9	4.5–25.7	9.4–17.1	2.5–16.5	9.9–15.3
Pb	μg/g	31	60.3–138	23–112	12.1–19.8	17.2–23.8	7.3–18.8	9.6–22.8	2.0–32.3	3.0–7.5
Zn	μg/g	120	356–907	126–621	66.7–117	120–160	26.3–137	50.9–149	17–244	33.3–53.4
Anth.	μg/g	0.22	0–0.21	0–0.27	0–0	0–0	0–0	0–0	0–9.6	0–0
B.(a)ant.	μg/g	0.32	0–0.57	0–1.26	0–0.39	0–0.19	0–0.58	0–0.58	0–15.0	0–0
B.(a)pyr.	μg/g	0.37	0–0.99	0–2.01	0–0.65	0–0.26	0–0.94	0–0.85	0–13.6	0–0
B.(b)flu.	μg/g	–	0–1.00	0–3.71	0.29–1.13	0.24–0.49	0.29–1.49	0–1.59	0–17.4	0–0
B.(ghi)per.	μg/g	0.17	0–0.37	0–0.88	0–0.74	0–0	0–0.75	0–0.98	0–5.18	0–0
B.(k)flu.	μg/g	0.24	0–0.42	0–1.19	0–0.42	0–0	0–0.51	0–0.53	0–6.3	0–0
Chrys.	μg/g	0.34	0–0.90	0–2.1	0.3–1.1	0.2–0.39	0.25–1.28	0–1.43	0–15.1	0–0
Flu.	μg/g	0.75	0–3.07	0.6–7.18	0.51–2.29	0.33–0.61	0.42–2.38	0–2.54	0.49–37.7	0–0
I.pyr.	μg/g	0.20	0–0	0–0.19	0–0.80	0–0	0–0.88	0–0.93	0–6.0	0–0
Phen.	μg/g	0.56	0–1.39	0–2.27	0.18–0.75	0–0.21	0–0.77	0–0.82	0–42.6	0–0
Pyr.	μg/g	0.49	0–2.92	0.46–5.83	0.39–1.59	0.27–0.49	0.32–1.87	0–2.00	0.41–28.5	0–0

LEL, lowest effect level, Anth., anthracene; B.(a)ant., benzo(a)anthracene; B.(a)pyr., benzo(a)pyrene; B.(b)flu., benzo(b)fluoranthene; B.(ghi)per., benzo(g,h,i)perylene; B.(k)flu., benzo(k) fluoranthene; Chrys., chrysene; Flu., fFluoranthene; I.pyr., indeno(1,2,3-cd)pyrene; Phen., phenanthrene; Pyr., pyrene.

Table 5 | Minimum–maximum values of the physical and chemical parameters measured in the pore water of the ponds studied (four sites per pond × three campaigns, $n = 12$; except Pond C: three sites × three campaigns $n = 9$ and Ponds A, B and G: four sites × two campaigns, $n = 8$; total number of observations $n = 81$)

	Units	PWQO	A	B	C	D	E	F	G	H
TOC	mg/L	–	7.53–46.3	5.91–18.5	7.99–22.2	8.54–25.1	4.83–14.9	6.27–23.5	4.24–10.8	5.14–47.1
TP	mg/L P	–	0.32–1.89	0.21–0.97	0.11–0.80	0.45–3.10	0.00–0.48	0.29–1.31	0.03–1.29	0.11–2.77
TKN	mg/L N	–	5.52–69.1	3.25–17.2	6.96–13.7	3.99–16.4	2.27–12.6	5.59–41.6	1.21–28.5	4.72–32.2
Chloride	mg/L	230[a]	215–4,690	214–2,220	82–1,260	120–174	98–940	168–4,300	48–191	23–151
Cd	μg/L	0.2	0.0–0.0	0.0–0.0	0.0–0.7	0.1–1.1	0.0–0.8	0.0–0.7	0.0–0.4	0.0–1.0
Cr	μg/L	8.9[b]	13.6–51.4	11.2–33.7	0.2–9.0	3.9–9.7	0.4–7.6	1.5–11.7	0.4–25.0	2.9–9.3
Cu	μg/L	5	27.8–145	13.2–98.4	0.0–13.6	8.2–23.0	0.0–15.5	0.0–20.6	2.1–72.4	5.4–15.8
Fe	mg/L	0.3	1.2–73.1	6.1–34.1	11.8–17.2	11.7–23.6	0.7–18.0	1.7–23.1	0.4–22.4	9.0–19.6
Ni	μg/L	25	0.5–7.4	0.3–8.7	1.4–5.4	2.5–4.9	2.6–5.7	0.0–5.5	1.8–13.6	3.3–10.8
Pb	μg/L	5	14.6–80.4	15–56.3	0.6–6.1	3.8–9.9	0.0–10.6	0.0–13.3	0.5–38.1	0.0–11.8
Zn	μg/L	20	71.6–404	28.5–218	6.9–38.5	21.6–57.0	12.3–49.7	6.7–79	6.5–160	11.3–48.4

PWQO, provincial water quality objectives.

[a]Chronic effect threshold (U.S. EPA 1988).

[b]Value for trivalent chromium (Cr III) (MOEE 1999). Values for heavy metals represent total concentrations.

in the sediment and by heavy metals in the overlying water and pore water. Among the 16 USEPA priority PAHs analyzed in the sediment, altogether seven were detected in all the ponds. Except in Pond H, where none were detected, PAHs occurred in maximum concentrations above the lowest effect level (LEL) (Fletcher *et al.* 2008) in most ponds (Table 4). The lowest concentrations were in Pond D (from below the LEL to 1× the LEL) and the highest in Pond G (from 30 to 76× the LEL). PCA on PAH concentrations revealed that 78.65% of the cumulative variance was significantly explained by the first component (Axis 1 = 65.4%) and the second component (Axis 2 = 13.2%). All seven PAHs detected in the ponds except anthracene were correlated negatively to Axis 1, while anthracene correlated positively to Axis 2. Heavy metals were found in the sediment in maximum concentrations above the LEL (Fletcher *et al.* 2008) in most ponds, especially in the case of Cu. The lowest maximum concentrations of Cu were found in Pond H (max. 1.4× the LEL) and the highest in Pond A (max. 18× the LEL). PCA on heavy metal concentrations showed that 72.7% of the variance was significantly explained by the first component (Axis 1). All metals but Cd were negatively correlated with Axis 1. Concentrations of nutrients TP, TKN and TOC in the sediment were generally high in all the ponds except Pond H, which showed the lowest levels. TP and TKN levels were highest in Pond D.

In the pore water (Table 5), heavy metals largely exceeded the Provincial Water Quality Objectives (PWQO) (MOEE 1999) for Cu, Fe and Zn. PCA on dissolved concentrations revealed that 58% of the cumulative variance was significantly explained by the first component (Axis 1 = 36%) and the second component (Axis = 22%). Cr, Cu, Pb and Zn were negatively correlated with Axis 1, while Cd, Fe and Mn were correlated with Axis 2. PCA on total concentrations revealed that 51.7% of the variance was significantly explained by the first component (Axis 1) negatively correlated with Cr, Cu, Fe, Pb and Zn. The concentrations of chloride in the pore water were highly variable between and within the ponds. The maximum concentrations were found in spring and greatly exceeded the U.S. EPA ambient water quality threshold of 230 mg/L (U.S. EPA 1988) in Ponds A, B, C, E and F.

In the overlying water (Table 6), DO was notably low in summer in Pond C and also in Ponds E and F, but at their deepest sites only. The maximum concentrations of Cu, Fe and Zn in the overlying water exceeded the PWQO in most of the ponds, except for Ponds C and D, which showed very little contamination by heavy metals. The maximum concentrations of Cu and Zn were comparatively high in Ponds A and B (4.5× to 8× the PWQO), however, Pond H showed extremely high values of all

Table 6 | Minimum–maximum values of the physical and chemical parameters measured in the overlying water of the ponds studied (four sites per pond × three campaigns, $n = 12$; except Pond C: three sites × three campaigns, $n = 9$; total number of observations $n = 93$)

	Units	PWQO	A	B	C	D	E	F	G	H
DO	%	–	37–127	13–167	5–114	59–165	12–150	7–185	95–150	41–115
pH	–	–	7.47–8.54	7.60–8.56	7.15–8.02	7.68–8.08	7.40–7.91	6.45–8.43	7.88–8.10	7.66–8.52
Cond.	mS/cm	–	0.62–6.52	0.50–2.79	0.78–4.66	0.87–1.10	0.70–1.47	0.82–11.30	0.53–0.73	0.35–1.72
Turb.	NTU	–	9–37	6–50	39–46	41–122	13–90	31–72	21–120	38–1,661
TOC	mg/L	–	3.98–9.94	3.89–6.57	3.86–6.98	3.43–5.31	3.55–6.15	2.96–14.80	2.68–4.49	3.34–6.34
TP	μg/L P	–	28–194	0–366	79–233	0–162	8–470	9–497	31–422	0–8,390
TKN	mg/L N	–	0.86–2.72	0.63–3.15	0.75–3.27	0.68–1.77	0.85–2.27	0.89–18.90	1.08–2.70	0.49–10.7
Chloride	mg/L	230[a]	140–2,160	121–805	52–1,390	123–155	89–270	158–3,570	47–65	3.0–29
Cd	μg/L	0.2	0.0–1.2	0.0–1.0	0.0–0.2	0.0–0.2	0.0–0.2	0.0–0.4	0–0.4	0.0–8.5
Cr	μg/L	8.9[b]	1.1–11.7	0.1–24.3	0.1–1.1	0.0–1.7	0.4–11.9	0.0–12.4	0.4–13.2	0.0–312
Cu	μg/L	5	0.0–42.0	0.0–39.8	0.0–2.5	0.0–2.1	0–16.2	0.0–21.7	1.1–17.9	0.0–266
Fe	mg/L	0.3	0.27–2.41	0.42–9.85	0.46–1.09	0.41–1.59	0.34–8.42	0.11–11.60	0.27–0.55	0.24–323
Ni	μg/L	25	0.0–32.1	0.0–19.4	0.1–6.1	0.0–1.5	1.0–8.4	0.0–6.2	0.0–6.3	0.0–195
Pb	μg/L	5	0.0–13.1	0.0–9.4	0.7–2.2	0.0–2.0	0.2–9.9	0.0–13.0	0.5–16.6	0.0–70.6
Zn	μg/L	20	16.5–89.9	9.8–192.0	2.2–7.7	1.7–10.9	3.5–56.3	3.3–103.0	5.7–86.2	0.0–542

PWQO, provincial water quality objectives; Cond., conductivity; Turb., turbidity.
[a]Chronic effect threshold (U.S. EPA 1988).
[b]Value for trivalent chromium (Cr III) (MOEE 1999). Values for heavy metals represent total concentrations.

the metals analyzed, from 8× the PWQO for Pb to >1,000× the PWQO for Fe. These extreme values were recorded during the summer sampling at the inlet of Pond H shortly after a rain storm. Slightly lower but consistent concentrations were also found at the other three sampling sites in Pond H during this campaign. Along with heavy metals, very high turbidity and high concentrations of TKN and TP were found in the water throughout the pond during this campaign. PCA on total metal concentrations revealed that 70.8% of the variance was significantly explained by the first component (Axis 1) negatively correlated with all eight metals analyzed. PCA on dissolved metal concentrations revealed that 63.6% of the variance was significantly explained by the first component (Axis 1 = 38%) and the second component (Axis 2 = 25.5%). Mn, Fe and Ni were positively and Pb negatively correlated with Axis 1, while Cd, Cr, Cu and Zn were negatively correlated with Axis 2.

3.2. Ecotoxicity tests

The results of the toxicity tests averaged by pond are shown in Figure 1(a) and 1(b). The tests generally showed a weak inter-pond variation of sediment toxicity. Pond A showed the highest toxicity of all the ponds. The survival rate of *Hexagenia* spp. in Pond A was reduced on average by almost 40% compared with the control sediment and we observed more than a 50% reduction at some sites. Pond F showed high toxicity variability, depending on the location sampled and a reduction of more than 20% in the survival rate of *H. azteca* was observed at one site. All the other Ponds (B, C, D, E, G and H) showed no acute toxicity (no sites showed a 20% or more reduction in survival endpoints compared with the control), though some chronic toxicity occurred. A reduction of more than 50% in the growth of *Hexagnia* spp. was found within Pond B and by more than 20% within Pond H. A reduction of more than 50% in the growth of *H. azteca* also occurred within Pond G.

3.3. Oligochaete metrics

A total of 14,321 oligochaete individuals belonging to 35 taxa were identified in the 279 core samples analyzed. The results of mean IOBL and mean %Ssp are shown in Figure 2. The IOBL showed considerable variation in Ponds A, B, C and F, which were also the ponds the most impacted by chloride. Ponds could be separated into two groups: (a) Ponds B, C and F with a mean IOBL < 10, showing very low to low metabolic potential; and (b) Ponds A, D, E, G and H with a mean IOBL >10, showing a medium to high metabolic potential. Based on the mean %Ssp, Ponds F and G were different from all the other ponds. Pond F showed the highest %Ssp (mean %Ssp 47), corresponding to good biological quality. Pond G showed a medium biological quality (mean %Ssp = 18). All the other ponds showed mean %Ssp <10 corresponding to poor or bad biological quality.

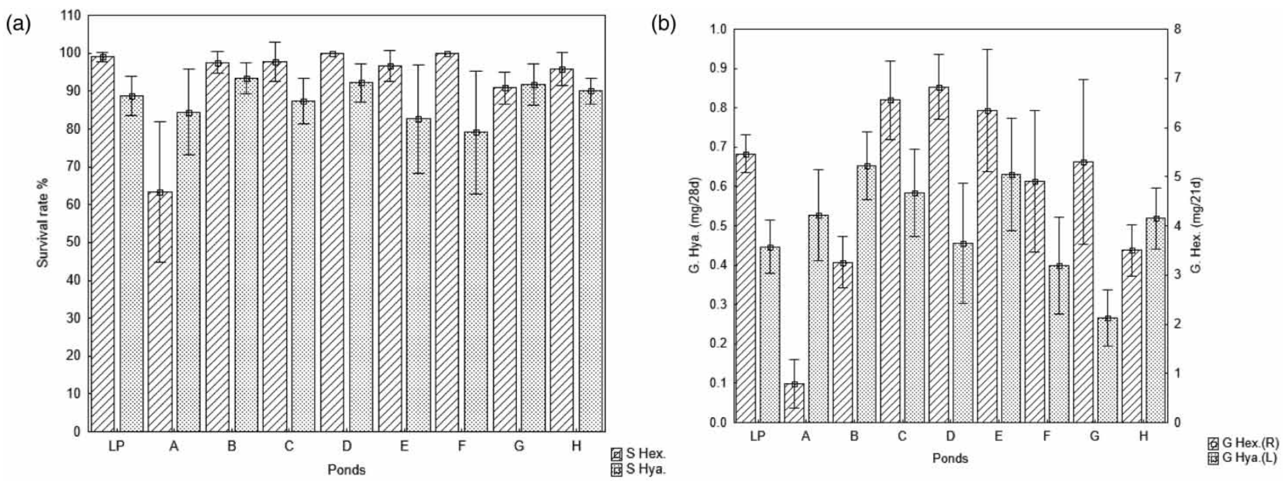


Figure 1 | (a) Results of ecotoxicity tests grouped by pond – mean survival rate (%) of *Hexagenia* spp. (S Hex.) and *H. azteca* (S Hya.) after 21 and 28 days, respectively (four sites per pond × three replicate samples, $n = 12$; except Pond C: three sites × three replicate samples, $n = 9$; LP (Long Point, Lake Erie) $n = 5$ replicate samples). Whisker: mean ± 0.95 confidence interval. (b) Results of ecotoxicity tests grouped by pond – mean growth rate of *Hexagenia* spp. (G. Hex.) and *H. azteca* (G. Hya.) after 21 (mg/21 days) and 28 days (mg/28 days), respectively (four sites per pond × three replicate samples, $n = 12$; except Pond C: three sites × three replicate samples, $n = 9$; LP (Long Point, Lake Erie) $n = 5$ replicate samples). Whisker: mean ± 0.95 confidence interval.

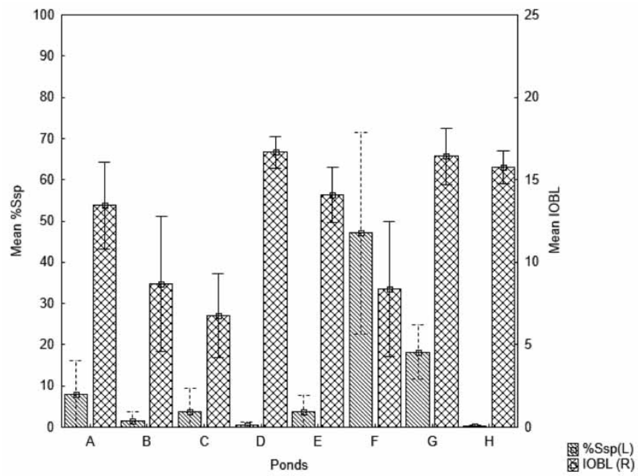


Figure 2 | Histogram of mean percentage of oligochaete pollution-sensitive species (%Ssp) and mean IOBL in the ponds (four sites per pond × three campaigns $n = 12$; except Pond C: three sites × three campaigns $n = 9$; total number of observations $n = 93$). Whisker: mean ± 0.95 confidence interval.

3.4. Spearman rank correlation

All significant relationships are displayed in Table 7. The strongest relationship was found between IOBL and the concentration of chloride in the pore water (Spearman rank correlation, $r = -0.532$, $n = 81$). The %Ssp was best correlated with the surrogate variable from the PCA on heavy metals in the sediment weighted by the AFDM (Spearman rank correlation, $r = 0.410$, $n = 93$).

3.5. Contour plots

Contour plots illustrated the variation of chloride concentrations in the pore water against IOBL and %Ssp in Figure 3(a) and the variation of heavy metal concentrations in the sediment against IOBL and %Ssp in Figure 3(b). According to the smoothing procedure, chloride concentrations ranging between 0 and 500 mg/L in the pore water (chloride reference conditions) were associated with IOBL values ranging from 6 to 22 and %Ssp ranging from 0 to 100. In the opposite data distribution tail, pore water chloride concentrations above 2,000 mg/L (chloride impacted conditions) were strictly associated with IOBL values < 4 and %Ssp < 20.

Table 7 | Spearman rank correlation for all significant relationships between environmental variables tested and biological variables (%Ssp and IOBL) (Bonferroni $p < 0.0018$)

%Ssp	Valid	*r*-value	*t*(*N* − 2)	*p*-value
SedMetal/AFDM	93	−0.410435	4.29361	0.000044
Fines (<63 μm)	93	−0.380479	−3.92472	0.000168
SedMetal	93	−0.359840	3.67910	0.000396
SedMetal/Ca	93	−0.357403	3.65053	0.000436
AFDM	93	0.333025	3.36918	0.001107
IOBL	**Valid**	***r*-value**	***t* (*N* − 2)**	***p*-value**
PoreCl	81	−0.531889	−5.58273	0.000000
SedMetal /TOC	93	−0.409151	4.27748	0.000047
WaterCl	93	−0.389648	−4.03600	0.000113
SedMetal/Ca	93	−0.323121	3.25709	0.001582

Note: SedMetal, SedMetal/AFDM, SedMetal/TOC and SedMetal/Ca are the surrogate variables from PCAs on corresponding concentrations of heavy metals in the sediments and their ratios with AFDM, TOC and Ca. The PCA results showed that the concentrations of heavy metals in the sediments and the ratios with AFDM, Ca and TOC increased with the decreasing PCA scores. To facilitate the understanding of the correlation matrix, PCA scores were converted to the opposite value. PoreCl and WaterCl are, respectively, the concentrations of chloride in the pore water and overlying water.

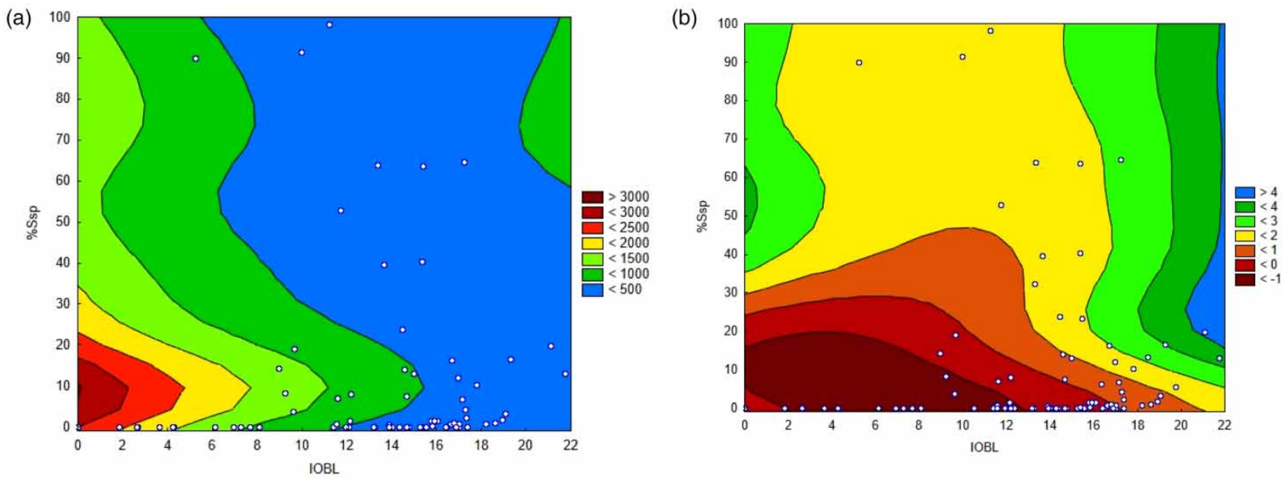


Figure 3 | (a) Contour plot of chloride concentrations (in mg/L) in the pore water against IOBL on the *x*-axis and percentage of oligochaete pollution-sensitive species (%Ssp) on the *y*-axis ($n = 81$). (b) Contour plot of heavy metal concentrations (represented as PCA score variable: SedMetal) against IOBL on the *x*-axis and percentage of oligochaete pollution-sensitive species (%Ssp) on the *y*-axis ($n = 93$).

With regards to heavy metals in the sediment, the highest concentrations (PCA scores < -1) were associated with IOBL $<$ 13 and %Ssp $<$ 15. In the opposite data distribution tail, the lowest concentrations of heavy metals in the sediment (PCA scores $>$ 2) were associated with IOBL values $>$ 16 and %Ssp $>$ 15 and in a more marginal way with IOBL $<$ 4 but %Ssp $>$ 40 since no data points were observed in the latter range. To test the significance of IOBL $>$ 16 and %Ssp $>$ 15 as threshold values corresponding to the lowest concentrations of heavy metals in the sediment, we performed a Kruskal–Wallis ANOVA using heavy metal PCA scores (SedMetal) as the independent variable and threshold values of 16 and 15, respectively, for IOBL and %Ssp as grouping variables (Group 1: IOBL $>$ 16 and %Ssp $>$15, Group 2: IOBL $<$ 16 and %Ssp $>$ 15, Group 3: IOBL $>$ 16 and %Ssp $<$ 15 and Group 4: IOBL $<$ 16 and %Ssp $<$ 15). According to the contour plot, we defined Group 1 as samples with the lowest heavy metal concentrations, Groups 2 and 3 as samples with moderate concentrations and Group 4 as samples with the highest concentrations. We found a significant difference in heavy metal concentrations among the four sample groups (Kruskal–Wallis ANOVA, $p < 0.0001$). Also, Group 1 had significantly lower heavy metal concentrations than Group 4 (multiple comparison test, $p < 0.0028$) but no significant differences were found between Group 1 and the other Groups 2 and 3.

4. DISCUSSION

The stormwater facilities studied showed a broad range of environmental conditions with a large variation of contamination by multiple contaminants, including chloride, heavy metals and PAHs, as shown by the results of chemical analyses. The integrated approach combined toxicity tests and analysis of *in situ* oligochaete community structure with the objective of determining whether or not the contamination had an impact on the biota and prevented it from prospering in the SMPs. Two types of contaminants were found to significantly influence the biological variables studied (i.e. IOBL and %Ssp): the chloride in the pore water and the heavy metals in the sediment and both cases were previously reported as indicating impacts on the biota in stormwater facilities (Bartlett *et al.* 2012a, 2012b; Tixier *et al.* 2012). No linear relationships were found between the heavy metal variables and the chloride concentrations; therefore, we concluded that they likely exerted separate impacts on oligochaetes in the ponds studied. Although we did not test for bioavailable forms of heavy metals in the sediment, we could not exclude a cumulative (synergistic) effect (Mayer *et al.* 2008). Elevated chloride concentrations in the pore water occurred seasonally during winter and spring in Ponds A, B, C, E and F and were caused by road salt applications in winter road maintenance (Marsalek 2003). However, not all the ponds were impacted by elevated concentrations of chloride and no seasonal increases in chloride concentrations in the pore water were observed in Ponds D, G and H. Either the drainage catchments of these ponds received negligible amounts of chloride or it did not accumulate in the pore water of these ponds. It is worth noting that Ponds G and H drained rural areas with lower road densities and less salt applied than residential areas or major highways (the case of Ponds A and B). Also, Pond D was the largest one in terms of surface area.

While some authors reported that SMP water surface area is an important factor determining pollutant retention rate in stormwater facilities (Starzec *et al.* 2005), that finding is unlikely to apply to chloride, which enters ponds as a gravity underflow because of the higher density of chloride laden runoff (Marsalek *et al.* 2000). Consequently, most of the incoming chloride load stays in bottom pond water layers and can easily partition into pond bottom sediment: note the maximum chloride concentration values in Tables 5 and 6, in the sediment interstitial water and bottom pond water, respectively: $C_{\mathrm{Cl\ max}} =$ 4,690 and 2,160 mg/L in Pond A and $C_{\mathrm{Cl\ max}} =$ 4,300 and 3,570 mg/L in Pond F. As suggested by Marsalek (2003), SMPs have distinct chloride regimes, comprising a period of chloride accumulation from late fall to late winter, followed by a period of chloride flushing out of the pond during the rest of the year. Other influential factors are the amount of road salt applied in the pond catchment annually; pond design (depth, geometry, wind reach, etc.), online or offline hydraulic configuration; intensity of annual pond flushing (defined here as the volume of annual inflow into the pond divided by the pond storage volume); and underwater topography (channelizing, berming, deep pools, etc.). The flushing of chloride from undisturbed pond bottom sediment would be caused by the diffusion of chloride from the sediment and would represent a much slower process than hydraulic flushing.

According to our results, the presence of high levels of chloride resulted in a drastic reduction of the IOBL. Both the abundance of oligochaetes and the percentage of sensitive species were expected to decrease at concentrations above 500 mg/L and be considerably reduced at concentrations above 2,000 mg/L, for which IOBL $<$ 4 and %Ssp $<$ 20. The episodic and acute effects of chloride on benthic biodiversity and on oligochaetes in particular, have been previously documented (Kefford *et al.* 2003; Tixier *et al.* 2012). Chloride concentrations varying abruptly (up to 25 times the background concentrations in Pond F to concentrations close to those defining brackish waters) can lead to toxicity for a large array of freshwater organisms (Van Meter *et al.* 2011). However, the biological recovery appeared to be rapid, following the decrease of chloride concentrations in the pore water in highly impacted ponds. For instance, in Pond B, the IOBL range from 4.0 in the spring, when concentrations of chloride in pore water were the highest, to 17.6 in the fall, when chloride concentrations were the lowest.

While chloride seemed to have a strong influence on the IOBL, the heavy metals in the sediment seemed to have a stronger influence on the %Ssp rather than on the IOBL. There was a significant negative correlation between the heavy metals in the sediment and %Ssp. The correlation was more significant when heavy metal concentrations were normalized by the organic AFDM content in the sediment. The negative correlation between %Ssp and heavy metals in the sediment first validated the list of species considered as pollution-sensitive. It is worth noting, therefore, that we did not find a significant correlation between the oligochaete total species richness and the heavy metals in the sediment. However, we did find a significant negative correlation between the abundance of oligochaetes and the heavy metals in the sediment. The correlation was more significant when heavy metal concentrations were normalized by the TOC. According to our results, the TOC and AFDM in the sediment could play a role in the sensitivity of the oligochaete community to the heavy metals by reducing their adverse effects. Though our study did not permit us to analyze either the effects of individual heavy metals or their bioavailability, it has been shown before that organic content is potentially a strong ligand to heavy metals, thereby reducing their bioavailability to benthos (Zhang *et al.* 2014).

Our results showed biota in SMPs can be subject to both acute and chronic toxicity, which reinforced the ecosystem resilience and resistance hypotheses (Tixier *et al.* 2011a).

With oligochaetes being linked to metabolic processes in the sediment, the IOBL as a metric of the oligochaete community diversity was referred to as a proxy of the sediment metabolic potential (Lafont *et al.* 2012) (see Supplementary Annex 2). We hypothesized before that, at the lowest levels of heavy metal concentrations in the sediment (preserved lakes) and under normal trophic conditions, both the IOBL and %Ssp would attain the highest values. On the opposite, heavily polluted lakes would result in low %Ssp and low IOBL, though it was previously observed that some oligotrophic lakes can show a low metabolic potential (low IOBL) without anthropogenic impairment (Lafont *et al.* 2012). The analysis of the species composition and percentage of sensitive species would then help discern between naturally low metabolic potential conditions (%Ssp high due to the absence of pollution) and low metabolic potential due to anthropogenic impairment (%Ssp low as a result of the pollution). We also hypothesized that under normal trophic conditions, the first signs of a gradual anthropogenic impairment would first result in a decrease in the percentage of sensitive species before a decrease in IOBL (Lafont *et al.* 2012). Indeed, sensitive species would first disappear, being replaced by more tolerant species, which would keep the IOBL high. Though as the impairment increases, more species would disappear leading to a decrease in the IOBL and the beginning of functional damage (i.e. reduction of sediment metabolic potential). Therefore, the %Ssp would be used as an early warning sign of the anthropogenic impacts on sediment quality (Tixier *et al.* 2011a; Lafont *et al.* 2012).

Unlike a sudden and drastic reduction of IOBL and %Ssp, in the case of episodic and acute toxic events related to the seasonal influx of high chloride concentrations in the pore water, the IOBL and %Ssp showed a gradual response to the increasing levels of contamination by heavy metals. Our results showed that samples with IOBL above 16 and %Ssp above 15 had significantly lower heavy metal concentrations than samples with IOBL below 16 and %Ssp below 15. Our data did not show enough evidence for verifying the hypothesis of an early impact of heavy metal contamination on high %Ssp, in part because our data set lacked sampling sites showing both high IOBL and high %Ssp. However, even though the differences were not found to be significant, results from the contour plot suggested that moderate contamination occurred at IOBL above 16 and %Ssp below 15. Similarly, sites with IOBL below 16 and %Ssp above 15 showed moderate contamination at best, which means that good quality conditions (i.e. least contaminated sediment) would not be attainable. Therefore, the combination of IOBL above 16 and %Ssp above 15 appeared to be a minimum threshold level that could serve as a suitable ecological quality goal for the sediment of urban SMPs. These results would be consistent with a previous study reporting that out of five small urban lakes investigated in France (Ulis, the Paris region), the only preserved one (i.e. not subject to polluted wet-weather inflows) showed IOBL = 16 and %Ssp = 39% (Lafont *et al.* 2012).

From the contour plot, conditions in which IOBL was below 16 but %Ssp above 40 were associated with either moderate (IOBL above 4) or light (IOBL below 4) sediment contamination. Sites showing IOBL below 16 but %Ssp above 40 were associated with Pond F, which uniquely showed a high density of a macrophyte cover, resulting in a very weedy bottom with high AFDM organic content in the sediment. It is hypothesized that the weedy bottom in this pond offered a less suitable sediment quality habitat for Tubificinae, thereby affecting the oligochaete total abundance, but also protected the sediment by lowering the bioavailability of heavy metals, which both resulted in increasing %Ssp. Such particular conditions in Pond F with low IOBL and high %Ssp would then resemble those of lakes with naturally low metabolic potential and uncontaminated sediment. However, low contamination extrapolations for IOBL below 4 and %Ssp above 40 from the contour plot should be regarded with caution, because they were not supported by actual data.

The weight-of-evidence analysis, illustrated by Table 8, showed the average result for each line of evidence, which means that the information is highly condensed and reduced and that the hypotheses and interpretations drawn are non-exclusive with respect to other factors that could contribute to the quality of the systems studied.

Pond A generally showed strong contamination by all pollutants analyzed (heavy metals, chloride and PAHs), which translated into very strong sediment toxicity in the lab testing. The *in situ* biological analysis, however, showed that in spite of a strong contamination, the sediment metabolic potential was still high (mean IOBL = 13.5) likely due to a low bioavailability of pollutants bound to strong ligands, such as TOC and AFDM, both present in high concentrations in the sediment. Also, Pond A was well designed to promote self-purification in this stormwater management system, with such features as a sediment forebay by the inlet and water exchange with an underground exfiltration vault and consequently displayed a strong longitudinal gradient of improving sediment quality conditions toward the outlet (Grapentine *et al.* 2008). The analysis of the oligochaete community confirmed the presence of Pristina species, indicative of groundwater exchanges, which are known to improve sediment quality (Lafont & Vivier 2006; Boulton 2007). Sensitive species (mean %Ssp = 8) indicated a poor sediment biological quality, suggesting a ligand effect becoming overwhelmed. Pond B, which was located immediately downstream of Pond A and received additional residential stormwater inputs, showed the second highest levels of the pollutants studied, but exhibited only a moderate toxicity in the laboratory testing. The metabolic potential, however, was lower than in Pond A (mean IOBL = 8.7) and %Ssp was very low (mean %Ssp = 2), which indicated a bad sediment biological quality, probably due to a higher bioavailability of heavy metals *in situ*, as TOC and AFDM were much lower in the sediment of Pond B than in Pond A. Pond C was slightly less contaminated by metals than Ponds A and B but was highly impacted by chloride and PAHs. We surprisingly found no toxicity in the toxicity tests, however, IOBL indicated a medium metabolic potential (mean IOBL = 6.8, the lowest value among the eight ponds) and very low %Ssp (mean %Ssp = 4) indicating a bad biological quality. Strong anoxic conditions found in this pond, especially during summer months, could have been a contributing factor for unsuitable *in situ* habitat for biota, probably enhancing *in situ* heavy metal toxicity, too. Pond D was one of the two least contaminated ponds showing no chloride contamination, low contamination by PAHs and moderately high contamination by heavy metals. There were no signs of sediment toxicity found in the laboratory testing and the metabolic potential was very high (mean IOBL = 16.7), while %Ssp remained very low (mean %Ssp = 1) indicating a bad biological quality. This outcome likely resulted from eutrophication, as the pond sediment showed the highest TKN level and the

Table 8 | Weight-of-evidence assessment of the ecological quality of the ponds studied by integrating the results of the SQT

Pond	Water		Porewater		Sediment			Biology		Scores
	Chloride	Metals	Chloride	Metals	Metals	PAHs	Ecotox	IOBL	%SSP	
A	Red 2	Red 4	Red 2	Red 4	Red 4	Red 4	Red 0	Green 3	Orange 1	Contamination: 20/20; Bio-ecotoxicological response: 4/12
B	Red 2	Red 4	Red 2	Red 4	Red 4	Red 4	Yellow 2	Yellow 2	Red 0	Contamination: 20/20; Bio-ecotoxicological response: 4/12
C	Red 2	Orange 3	Red 2	Red 4	Orange 3	Red 4	Blue 4	Yellow 3	Red 0	Contamination: 18/20; Bio-ecotoxicological response: 6/12
D	Blue 0	Orange 3	Blue 0	Red 4	Orange 3	Green 1	Blue 4	Blue 4	Red 0	Contamination: 11/20; Bio-ecotoxicological response: 8/12
E	Yellow 1	Red 4	Red 2	Red 4	Red 4	Red 4	Blue 4	Green 3	Red 0	Contamination: 19/20; Bio-ecotoxicological response: 7/12
F	Red 2	Red 4	Red 2	Red 4	Orange 3	Red 4	Orange 1	Yellow 2	Green 3	Contamination: 19/20; Bio-ecotoxicological response: 6/12
G	Blue 0	Red 4	Blue 0	Red 4	Red 4	Red 4	Yellow 2	Blue 4	Yellow 2	Contamination: 16/20; Bio-ecotoxicological response: 8/12
H	Blue 0	Red 4	Blue 0	Red 4	Yellow 2	Blue 1	Green 3	Blue 4	Red 0	Contamination: 11/20; Bio-ecotoxicological response: 7/12

Note: Contamination level scoring: red: highest (4), orange: strong (3), yellow: moderate (2), green: low (1), blue: lowest (0); except chloride red: high (2); yellow: moderate (1); blue: low (0). Maximum combined contamination level score ($2 \times 2 + 4 \times 4 = 20$). Bio-ecotoxicological response scoring (ecotoxicological tests and oligochaete indices): red: highest (0); orange: strong (1); yellow: moderate (2); green: low (3); blue: lowest (4); maximum combined bio-ecotoxicological score (3×4) = 12. Decreasing contamination level: A, B > E, F > C > G > D, H. Decreasing bio-ecotoxicological response: A, B > C, F > E, H > D, G.

highest minimum P concentration among all the ponds studied. Organic enrichment, especially in N and P, is known to support oligochaete abundance and especially of Tubificinae (Brinkhurst 1965) that would have resulted in a high IOBL, but a low %Ssp. Even though Pond E showed high contamination by heavy metals, PAHs and relatively high chloride concentrations in the pore water, it showed no signs of sediment toxicity in the laboratory testing and a high metabolic potential (mean IOBL = 14.1). However, the quality of sediment was not good enough to support sensitive species as %Ssp remained low (mean %Ssp = 4) indicating a bad biological quality that suggests that the beneficial effects of ligands were overwhelmed. Pond F was the second pond in the set showing signs of acute toxicity in the laboratory testing. Even though contamination by heavy metals or PAHs in the sediment was strong, it stayed in the same range as in Ponds C and D, showing no signs of toxicity. While Pond F was also highly impacted by episodic chloride shock loads in the spring, the sediment toxicity demonstrated in the laboratory might have also resulted from an untested (untargeted) contaminant at this location. Medium metabolic potential (mean IOBL = 8.4) and high %Ssp (mean %Ssp = 47) indicating a good biological quality were the results of the unique macrophyte-covered sediment and its consequence on habitability and bioavailability of pollutants discussed earlier. Pond G showed high concentrations of PAHs, the highest levels of all the ponds studied and high heavy metal contamination, but showed no signs of chloride contamination. The ecotoxicological tests revealed a moderate toxicity, which could have been attributed to PAHs even though we did not notice strong detrimental effects *in situ*. The metabolic potential was very high (mean IOBL = 16.4) and %Ssp was relatively high (mean %Ssp = 18) indicating a medium biological quality. The proximity of this pond to a river could have facilitated the pond colonization by clean water species through groundwater exchanges (a significant presence of *Pristina* spp. was observed), which would also improve the sediment quality. In spite of high concentrations of PAHs, Ponds G and D showed the highest bio-ecotoxicological ratio (8/12) of all ponds studied (Table 8). Pond G was also the only pond with IOBL and %Ssp mean values meeting the ecological quality objective goals we defined in this study (IOBL = 16; %Ssp = 15). The sediment in Pond H showed the least contamination by PAHs, heavy metals or chloride. TOC and AFDM were also the lowest of all ponds. The sediment showed slight toxicity in the laboratory testing and a very high metabolic potential (mean IOBL = 15.8), but surprisingly, very few sensitive species were present (mean %Ssp < 1) indicating a bad biological quality. Pond H was the youngest pond among all the ponds studied, having operated for just 5 years. The sediment layer being relatively new could probably explain why it showed the lowest level of contamination and ligand potential overall. However, the overlying water contained extremely high heavy metal concentrations during the summer campaign, when sampling took place after a storm event. This indicated that Pond H could receive highly contaminated overlying waters, especially by heavy metals. Even though the sediment was still relatively preserved, such flash toxic events could have prevented pollution-sensitive species from settling in the pond.

5.CONCLUSION

Among the most abundant taxa inhabiting the sediment of SMP facilities, the oligochaete communities could serve as valuable biological endpoints to be integrated into the SQT and serve for setting up the sediment ecological quality goals by implementing the existing oligochaete indices, such as the ones tested in this study. Our results showed that two major contaminants were responsible for detrimental effects on the ecological quality of the sediments in the ponds studied: (i) heavy metals in the sediment and (ii) chloride concentrations in the interstitial water in the pond sediment. According to our results, the IOBL value of 16, combined with a %Ssp of 15, could serve as a realistic objective of good ecological quality for the sediment of the SMPs studied. More research is necessary to validate and generalize these quality objectives to other SMPs.

DATA AVAILABILITY STATEMENT

All relevant data are included in the paper or its Supplementary Information.

CONFLICT OF INTEREST

The authors declare there is no conflict.

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