

Biomonitoring of wet-weather flow impacts on fine sediments in urban waters: Canadian and French experience

Bioindication de l'effet des rejets de temps de pluie sur les sédiments fins des hydrosystèmes urbains: expériences canadiennes et françaises

Lafont M.*, Grapentine L.**, Rochfort Q.**, Marsalek J.**, Breil P.*

**Cemagref*, 3bis quai Chauveau, C.P. 220, F-69336 Lyon
(lafont@lyon.cemagref.fr)

**Environment Canada, 867 Lakeshore Rd., Burlington, Ontario, L7R 4A6,
Canada (Lee.Grapentine@ec.gc.ca)

RÉSUMÉ

Les invertébrés benthiques sont couramment utilisés pour évaluer l'effet des rejets polluants sur les milieux aquatiques. L'expérience du *Cemagref* en France a montré que les indices normalisés, comme par exemple l'indice oligochètes IOBS, pouvaient être utilisés pour apprécier l'état écologique des cours d'eau urbains et répondre aux exigences de la DCE. Les études canadiennes sur les cours d'eau et les bassins d'orage, utilisant des analyses chimiques, des tests de toxicité benthiques et la structure des communautés d'invertébrés, ont montré que la toxicité apparaissait de préférence dans les sédiments fins accumulés dans les plans d'eau urbains les plus profonds. Une évaluation plus précise de l'effet des rejets de temps de pluie pourrait être obtenue en combinant les approches canadiennes et françaises.

ABSTRACT

Benthic invertebrate assessments can be used to gauge the impact of urban wet-weather flows in receiving waters. Experiences from *Cemagref* in France have shown that standardized benthic indices (e.g. Oligochaete Index of Sediment Bioindication - IOBS) can be used to reliably determine the ecological status of urban streams and can be incorporated into the new European Water Framework Directive. The Canadian studies on streams and stormwater ponds using chemical analyses, benthic toxicity testing and benthic community structure comparisons have shown that toxicity was more likely to occur in deeper ponds where fine sediments from urban runoff accumulated. A more comprehensive evaluation of wet-weather flow impacts could be obtained by combining approaches from both countries.

KEYWORDS

Benthic communities, Benthic toxicity assays, Impacts on receiving waters, Sediment chemistry, Urban wet-weather flows.

1 INTRODUCTION

The assessment of impacts of urban wet-weather flows (i.e. stormwater discharges and combined sewer overflows) on habitat quality in receiving waters is conducted frequently in connection with restoration of degraded waters (Walsh et al. 2005), for ensuring compliance with water quality regulations (Nabelkova et al. 2004), or when planning new developments (Walsh 2004). The wet-weather flows represent intermittent sources of pollution associated with rainfall and snowmelt events. The pollutants conveyed by wet-weather flows may accumulate in fine sediments in receiving waters (Rochfort et al. 2000; Pitt 2003; Grapentine et al. 2004) and, consequently, the biomonitoring of these habitats is important for the preservation and rehabilitation of urban waters. Many assessment approaches have focused on fine sediment quality recognizing that it reflects water quality in the water bodies where this habitat predominates, and that good sediment quality is of paramount importance for maintaining the abundance, diversity, and composition of benthic macroinvertebrates, which represent the main source of energy for higher trophic levels (fishes). During the last 20 years, various methodologies for assessing sediment quality have been developed and applied to a variety of urban waters, including streams, rivers, man-made impoundments (e.g. stormwater ponds or detention basins), lakes, and estuaries. Early studies targeted heavy metals, PCBs and PAHs burdens in sediments, which were often related to extreme cases of pollution. In later studies, preference was given to biological indicators over rather complex (and often expensive) chemical analyses needed to determine bioavailability of pollutants. Consequently, assessments of benthic communities have been used to make inferences about water and sediment quality in urban waters in many jurisdictions. Experience with applications of such methodologies by two research teams in France and Canada (*Cemagref* in Lyon and the National Water Research Institute in Burlington, Ontario, respectively) are compared and analyzed in this paper, with the overall objective of achieving further methodological refinements.

2 FRENCH EXPERIENCE FROM CEMAGREF

The *Cemagref* developed several biomonitoring methodologies. The biological quality of fine sediments in watercourses is assessed by the standardized IOBS index (Oligochaete Index of Sediment Bioindication, AFNOR 2002; Lafont et al. 2003). The index $IOBS = 10 S T^{-1}$, where S = number of species in a sediment sample and T = percentage of the predominant group of Tubificidae in the same sample, either Tubificidae with hair setae (TUCP) or without hair setae (TUSP). When TUCP and TUSP both equal zero, $IOBS = S$, which is a situation found only in mountainous areas. The IOBS value varies from 0 to more than 6 (Table 1), with high values equal to 27 or more in reference sediments (Lafont et al. 2003). The classification colours listed in Table 1 are recommended by the European Water Framework Directive WFD (UE 2000). A nomograph is used for interpretation of the sediment quality and prediction of pollution effects (Figure 1). It was tested in pollution gradients at more than 200 sites in all French river basins (Prygiel et al. 1999; Lafont et al. 2003), including urbanized areas. It may be used as a stand-alone tool when fine sediment habitats cover more than 60% of the bottom at a given site. If fine sediments do not predominate, it is better to use a harmonization system grouping the IOBS index and other biotic indices (invertebrates, diatoms and fish, Lafont et al. 2001). The examples of experimental sites given in Figure 1 include both urban and rural (natural) sites. Thus, it is possible to link the presence of specific benthic communities (as expressed by the benthic indices) to potential pollutant sources with an acceptable degree of precision. Trace metals and organic toxicants like PCBs are generally related to the predominance of Tubificidae without hair setae (TUSP, *Limnodrilus* spp.),

whilst PAHs, and strong organic pollution (Prygiel et al. 1999) and copper (Lafont et al. 2003) are related to that of Tubificidae with hair setae (TUSCP).

IOBS	IOBS ≥ 6	6 > IOBS ≥ 3	3 > IOBS ≥ 2	2 > IOBS ≥ 1	IOBS < 1
Status	Very good	Good	Moderate	Poor	Bad
Colours	Blue	Green	Yellow	Orange	Red

Table 1. Ecological status of sediments according to the Oligochaete Index of sediment Bioindication (IOBS); modified after Lafont et al. (2003); ecological status and associated colours are requirements of the Water Framework Directive (WFD).

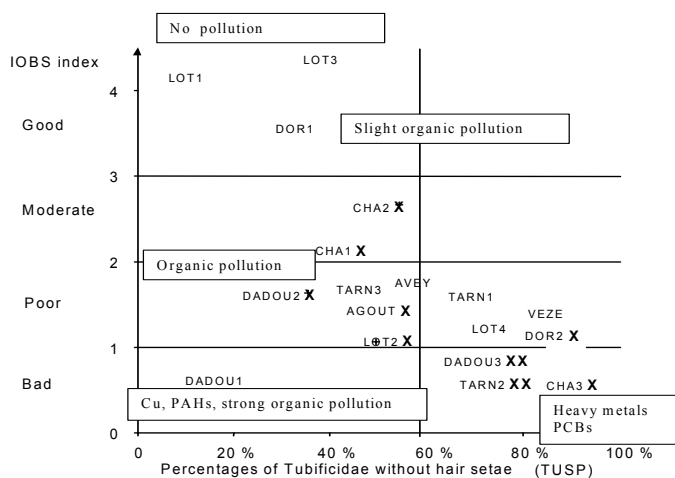


Figure 1. Nomograph for IOBS interpretation; after Lafont et al. (2003); **X**: periurban sites; **XX**: urban or densely populated periurban sites.

In stagnant ecosystems, the percentages of Tubificidae + *Dero digitata* (Ruperd et al. 1986) was selected for the study of shallow urban ponds (Table 2). This index is known as the IOPP (Oligochaete index for shallow lake bioindication) and is calculated as $IOPP = 1 + [100 - \% (\text{Tubificidae} + D. \text{digitata})]$. The Oligochaete Index of Lake Bioindication (IOBL) ($IOBL = \text{number of species} + 3 \log_{10} [\text{oligochaete numbers} + 1 \text{ per } 0.1 \text{ m}^2]$, AFNOR, 2005) can also be used in shallow urban lakes, but it is better suited for natural lakes with a depth greater than 5 m. The IOBS index (Table 1) can also be tested in ponds. The three indices (IOBL, IOPP, IOBS) give relatively corroborating diagnostics (Table 3). The environmentally protected ponds (Ulis IV and GdL.3) exhibited the highest values of indices. The Tubificidae without hair setae (TUSP) dominated in Ulis ponds, which is consistent with the contamination by heavy metals (SINC: sum of contamination indices of 6 heavy metals, Zn, Cu, Pb, Cr, Cd and Hg; Lafont 1989), with an apparent toxicity threshold of about $SINC = 15-16$. In the Grand Large (Lafont, unpublished data), the central deep pool (GdL.1) was the site most affected by pollution, whereas relatively good quality ($IOPP = 83.2$) observed at the pond outlet (GdL.3) could be explained by remoteness from urban pollution sources, and additional influxes of groundwater.

SESSION 7.1

Status	Very good	Good	Moderate	Poor	Bad
Colours (WFD)	Blue	Green	Yellow	Orange	Red
IOPP*	> 50	50-40	39-21	20-10	<10
IOBL**	> 25	25-20	19-15	14-10	<10

Table 2. Presentation of two oligochaete indices for fine sediments: IOPP (Oligochaete index for shallow lake bioindication) *, modified from Ruperd et al. 1986), and IOBL: Oligochaete Index of Lake Bioindication; **, which is not suited for shallow lakes and ponds; WFD: Water Framework Directive.

	Ulis IV*	Ulis III*	Ulis II*	Ulis I*	Ulis S*	GdL.3	GdL.2	GdL.1
IOBL	16.0	12.2	13.9	11.2	7.2	18.4	15.5	11.1
IOPP	44.1	12.2	14.1	1.4	1.0	83.2	31.7	1.0
IOBS	6.3	1.3	1.7	1.9	0.2	7.1	1.6	0.5
TUSP	13.8	37.0	34.6	65.8	96.1	12.6	51.0	91.3
SINC	-	7.9	14.9	17.2	28.9	-	-	-

Table 3. Use of three oligochaete indices in 6 urban ponds, less than 5 m deep; SINC: sum of contamination indices of 6 heavy metals (Zn, Cu, Pb, Cr, Cd, Hg); GdL.1 to 3: Grand Large pond; TUSP: % of Tubificidae without hair setae; - : no data.

But, if the toxicity of trace metals depends on their sediment concentrations, it depends also on their bioavailability which results from very complex processes. The relationships between concentrations and toxicity are so very difficult to evidence in the field (Prygiel et al. 1999).

3 CANADIAN EXPERIENCE

Research studies conducted by the National Water Research Institute (NWRI) focused on benthic sediments deposited along urban streams beds and in stormwater facilities in Ontario, Canada. Three issues were addressed: (i) sediment chemistry, (ii) sediment toxicity, and (iii) the benthic community structure. Incorporating both the biological and chemical analyses into these investigations provided a more complete picture of the impacts of wet-weather discharges on the receiving environments. This methodology is known as the sediment triad approach (Chapman et al. 1992). Past studies of sediment chemistry have focused on ubiquitous constituents in the urban environment, for which some sediment quality guidelines exist in Canada, such as selected heavy metals, polycyclic aromatic hydrocarbons (PAHs), and some other constituents (e.g. PCBs) (MOEE 1997; CCME, 2002). Where sediments exceed specific chemical guidelines, biotesting is generally triggered, since the chemicals may not be in a bioavailable form and depending on overlying water quality, the site may still be capable of supporting a rich and diverse benthic community. Biotesting still provides the most suitable measure of ecosystem health and impact assessment, since even where guidelines are not exceeded for individual constituents, an additive effect of multiple pollutant stressors can occur, rendering the sediment unsuitable for sensitive benthic communities. Where chemistry data show signs of sediment contamination, biological forms of testing, including bioassays using benthic invertebrates, are used to determine bioavailability. Since toxicity can also affect different species in different ways and certain species prefer sediment with different grain sizes (e.g. sand vs. silt), tests using only a single species can be misleading. Benthic sediment toxicity is tested by means of four bioassays, each with two

endpoints: *Hyalella azteca* 28-day survival and growth test, *Chironomus riparius* 10-day survival and growth test, *Hexagenia* spp. 21-day survival and growth test, and *Tubifex tubifex* 28-day survival and reproduction test. Sediment toxicity is determined by comparing endpoint means with criteria derived from tests with uncontaminated reference sediment from the Great Lakes and indicates three toxicity categories: (i) non-toxic, (ii) potentially toxic and (iii) toxic. Benthic community structure analyses are also performed to provide additional information on the overall quality of habitat. This information is obtained by performing counts of each species present at a site. Factors including taxon richness and abundance and total number of organisms can be compared to representative healthy sites to determine the tendency towards a healthy or poor community structure. A reduction in the presence of pollutant intolerant species, reduction of diversity and an increase in the presence of pollution tolerant species suggests a poor quality habitat.

In a survey of sediment accumulated in six different types of stormwater management facilities, results indicated that the sediment quality depended strongly on pollutant sources, rather than the type of facility. Other influential factors included particle sizes and organic contents. Benthic communities were assessed with respect to organism abundance, diversity (at the family level), and occurrences of pollution tolerant or sensitive species and related to the presence of chemical contaminants using principal component analyses. Results from surveys and field experiments involving over 30 sites exposed to various intensities of urban wet-weather flows indicate that while sediments in sites exposed to discharges were generally enriched with nutrients, metals and PAHs (exceeding sediment quality guidelines in some cases), biological degradation was not common (Grapentine et al. 2004). In streams, acute and chronic toxicity of sediment collected from depositional areas was absent or low. Benthic invertebrate communities below outfalls showed minor alterations in taxonomic composition relative to communities upstream of outfalls, but were not significantly depauperated. Overall, wet-weather flows appeared to impact sediment physico-chemical conditions of streams, but not to levels harmful to the majority of benthic invertebrates.

Benthic conditions in ponds receiving wet-weather flows appear more disturbed than those in streams. In a treatment facility for runoff from a major highway and residential areas, concentrations of metals, PAHs and chlorides (from winter road maintenance) were substantially higher than those in urban streams, particularly in deeper parts of the ponds. Benthic invertebrate community structure was related to depth; total abundance, taxon richness, and abundance of the two dominant taxa (oligochaetes and chironomids) were lowest in the deepest sites (Figure 2). Although sediments from most sites in the treatment ponds were not acutely toxic in laboratory tests, reduced survival and growth were observed in sediments collected from inlet areas of the ponds. Overall toxicity (based on multiple endpoints) was associated with elevated levels of metals and heavy PAHs (Figure 3). Benthic impairment in these stormwater ponds likely resulted from a number of factors, including poor physical habitat (predominance of fine sediment and high embeddedness), poor sediment quality (indicated by severe levels of contamination), and poor water quality (most importantly in terms of high salinity). Repeated assessments in several seasons and years indicate that such adverse conditions persist through time, at least in the deep areas.

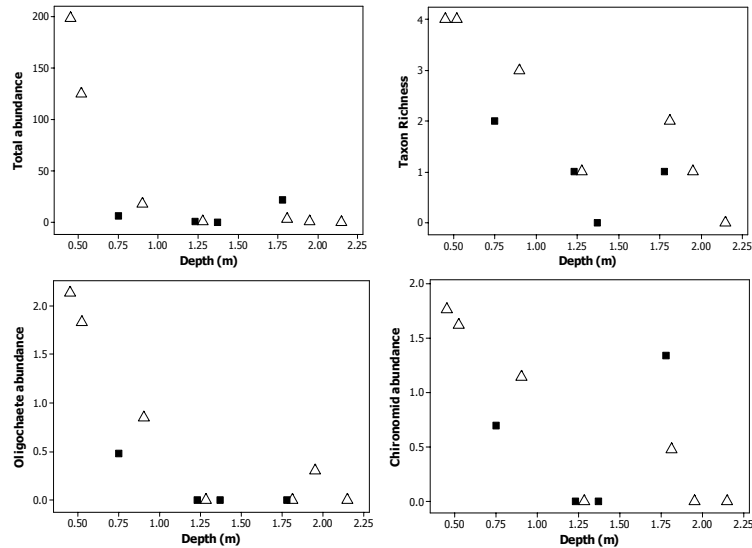


Figure 2. Effect of depth on sediment benthic invertebrate communities in an urban stormwater treatment facility. Data are medians for 5 replicate cores (33.2-cm² area x 10-cm depth) from 2 connected ponds (Terraview, open triangle; Willowfield, solid square). Abundances of oligochaetes and chironomids, the two dominant taxa, are on log₁₀ scale. Deeper, more central locations in the ponds have fewer benthic invertebrates and fewer taxa than shallow, shoreline.

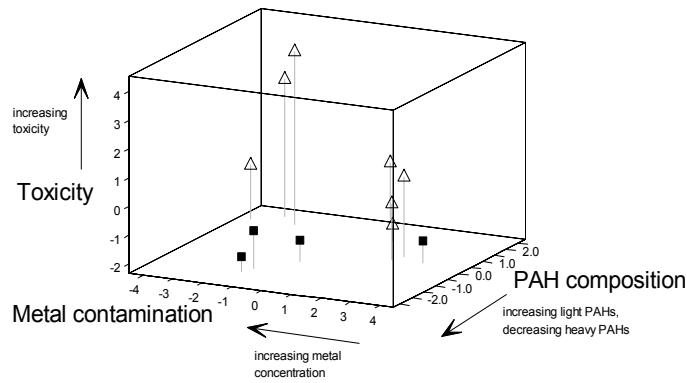


Figure 3. Relationships between sediment toxicity and sediment metals and PAHs in an urban stormwater treatment facility. Axes describing toxicity, metal and PAH conditions are based on principal components analyses of multiple endpoints or contaminant concentrations for sediment from 2 connected ponds (Terraview, open triangle; Willowfield, solid square). Correlation analyses suggest a positive association of toxicity with metals and high molecular weight-PAH compounds.

4 DISCUSSION

The priority given to the biomonitoring of fine sediments is justified because they represent a record of water quality conditions in the water body studied, serve as an important part of the aquatic habitat inhabited by benthic communities, and at the

same time, also store pollutants, which could be, over the long-term, released into the water column when the ambient chemistry changes. When pollutants accumulate in sediments, they do not inevitably affect the biodiversity of the aquatic habitats over the short-term, which can lead to a fairly optimistic but potentially misleading view of the degree of impairment. Some field studies (Grapentine et al. 2004) indicate that the sediment toxicity is not always strongly related to contaminant concentrations from urban wet-weather discharges. Detection of the effects can be indeed limited or obscured by such inherent conditions as natural heterogeneity in the distribution of benthic invertebrates, episodic exposure, and background levels of disturbance found in urban aquatic systems. In addition, each site is unique, since each form of a given pollutant does not have the same bioavailability and toxicity. But the likelihood of occurrence of bioavailable chemical species increases in those parts of aquatic habitats which serve for deposition of fine sediments and represent sinks for pollutants (Prygiel et al. 1999). Moreover, fine sediments can frequently accumulate in urban landscapes and in their man-made features (e.g. constructed wetlands, ponds), or greatly modified natural water bodies (ponds, lakes, streams, rivers). The study of urban stormwater ponds is becoming quite important in urban areas because such stormwater ponds are being widely used in new developments by environmental planners striving to provide recreational, aesthetic and habitat amenities in urban developments. However, it is important to ensure that the habitat created by stormwater ponds and wetlands is not rapidly degraded by wet-weather flow discharges, leading to creation of contaminated habitats over time (Bishop et al. 2000). The construction and placement of new stormwater facilities (ponds or wetlands) often focuses on lands adjacent to open water courses, because such lands are generally open and unsuitable for residential or commercial development. It is also feasible to restore habitat conditions in existing ponds or wetlands, like old gravel-ponds (cases of Grand Large and Ullis I to IV) or storm water facilities (case of Ullis Sud) by implementing source controls and treatment trains providing improved sediment and water quality. Finally, this brief comparative study of research approaches practiced by the *Cemagref* and NWRI teams opens new opportunities for sharing and advancing the knowledge in this important field.

5 CONCLUSIONS

The comparison of methodologies used by the two research teams offers new directions for future studies. There is a great deal of experience obtained from applications of the French index-based methods in watercourses (IOBS and the harmonization system) and natural lakes (the IOBL index), however, so far, the EOPP index has only been tested in six urban ponds. In the near term, the testing of the comparative relevance of various methodologies, including the index-based methods and sediment quality triads, on fine sediments of urban ponds and streams seems to be the first step of the future collaborative research. In particular, the experiences from Canadian studies on pond biomonitoring and routine sediment toxicity tests, combined with the French index-based methods, offer a promising approach to advancing knowledge in this field. The second phase of such a collaboration should address self-purification processes in fine sediment habitats (Datry et al. 2003, Nogaro et al. 2006), including investigations of fine sediments as pollution abatement systems, and such associated issues, as the assessment of self-purification capacities and their relationships with biodiversity. Once more data and experience has been obtained, other fields of study might be promoted, including the ecology of porous habitats (surficial coarse sediments and hyporheic systems; Lafont et al. 2006), which also act as sinks for pollutants, in a similar way as fine sediments.

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