ECOHYDROLOGY & HYDROBIOLOGY

DOI: 10.2478/v10104-012-0004-4

Vol. 12
No. 1, 9-20
2012

From research to operational biomonitoring of freshwaters: a suggested conceptual framework and practical solutions

Michel Lafont[1], Guillaume Tixier[2], Jiri Marsalek[2], Céline Jézéquel[1], Pascal Breil[3], Laurent Schmitt[4]

[1] Irstea UR MALY, 3 bis quai Chauveau – CP 220, F-69336 Lyon, France,
e-mails: miche.lafont@irstea.fr; clinejez@hotmail.fr
[2] NWRI (National Water Research Institute), Environment Canada, 867 Lakeshore Road,
P.O. Box 5050, Burlington, ON L7R 4A6,
e-mails: gtvoi@hotmail.com; Jiri.Marsalek@ec.gc.ca
[3] Irstea UR HHLY, 3 bis quai Chauveau – CP 220, F-69336 Lyon, France,
e-mails: pascal.breil@irstea.fr, pascal.breil@cemagref.fr
[4] University of Strasbourg, ERL 7230, 3 rue de l'Argonne, F-67083 Strasbourg cedex
(France), e-mail: laurent.schmitt@unistra.fr

Abstract

To contribute to solving the dilemma of the conflicting needs of water managers and ecologists, we are proposing an approach for the use and development of existing biomonitoring tools. For running waters, a harmonization system allows to use a set of various biotic indices. Functional traits are regarded as a basis for assessing ecological functioning. They take into account the dynamics of water exchanges between surface water and groundwater. In lakes, the oligochaete index IOBL describes the metabolic potential of deep-water sediments. Coupled with percent pollution-sensitive oligochaete species, it allows to define a typology of lake sediments. The presented tools are integrated through a conceptual framework, including research management and technology transfer procedures.

Key words: running and stagnant waters, ecohydrology, biological methodologies.

1. Introduction

The concept of water quality assessment based on the examination of "indicator organisms" is more than a century and a half old, if one refers to the contributions of Kolenati (1848), Cohn (1853) (authors cited by De Pauw, Vanhooren 1983; Blandin 1986), Kolkwitz and Marsson (1902; 1908; 1909, cited by Hynes 1960; Sládečeck 1973). These ancient authors highlighted that organisms living in organically polluted waters were different from those living in clean waters. These works were forerunners because they were first attempts to express by numerical values the effects of fouled waters on aquatic life. We can consider that ecological studies intended for the protection of aquatic life were born from those contributions. The brilliant book of Hynes (1960) on the biology of polluted waters was a major and presently still valid synthesis, and the first "biotic indices", based on benthic macroinvertebrate assemblages, were presented by Woodivis (1964) in Great Britain and by Verneaux and Tufféry (1967) in France (see also Bouleau *et al.* 2009). These qualitative methodologies were relevant and well-suited for applications by end-users. They have served us well during the period when politicians, stakeholders and managers have become aware of the need to protect aquatic ecosystems against anthropogenic alterations. They also enabled quick and lasting advancement of research (R), research and development (R&D), and development (D) projects in freshwater ecology. However, in the growing concern of the need to protect or restore freshwater ecosystems, there are increasing conflicting demands of water management and water science (Cullen 1990). If water managers would rather privilege quick universal and operational procedures, the ecologists' need for understanding ecosystem processes and responses to different kind of perturbations requires more time consuming and specific in-depth research. For instance, it is commonly admitted that integrated approaches including multiple lines of assessment emphasizing *in situ* biological indicators rather than single universal approaches are necessary to accurately address ecological integrity damage, protection and restoration issues (Barbour *et al.* 2000). From a management point of view, difficulties in finding appropriate and comparable methods when confronted to a multitude of biological tools can be discouraging and could lead to favoring over-reductionist approaches (Chapman 2007). Moreover, the need to meet nowadays large-scale water policy requirements e.g., European Water Framework Directive, (EU 2000) challenges our ability to federate all the methodologies with minimizing the risk of loosing knowledge or valuable experience that have been gained in decades of water quality assessment science (Gabriels *et al.* 2010) We are therefore suggesting that the conflicting demands of managers and ecologists can be reconciled in several ways, including the general approach proposed here, which are currently being developed on the basis of a multidisciplinary research and follow a general objective of sharing research findings with others.

2. Material and methods

The approaches applicable to running waters were presented in several papers listed in Table I. The conceptual pattern is given by the EASY concept (Ecological Ambience System), which is based on the idea that biocenoses (BIO) are not only related to the inputs of organic and mineral substances (IN), but also to the way such substances are stored and processed by the ecosystem. Thus, storage, assimilation and self-purification processes ("ecosystem defenses", ED) are likely to vary among different functional units (FUs) of the ecosystem. The structure of the conceptual model EASY illustrates the complexity of the physical, chemical and biological interactions in the receiving aquatic systems. This concept illustrates the interactions between mass flux (water and chemical substances), forms (geomorphology, physical structure of aquatic habitats in the riverscape) and biocenoses (all living organisms, from bacteria to fish). The LOUE (Lowest Observed Urban Effect, Table I) is an adaptation of the EASY concept to urban aquatic habitats, and the "4Ws strategy" is a management strategy for ecological studies of such habitats. Consequently, we have adapted the existing qualitative biological methodologies (a qualitative approach) and created new ones (an ecohydrological approach) with the overall goal of contributing to solving the puzzle of complex interactions linked to the healthy and sustainable functioning of the river system.

For the selected approach to lacustrine waters, the material and methods were presented in several previous papers: AFNOR (2005); Juget *et al.* (1995); Lafont (1989; 2007); Lafont, Juget (1985); Lafont *et al.* (1991; 2007); Tixier *et al.* (2011a; 2011b).

3. Results

3.1. Approaches applicable to running waters

The qualitative approach

It is based on an examination of operational qualitative biotic indices (harmonization system – HS, Table I). "Operational" means that the indices are standardized, or at least documented by reproducible field and laboratory protocols, have well-defined ecological meaning and context, and have been transferred to end-users for routine use. The HS was initially built with French indices (Lafont *et al.* 2010). The first component is named "the general ecological quality" and was meant to represent a general ability of the system to support biodiversity. Its related index includes all benthic invertebrate assemblages (the IBGN index, AFNOR 2004a or its successor for WFD compliance). The second component is "the biological quality of water" (the related index: diatom index IBD, AFNOR 2007). The third component is "the quality of fine sediments", recognizing that such sediments can store great quantities of pollutants (the related index: the oligochaete index IOBS, AFNOR 2002). The last component represents "the fish assemblages", recognizing that fish ecology is very different from that of other animals or vegetal biocenoses (the related index: the fish index IPR, AFNOR 2004b). It is expected that a general qualitative understanding of the river functioning can be gained by considering

Table I. Concepts and tools developed for running waters.

Concepts	Tools
EASY (Ecological Ambience System) conceptual model (Lafont 2001) is intended for multidisciplinary studies; it currently constitutes a conceptual basis for ecohydrological research.	BIO = f(IN) - g(ED) BIO: biodiversity, including taxon richness and processes; IN: inputs; ED: ecosystem defenses.
LOUE (Lowest Observed Urbanization Effects) concept (Lafont *et al.* 2008), integrated in resilience and resistance domains, is intended to define the limiting condition of the ecosystem that must not be exceeded in urban aquatic habitats (i.e., the point of no return).	Comprises a curve illustrating resilience, resistance, and LOUE domains; suggestions of associated biomonitoring tools for defining these domains and metrics for assessing the LOUE boundary (HS, FTrs; see following sections of this Table).
Harmonization system (HS) (Lafont *et al.* 2010) is intended for the assessment of a global ecological status (or quality) at a given site and the targets (= compartments) that have to be rehabilitated, restored or preserved, even if the global ecological status is good; the true or suspected causes of physical and chemical alterations can be documented.	Integrates four components and their associated French standardized qualitative indices, harmonized by the five classes of ecological status (high, good, moderate, poor, bad) (EU 2000); a mean value is calculated but specific information given by each index is kept; a weighing procedure can be applied according to percent coverage of the river-bed by fine sediment (also referred to as the embeddedness); ecological damage assessment is defined by the loss of ecological status classes compared to the selected objective (high or good ecological status).
Ecohydrological approach (EA) (Vivier 2006; Lafont *et al.* 2006; 2010); intended for the assessment of the role of interactions between chemical and physical factors (mainly the dynamics of water exchanges between surface waters and groundwater) and their influence on biodiversity.	Functional traits (FTrs) and ecological potential (EP) calculation; FTrs are defined by oligochaete species assemblages from porous habitats (coarse surficial sediments, hyporheic system); FTrs reflect various physical and chemical factors that interact in the functioning of a stream; EP = [(FTr1+FTr2)+1]/[(FTr3+FTr4)+1], i.e. the ratio of the FTrs characteristic for a preserved state of functioning to those of the most impaired ones.
The “4 Ws strategy”: considerations for defining protection and rehabilitation strategies in urban aquatic systems (Breil *et al.* 2008).	Range of rehabilitation targets for urbanized streams and the associated metrics; guidance on “why”, “what”, “where” and “when” to monitor indicators for assessing aquatic ecosystems.
Riverscape typology (Lenar-Matyas *et al.* 2009; Poulard *et al.* 2010); intended for finding engineering solutions providing both flood protection (dry emergency reservoirs, river training) and biodiversity preservation.	Four riverscape types according to a gradient of increasing habitat richness, from fully artificial (man-made) beds to the natural ones; proposals of associated biomonitoring tools depending on the habitat richness (HS and EA, see above)

these four important components and by keeping the information given by individual indices intact, even if they produce conflicting information (Lafont *et al.* 2010).

The functional approach (functional traits, ecohydrological approach).

The functional traits (FTrs) were defined from oligochaete assemblages inhabiting porous habitats (i.e., surficial coarse sediments and the hyporheic system; Table I). Each functional trait (FTr1, FTr2, FTr3, FTr4, FTri) is defined as percent oligochaete species indicative of specific environmental conditions and is calculated as the number of organisms belonging to the indicator species group expressed as the percentage of the total number of oligochaete organisms in the same sample. FTr1, “permeability”, was obtained by measuring percent oligochaete species indicative of active flow exchanges between surface water and groundwater. FTr2 is defined as percent pollution-sensitive oligochaete species, which are associated with good quality water. The FTr3 trait is defined by percent water pollution-tolerant oligochaete species. The FTr4 trait (“sludge effect”) indicates the presence of polluted sludge (sediment) within the interstices of porous habitats and is associated with heavy pollution by urban and industrial discharges. The FTri trait is defined by the percentage of species which characterize moderately impacted conditions. The “Ecological Potential” EP is the ratio of FTrs characteristics of the most preserved habitats (FTr1 + FTr2) to those of the most impaired ones (FTr3+ FTr4). The FTrs and EP calculations are tools derived from the EASY concept and are intended for the assessment of interactions between chemical and physical factors. These interactions mainly address the dynamics of water exchanges between surface waters and groundwater, which is a major factor for understanding stream functioning (Jones, Mulholland 2000).

The riverscape typology (ecohydrological approach)

It is intended to address technical solutions providing both flood protection and biodiversity preservation (Table I), and apply hydrology to issues concerning, e.g., building of dry flood protection (emergency) reservoirs or river training (Lenar-Matyas *et al.* 2009; Poulard *et al.* 2010). It illustrates the known fact that biodiversity (including the related processes) is positively related to the habitat richness in the riverscape of the minor bed. The more the habitat richness is preserved or restored, the more the biodiversity will increase. However, the habitat richness in the riverscape has to follow the general geomorphologic pattern of the investigated sites, rather than trying to create habitats where none have previously existed. The riverscape typology was derived from an earlier approach developed for urban streams (Breil *et al.* 2008, Table I).

3.2. A selected approach to lacustrine waters

We have formerly viewed the well-established lake eutrophication classification (oligotrophic, mesotrophic, or eutrophic lakes) as masking the fact that identically classified lakes strongly differed in other aspects (Lafont 1989; Lafont, Juget 1985; Lafont *et al.* 1991): a eutrophic small lake or a eutrophic great Alpine lake were considered sustaining the same trophic level. Instead of taking a new look at the eutrophication concept, which was most relevant when considering phytoplankton and phosphorus contents, or building new traditional qualitative indices, we have developed a new approach based on the study of oligochaete assemblages inhabiting sublittoral and deep-water lacustrine sediments (Lafont 1989; Lafont, Juget 1985; Lafont *et al.* 1991). It was based on the study of different French lakes, including the French sections of Lake Léman, crystalline lakes in the Vosges mountains and calcareous lakes in the Jura and Alp mountains. Firstly, chemical inputs brought in with water interact with the physical properties of the lake, like the water mass and physico-chemical characteristics of sediments, and particularly their calcium carbonate contents (Lafont 1989; Lafont, Juget 1985; Lafont *et al.* 1991). $CaCO_3$ is considered as an active agent of mineralization of organic matter in sediments; sediments poor in $CaCO_3$ are generally rich in organic matter. The mineralizing role of calcium was formerly observed in running waters by Egglishaw (1968, cited by Macan 1974). Mineralization processes are among the key ecological functions of sediments in standing waters. Such transformation and recycling of organic matter is at the very base of the food web, providing resources for primary producers and enabling further transfer of energy and matter through the ecosystem. Among other sediment-dwelling invertebrates, oligochaetes play a great role in the mineralization of organic matter in sediment. We found an inverse relationship between oligochaete species richness and densities, and organic contents in sediments; the greatest oligochaete species richness and densities were observed in sediments with the lowest organic (C, N and P) and the highest $CaCO_3$ contents (Lafont 1989; Lafont *et al.* 1991). This relationship was evident when comparing calcareous lakes with crystalline lakes, and when comparing lakes from strictly calcareous areas (Lafont 1989). The oligochaete methodology was therefore based on species richness and abundance, and served as a surrogate measure of the mineralization potential of lake sediments. A multiple regression analysis yielded best results for the following formula:

$$IOBL = N_{SP} + 3 \log_{10} (N+1)$$

where IOBL is the Oligochaete Index of Lake Bioindication; N_{SP} is the number of oligochaete species in a sediment sample (also referred to as ‘species richness’), and N is the abundance of oligochaetes per 0.1 m^2 in the same sample. Numerical values of IOBL depend on the mesh size of the screen used in processing sediment samples. Such a size should be chosen according to the objectives of the survey; smaller mesh sizes retain samples with more small-size oligochaetes and contribute to both higher measured abundance of such species (see Table II) and higher taxon richness, but also increase the level of effort required to analyze such samples. Originally, the IOBL index was derived from the analysis of sediment samples retained by a 0.16 mm mesh size sieve and was referred to as the “Etat écologique Oligochètes LAcustres” (EOLA index, Lafont 1989). The method was simplified for practical applications (Juget *et al.* 1995) and further standardized as “Indice Oligochètes de Bioindication Lacustre” (IOBL) by considering only sediment samples retained by a 0.5 mm mesh size (AFNOR 2005). The relationship between the IOBL index calculated from sediment samples retained by the 0.16 mm mesh size, or EOLA, and the IOBL index standardized with the use of 0.5 mm mesh size was described by the formula proposed by Lafont (1989): EOLA = 1.3 × IOBL – 0.6 ($r^2 = 0.92$; $F = 349.8$; $n - 2 = 35$). In the spirit of retaining flexibility in refining the biomonitoring methods, Tixier *et al.* (2011a) further experimented with sample preparation and used yet another mesh size, 0.25 mm, which provides a compromise between losing some information on N_{SP} and N, when using relatively coarse screens (0.5 mm), and increasing

Table II. Indicator oligochaete species in deep-water lake sediments (modified after Lafont 1989; 2007; AFNOR 2005).

Group 1. "Sensitive" species in deep-water lake sediments (= pollution-intolerant species)
*Amphichaeta leydigii**, *Chaetogaster* spp.*., *Nais* spp., *Ophidonais serpentina, Piguetiella blanci, Slavina appendiculata, Specaria josinae, Stylaria lacustris, Uncinais uncinata, Vejdovskyella intermedia, V. comata, Bichaeta sanguinea*⁺, *Stylodrilus* spp.⁺, *Dorydrilus michaelseni*⁺, *Spirosperma velutinus*⁺, *Rhyacodrilus falciformis*⁺, *Psammoryctides barbatus, Marionina argentea**⁺, *Cernosvitoviella* spp.*⁺
* these small-size species are generally not found in sediments retained in the laboratory by the 0.5 mm mesh-size sieve, except for *Chaetogaster diaphanus*;
⁺ AED species indicating active water exchanges between surface waters and groundwater (Lafont, Vivier 2006).
Group 2. Species, which if found alone or significantly predominating, characterize a natural dystrophy (due to peat, coarse vegetal detritus, abundance of Characea)
Tubifex tubifex, Haemonais waldvogeli, Aulodrilus pluriseta, Vejdovskyella comata
Group 3. Species indicating high pollution effects, particularly when they are found alone in deep sediments
Potamothrix heuscheri, P. hammoniensis, Limnodrilus hoffmeisteri, L. claparedeanus, L. udekemianus, Lumbriculus variegatus, Bothrioneurum vejdovskyanum, Dero digitata
Group 4. AED species not yet found in deep-water sediments of French natural lakes or reservoirs, but found in Canadian urban stormwater pond facilities (Tixier *et al.* 2011a)
Pristina aequiseta foreli, P. aequiseta aequiseta, P. menoni, P. longiseta

substantially the size of samples retained by the fine screens (0.16 mm). To avoid any ambiguity, the IOBL index will be further mentioned herein with the mesh size as a subscript ($IOBL_{0.5}$ or $IOBL_{0.16}$) when referring to values derived from particular (historical) surveys.

The IOBL index varies from 0 to > 20 and is regarded as an indicator of the "metabolic potential of sediments", i.e., the capacity of sediments for mineralizing organic matter and favouring benthic oligochaete assemblages in deep-water sediments. This potential was naturally higher in carbonate rich sediments of great Alpine lakes ($IOBL_{0.16} > 15$, for example in Lake Léman, Fig. 1) where oligochaetes are abundant and diverse. Conversely, particular environmental conditions can influence the natural mineralization potential of lakes. For instance, a low-volume system with abundant coarse vegetal detritus (twigs, peat) or characeal algae, which are naturally hard to mineralize, would be unsuitable for benthic colonization. Consequently, lowest values ($IOBL_{0.16} < 5$) were found in small peaty crystalline or calcareous lakes. Since the organic matter accumulates by naturally-reduced mineralization, this condition was considered as a characteristic of the "dystrophic" state. Note that the null potential ($IOBL_{0.16} = 0$) may also occur in natural lakes, for example in high-altitude mountain lakes, or in excessively peaty or polluted sediments, where no benthic organisms are found (Lafont 1989; Bazzanti, Lafont 1985; Collado, Schmelz 2001).

Consequently, the IOBL index is not an indication of the quality status of lakes with respect to the potential effects of pollution, but merely one of the measures of the lake ecosystem functioning, which however can be impaired by the pollution. Since it can be difficult to separate naturally low metabolic potentials (dystrophic lakes) and pollution effects, the concept of a "sensitive species" factor was introduced to complement the diagnosis (Table II). The list of sensitive species (also refereed to pollution-intolerant) is not fully transferable to running waters, because of great ecological differences between the running water and lake ecosystems, except for the species indicating water exchanges between surface waters and groundwater, which are the same for running and standing waters. The IOBL index, combined with percent sensitive species (determined per 0.1 m^2), allowed establishing a lake typology, which is displayed in Figure 1 and comprises 31 lake types defined by combining classes of metabolic potential and classes of percent sensitive species. Many intermediate ecological conditions may exist, from low metabolic potentials without sensitive species to strong potentials without sensitive species or with less than 30% sensitive species. For instance where sensitive species dominate, even if the metabolic potential is low ($IOBL_{0.16}$<10), sediments have a well preserved functional status, but are not very productive. Results of two surveys of the Bay of Sciez, Lake Léman, conducted in the 1960s and 1980s, mainly differed by percent sensitive species, which were generally more elevated in the 1960s, even though the metabolic potentials were similar. The sensitive species in Lake Léman are also AED species (Table II), which is consistent with the well-known presence of upwellings of groundwater (below-lake springs) at 70 m depths in the vicinity of the Bay of Sciez. The increasing pollution was thus more detrimental to sensitive and AED species than to the metabolic potential, demonstrating the great resistance of such great

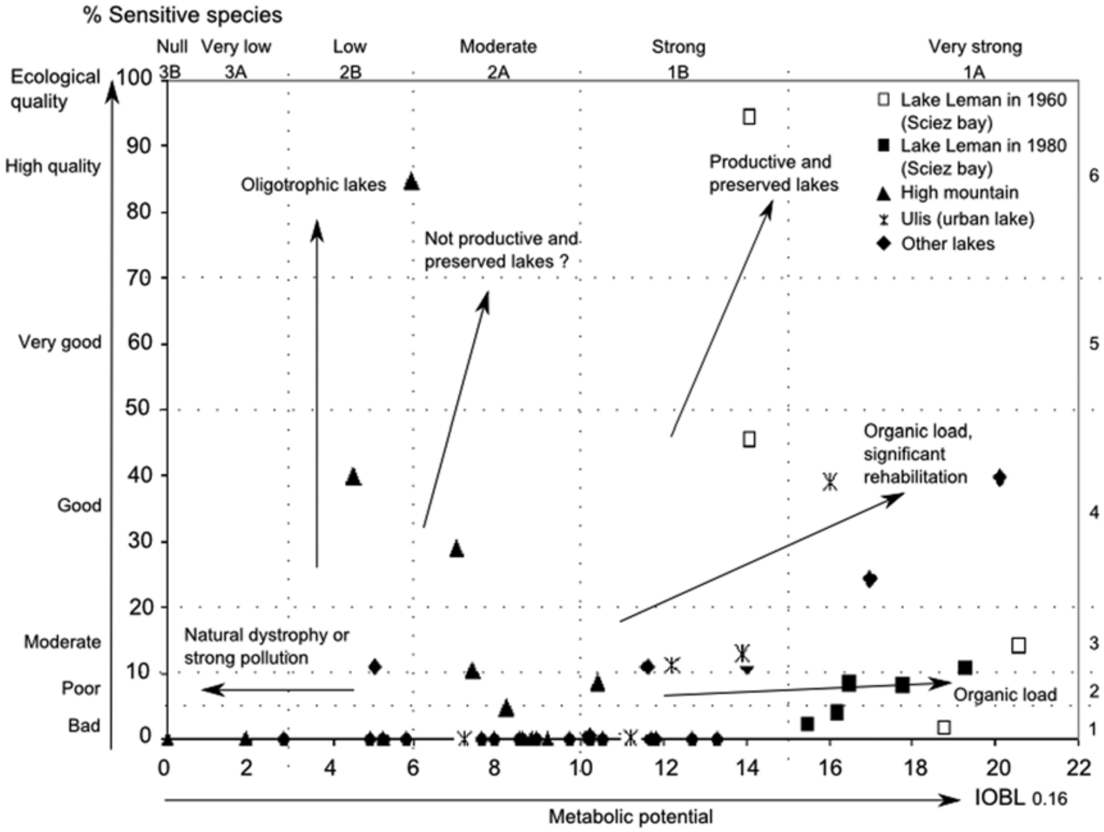

Fig. 1. Representation and interpretation of the conditions of deep-water sediments sampled from 30 French lakes (55 surveys) in a two dimensional diagram of percent sensitive species (% Sensitive species, Y axis) vs. the values of the $IOBL_{0.16}$ index (X axis) from samples retained by the 0.16 mm mesh size, which describe the metabolic potential.

Alpine lakes to pollution. Another example, AED species *Rhyacodrilus falciformis* (Table II), was frequently collected in Lake Léman in the 1910s at every depth, including the deepest zones (Piguet, Brestcher 1913). This species was not abundant in the 1960s in the Bay of Sciez (2.6%) and not found at all in the 1980s.

Additional examples illustrating the comprehensive features of the IOBL index can be drawn from the literature. Using the data published by Nagell *et al.* (1977) for Lake Vänern, Sweden (Fig. 2), each inlet embayment of the lake can be treated as a "specific lake". According to the data, the percent sensitive species metric is the first to respond to an increasing gradient of pollution, from unpolluted to moderately polluted. The metabolic potential of sediments is then altered in the second phase, when the pollution becomes "moderate to strong". Conversely, the central part of Lake Michigan (USA), shows well preserved conditions according to the oligochaete typology (Fig. 2), which is consistent with the findings of Howmiller (1974a), and the same conclusions apply to the preserved deep areas of Lake Konnevesi, Finland (Särkkä 1972). All these examples (Fig. 1 and 2) illustrate the complementary features of the IOBL index associated with percent sensitive species and helping gain a good understanding of the functioning and of the quality status of lakes.

In the case of lake systems with very low metabolic potentials, the species from groups 2 and 3 in Table II generally help distinguish between the pollution and dystrophy effects. In some particular cases where characteristic species are not abundant, it is always possible to refer to other components of the lake system, for example planktonic assemblages or organic load. For example in ten Wisconsin lakes studied by Howmiller (1974b), no species from groups 2 and 3 were found alone and sensitive species were absent (Table III). By comparing $IOBL_{0.5}$ values and physico-chemical factors such as total phosphorus or organic seston contents in water, the need to separate organically enriched lakes (null to low $IOBL_{0.5}$ values, but high organic and phosphorus contents) from "oligotrophic" lakes (null to low $IOBL_{0.5}$ values, but low organic and phosphorus contents) became evident.

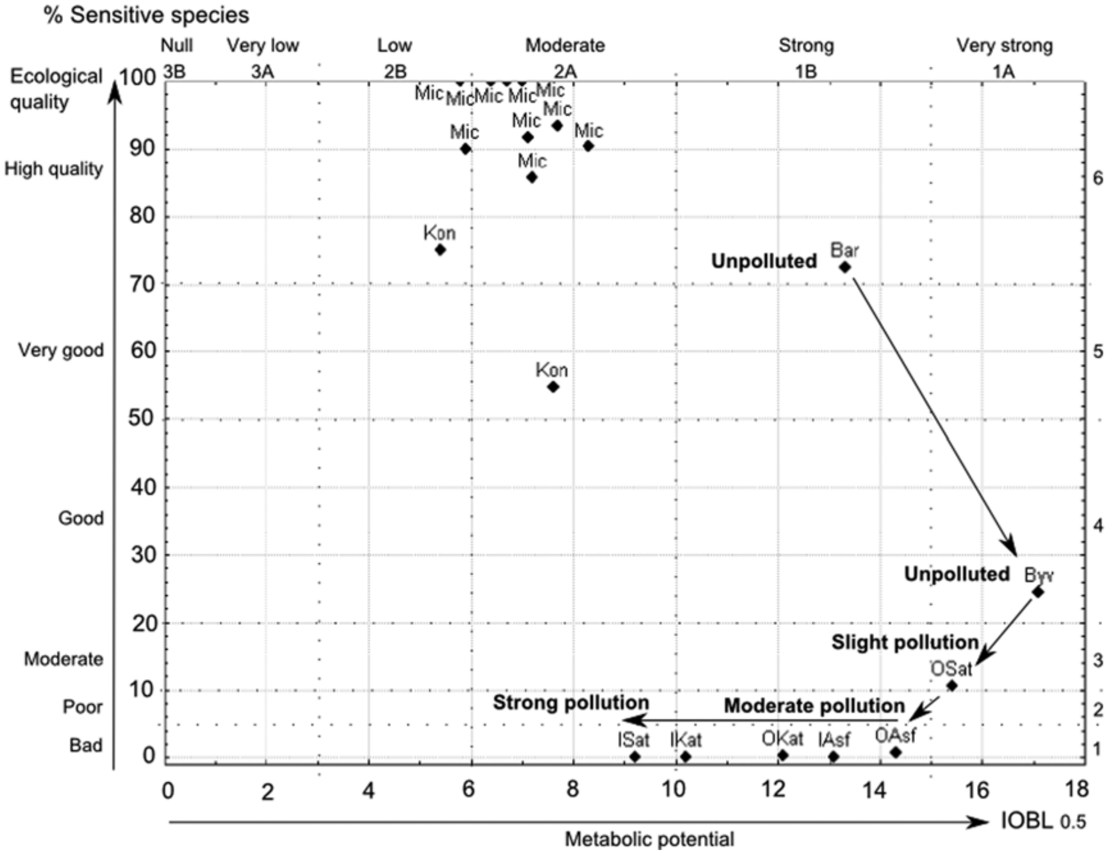

Fig. 2. Representation of various inlets of Lake Vänern (after Nagell *et al.* 1977), and sites in a central part of Lake Michigan (after Howmiller 1974b) and Lake Konnevesi (after Särkkä 1972) in a two-dimensional diagram of percent sensitive species (% Sensitive species, Y axis) vs. the values of the $IOBL_{0.5}$ index (X axis) from samples retained by the 0.5 mm mesh size, which describe the metabolic potential. Abbreviations: Bar, Byv, Osat, Oasf, Iasf, Okat, Ikat, ISat: various inlets of Lake Vänern; Kon: Lake Konnevesi; Mic: central parts of Lake Michigan.

The typology presented is relatively robust and operational, and furthermore, it provides a framework for comparing the functioning and ecological quality states of lake ecosystems (Fig. 3). Not only does the typology indirectly quantify the effects of anthropogenic perturbations, but it also enables to predict the resistance capacity of the system to pollution. In this way, the typology also facilitates defining functional objectives to be preserved or restored in various types of lakes, including dam reservoirs or small urban lakes. For example, in the case of Lake Vänern, an objective of $IOBL_{0.5} > 12$ (class 1B, 'strong potential', Fig. 2) might be selected for deep-water sediments, as this metabolic potential level can be found in unpolluted parts of the lake. In the case of urban lakes, the lone relatively preserved system (no polluted wet-weather inflows) out of 5 small urban lakes investigated in France (Ulis, the Paris region) showed an $IOBL_{0.16} = 16$, with 39.3% of sensitive species (Fig. 1), which could be retained as a realistic model for rehabilitation of the four

Table III. Calculated oligochaete index IOBL0.5 and percent number of sensitive species (% SSp) on 10 Wisconsin lakes (mesh-size sieves: 0.5 mm); TP: total phosphorus contents of waters; Org. Sest: organic seston of waters; after Howmiller (1974b) and Lafont (1989).

Lakes	$IOBL_{0.5}$	% S_{Sp}	TP [µg dm^{-3}]	Org. Sest. [mg dm^{-3}]
"Dystrophic"				
Round	0	0	< 10	1-2
Trout S.	1.9	0	< 10	1-2
Devils	7.3	0	< 10	1-2
Trout N.	7.7	0	< 10	1-2
Crystal	9.2		< 10	1-2
Polluted				
Delavan	0	0	> 50	> 5
Yellow	5.2	0	> 50	> 5
Winnebago	5.3	0	> 50	> 5
Kegonsa	5.8	0	> 50	> 5
Green	7.2	0	> 50	> 5

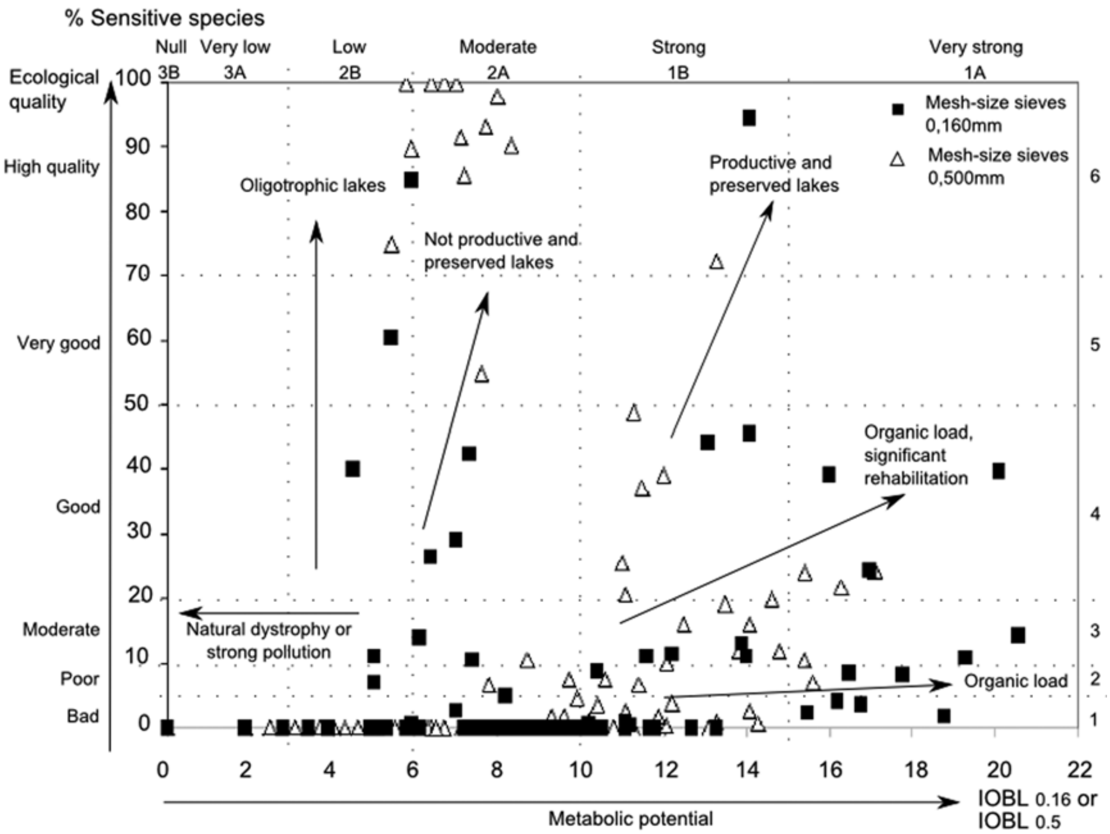

Fig. 3. Representation and interpretation of the conditions of various deep-water sediments sampled in various lakes from France or other countries (165 surveys) in a two dimensional diagram of percent sensitive species (% Sensitive species, Y axis) vs. the values of the IOBL index (X axis) from samples retained by the 0.5 mm and 0.16 mm mesh sizes (IOBL0.5, IOBL0.16), which describe the metabolic potential.

other lakes. The validity of this model is currently being investigated in some urban lakes (stormwater management ponds) in the Toronto region in Canada where the oligochaete methodology is tested as both a general stand-alone guideline and also as a component incorporated into Chapman's sediment triad approach (Tixier *et al.* 2011a; 2011b).

4. Discussion

4.1. Harmonization system (HS)

The basic idea of the HS followed Chapman's sediment triad approach (Chapman 1990), which was developed as an innovative method consisting of three factors: sediment chemistry, sediment ecotoxicology, and benthic organisms. The advantages, limitations and future developments of the HS have been thoroughly discussed earlier (Lafont *et al.* 2010; Table IV). The HS was intended to integrate operational indices beyond those used in the French standards. It might be eventually used for intercalibration projects required to meet large-scale water biomonitoring requirements, provided that all the methods are fully operational in three aspects: (i) well-documented ecological meaning, (ii) standardized or with reproducible protocols, and (iii) transferred to end-users for routine use. Our definition of "the operational index", as well as the term "ecological indicator", might be subject to sound debates (Heink, Kowarik 2010). However, based on our experience and stimulating discussions with colleagues, who are facing confusing results from methods that are not "operational", we suggest that a methodology complying with the three above-mentioned attributes has greater chance to be relevant without ecological ambivalence, routinely useable and reproducible on a large scale, rather than methods which lack adequate background, especially with respect to transfer to end-users. Furthermore it is necessary to assign greater importance to the aquatic habitats that can physically store the pollutants (fine sediments and the porous matrix). The storage of pollutants can often mask the reality of the situation and lead to an optimistic view of the actual ecological situation when indices related to

Table IV. Summary of concepts, associated tools and future actions. Legend: R: research; R&D: research and development; D: development; HS: harmonization system; EA: ecohydrological approach; FTrs: functional traits; EP: ecological potential.

Concepts and tools	Needs
HS (biological components concept)	D = R&D; add other compartments (eutrophication, porous matrix); test on other datasets; but no need for R on qualitative tools, only develop the integration of functional tools.
EA (EASY concept, LOUE)	R = R&D > D; add other oligochaete FTrs and other living assemblages; ongoing research on the role of interactions between physical and chemical factors; improve the notion of EP; establish a numerical version of the EASY concept.
Riverscape concept	R&D > D; test the methodology on various systems; refine the riverscape types.
Lake approach	R = R&D = D; add benthic and plankton data to the typology; need for R and R&D in reservoirs and urban lakes, including ecotoxicological research; need for routine use of Lake approach (D); consider smaller mesh-size sieving (0.25 or 0.16 mm).
All approaches	R = R&D: collaboration with human and social sciences and restoration practitioners for defining the best adapted choices for restoration or preservation purposes; writing methodological guides and taxonomical identification keys.

other system components give high values for the ecological status. The storage of pollutants is an ecological time-bomb, because such pollutants can be released as ambient conditions change.

4.2. Ecohydrological approach (EA)

The discussion of the EA approach with functional traits FTrs and the ecological potential EP calculation was presented elsewhere (Lafont *et al.* 2010; Table IV). The ongoing research focuses on the establishment of new FTrs and extension of the concept of FTr to other biocenoses beyond the oligochaetes. This process is already under way in the case of macrophyte assemblages (Trémolières, pers. com.). The FTrs were not conceived for assessing chemical or physical disturbances, but for integrating the effects of interactions between chemical and physical factors, in particular for the role of water exchanges between surface water and groundwater. In this case, the ecological status assessment is abandoned and we try to understand which interactions between physical and chemical factors are important to support biodiversity. Consequently, the EA approach diverges from the traditional bioassessment methodologies recently developed (Gabriels *et al.* 2010; Sánchez-Montoya *et al.* 2010; Torrisi *et al.* 2010). The notion of the ecological potential EP has to be reviewed by adding other elements than oligochaete assemblages. Both approaches, based on FTrs and riverscape typology, are integrated in the ecohydrological approach proposed by Zalewski (2006) and Zalewski and Wagner (2008). The conceptual framework is established by the EASY concept (Lafont 2001), and the role of hydrologic and geomorphologic factors is fully integrated (Schmitt *et al.* 2011). Biodiversity includes processes, which have not been yet addressed, because we lack the knowledge of such aspects as microbial communities. The FTrs can only give a rough idea of these processes. The FTr4 for example represents a functional trait corresponding to the situation where metabolic processes are very active. Microcosm experiments in porous habitats contaminated by urban sewage showed that Tubificidae oligochaetes (= FTr4) were very effective in activating bioturbation and metabolic processes (Datry *et al.* 2003; Mermillod-Blondin *et al.* 2003; Nogaro *et al.* 2006; 2009).

4.3. Lake approach (LA)

The LA was initially conceived for lake restoration purposes by providing examples of preserved functioning which were likely to be used as models for preservation actions. The LA was put forward as an alternative to more traditional approaches, based on qualitative lake biotic indices or on species indicating the trophic status (see Wiederholm 1980; Lang 1984; 2010; Milbrink *et al.* 2002; Rossaro *et al.* 2007; Beck, Hatch 2009; Gabriels *et al.* 2010). A very similar approach was developed by Verneaux *et al.* (2004) and Borderelle *et al.* (2008) (Lake Biotic Index LBI). Moreover, these authors highlighted a concept of the "lake biogenic capacity", similar to the "metabolic potential", with a lacustrine index varying from 0 to 20, which is similar to the IOBL. Furthermore, we have been aware of the fact that index-based methodologies compress the ecological information and therefore, need to be incorporated into a more global framework. Consequently, the IOBL was originally intended to be integrated into a multidisciplinary assessment (Lafont 1989), and is now integrated into Chapman's sediment quality triad in studies of constructed urban lakes (Tixier *et al.* 2011a; 2011b).

The discussion of 'the reference system' concept is beyond the scope of this paper. Furthermore, this concept is ambiguous in the case of lakes, because naturally dystrophic lakes with low biodiversity are in a preserved state, and attempts to increase their biodiversity represent an anthropogenic alteration. We have avoided defining the "reference systems" and preferred to define functional states that are clearly identified as preserved (= devoid of human alterations) on the basis of field survey findings examined by experienced experts (Figs. 2 and 3), and such states are likely to be considered as models for preservation or rehabilitation actions during the aquatic phase of a lake. Furthermore, the identification of oligochaete assemblages to the species level allows gaining information on the functioning of lakes, and paying special attention to the dominant species might be more informative than considering the entire bottom fauna (Särkkä 1972). Moreover, the importance of species identification in water quality monitoring has been emphasized by Resh and Unzicker (1975) earlier.

Research management guidance

We follow a multi-disciplinary perspective when suggesting what should be developed or abandoned, and where research is needed or not (Table IV). It empowers us to optimize the organization of our multi-disciplinary research and to avoid unwanted "dispersion" of scientific activities. Future research should address sound practical adaptations for development by the way of R&D, and such a development can yield new research directions. It means that a research team has to bring together researchers, engineers, technicians, and ensure their collaboration in: (i) conducting research projects, (ii) establishing structural links with end-users from the public and private sectors, (iii) contributing to minimizing the conflicting demands of water managers and ecologists by optimizing the role of stakeholders, and, (iv) reducing the time-lag between research and its application. In addition, our experience indicates that it is easy to engage in discussions with stakeholders and managers, or even decision-makers, provided that we use "sensible" language and do not overuse scientific jargon. We have also developed a protocol for technology transfer (Table V), which consists of six stages; the last stage allows accelerating this cycle by initiating improvements and generating new ideas and new research. In Stage 3, an official agreement for technology transfer might be signed with public or private end-users.

General conclusions

All our approaches and biomonitoring tools (Tables I) have been certainly dedicated to establishing ecological diagnosis, but originally, they were mostly conceived as guidelines for rehabilitation or restoration purposes, and integrated in the general domain of restoration ecology (see Western 1992) of which major component is ecohydrology. Furthermore, we suggest that using the existing operational tools encompassed in scientifically open typologies might save a significant amount of time in solving the problem of the conflicting needs of water managers and ecologists. For example, the recently developed ecohydrological approach employing FTrs and EP is already operational and available to colleagues and end-users, even though the research in this field is still continuing and began only 6 years ago. Furthermore, we claim that multidisciplinary research and development cannot be done without disciplinary excellence. A good way to unite various disciplines is to present a common and shared conceptual framework, including a protocol for research management (Tables IV and V). We believe this top-down approach provides us with the means to quickly acquire the most up-to-date tools, gives us the possibility to incorporate the older tools, and allows us to fully benefit from the experience of other colleagues. We suggest that such an approach is beneficial during 'lean periods' of limited financial support for research and when there are urgent needs to quickly respond to water managers' demands. In agreement with the technology transfer protocol in Table V, it is possible to organize training sessions for those who need to learn new methodologies, integrate their own methodologies into a flexible conceptual framework, or incorporate some of the tools we proposed into their own conceptual framework. Finally, we firmly believe in a holistic approach benefiting from the richness and complementarity of all methodologies, as long as they are fully operational, rather than a reductionist approach.

Table V. Stages of technology transfer (modified after Vivier 2006).

Stages	Needs
1	Research projects (R); elaboration of scientific fundamentals and concepts; papers in scientific journals.
2	Research & Development (R&D) projects and actions; testing of tools in various situations; papers in scientific and technical journals.
3	Research & Development (R&D); establishment of technical guidelines, standardization; convention of technology transfer with end-users.
4	Development (D); training of end-users.
5	Development (D); checking of end-user results and their conformity to a quality control chart.
6	R, R&D and D; experience return → improvements, new ideas, new research.

Acknowledgements

The presented paper represents a synthesis of research findings produced in studies spanning many years. Such work has been supported by numerous funding providers and it is impossible to list them all without forgetting some. The authors would like to thank Dr. Liz Day, Kamila Belka, and the two anonymous reviewers, for their benevolent and constructive comments.

References

AFNOR 2002. *Qualité de l'eau - Détermination de l'indice oligochètes de bioindication des sédiments (IOBS)*. Norme Française NF T 90-390.

AFNOR 2004a. *Essai des eaux: détermination de l'indice biologique global normalisé (IBGN)*. Norme française NF T 90-350.

AFNOR 2004b. *Qualité de l'eau – Détermination de l'indice poissons en rivière (IPR)*. Norme Française NF T 90-344.

AFNOR 2005. *Qualité de l'eau – Détermination de l'indice oligochètes de bioindication lacustre (IOBL)*. NF T90-391.

AFNOR 2007. *Détermination de l'indice biologique diatomées (IBD)*. Norme française NF T 90-354.

Bazzanti, M., Lafont, M. 1985. Variabilité des soies spermathécales de *Potamothrix heuscheri* (Bretscher) (Oligochaeta, Tubificidae) dans le lac de Nemi et remarques sur la relation entre cette espèce et le degré de trophie des lacs. *Annls. Limnol.* **2**, 107-115.

Beck, M.W., Hatch, L.K. 2009. A review of research on the development of lake indices of biotic integrity. *Environ. Rev.* **17**, 21-44.

Blandin, P. 1986. Bioindicateurs et diagnostic des systèmes écologiques. *Bulletin d'Ecologie* **17**, 215-307.

Borderelle, A.L., Gilmette, C., Lovy, C., Gerdeaux, D., Verneaux, V. 2008. Macroinvertebrate δ13C variability analysis for the assessment of lake trophic functioning. *Fundamental and Applied Limnology, Archiv für Hydrobiologie* **172**/4, 289-300.

Bouleau, G., Argillier, C., Souchon, Y., Barthélémy, C., Babut, M. 2009. How ecological indicators construction reveals social changes. The case of lakes and rivers in France. *Ecological Indicators* **9**, 1198-1205.

Breil, P., Lafont, M., Fletcher, T.D., Roy, A. 2008. Aquatic ecosystems. In: Fletcher, T.D., Deletić, A. [Eds] *Data requirements for Integrated Urban Water Management*, Chapter 20. Taylor and Francis Group, London, pp. 259-272.

Chapman, P.M. 1990. The sediment quality triad approach to determining pollution-induced degradation. *The Science of the Total Environment* **97/98**, 815-825.

Chapman, P.M. 2007. Do not disregard the benthos in sediment quality assessments! *Marine Pollution Bulletin* **54**, 633-635.

Collado, R., Schmelz, M. 2001. Oligochaete distribution patterns in two German hardwater lakes of different trophic state. *Limnologica* **31**, 317-328.

Cullen, P. 1990. The turbulent boundary between water science and water management. *Freshwater Biology* **24**, 201-209.

Datry, T., Hervant, F., Malard, F., Vitry, L., Gibert, J. 2003. Dynamics and adaptive responses of invertebrates to suboxia in contaminated sediments of a stormwater infiltration basin. *Archiv für Hydrobiologie* **156**, 339-359.

De Pauw, N., Vanhooren, G. 1983. Method for biological quality assessment of watercourses in Belgium. *Hydrobiologia* **100**, 153-168.

EU 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities* L327, 1-72.

Gabriels, W., Lock, K., De Pauw, N., Goethals, P.L.M. 20010. Multimetric Macroinvertebrate Index Flanders (MMIF) for biological assessment of rivers and lakes in Flanders (Belgium). *Limnologica* **43**, 199-207.

Heink, U., Kowarik, I. 2010. What are indicators? On the definition of indicators in ecology and environmental planning. *Ecological Indicators* **10**, 584-593.

Howmiller, R.P. 1974a. Composition of the oligochaete fauna of central Lake Michigan. In: *Proc. 17th Conf. Great Lakes Res.*, pp. 589-592.

Howmiller, R.P. 1974b. *Studies on aquatic Oligochaeta of inland waters of Wisconsin*. Wisconsin Academy of Sciences, Arts and Letters 62, 337-356.

Hynes, H.B.N. 1960. *The biology of polluted waters*. Liverpool University Press, Liverpool.

Jones, J.B., Mulholland, P.J. 2000. *Streams and ground waters*. Academic Press, San Diego.

Juget, J., Lafont, M., Mouthon, J., Gerdeaux, D. 1995. Structure des communautés benthiques et pisciaires. In: Pourriot, R., Meybeck, M. [Eds] *Limnologie générale*. Masson, Paris, Milan, Barcelone, pp. 494-513.

Lafont, M. 1989. *Contribution à la gestion des eaux continentales: utilisation des oligochètes comme descripteurs de l'état biologique et du degré de pollution des eaux et des sédiments*. Doctorat d'Etat ès Sciences, Université Lyon 1.

Lafont, M. 2001. A conceptual approach to the biomonitoring of freshwater: the Ecological Ambience System. *Journal of Limnology* (Suppl. 1) **60**, 17-24.

Lafont, M. 2007. *Interprétation de l'indice lacustre oligochètes IOBL et son intégration dans un système d'évaluation de l'état écologique*. Cemagref/ MEDAD. (File .pdf available on demand)

Lafont, M., Juget, J. 1985. Les Oligochètes de quelques lacs français: propositions en vue de leur utilisation pratique pour apprécier l'état biologique des sédiments profonds. *Verh. Internat. Verein. Limnol.* **22**, 3019-3023.

Lafont, M., Vivier, A. 2006. Oligochaete assemblages in the hyporheic zone and coarse surface sediments: their importance for understanding of ecological functioning of watercourses. *Hydrobiologia* **564**, 171-181.

Lafont, M., Juget, J., Rofes, G. 1991. Un indice biologique lacustre basé sur l'examen des oligochètes. *Rev. Sci. Eau* **4**, 253-268.

Lafont, M., Vivier, A., Nogueira, S., Namour, P., Breil, P. 2006. Surface and hyporheic oligochaete assemblages in a French suburban stream. *Hydrobiologia* **564**, 183-193.

Lafont, M., Grapentine, L., Rochfort, Q., Marsalek, J., Tixier, G., Breil, P. 2007. Bioassessment of wet-weather pollution impacts on fine sediments in urban waters by benthic indices and the sediment quality triad. *Wat. Sci. Tech.* **56**, 13-20.

Lafont, M., Marsalek, J., Breil, P. 2008. Urban aquatic habitat characteristics and functioning. In: Wagner, I., Marsalek, J., Breil, P. [Eds.] *Aquatic habitats in integrated urban water management*, Chapter 2. Urban water Series-UNESCO-IHP. Taylor and Francis Group, The Netherlands, pp. 9-24.

Lafont, M, Jézéquel, C., Vivier, A., Breil, P., Schmitt, L., Bernoud, S. 2010. Refinement of biomonitoring of urban water courses by combining descriptive and ecohydrological approaches. *Ecohydrol. Hydrobiol.* **10**, 3-11.

Lang, C. 1984. Eutrophication of lakes Léman and Neuchâtel (Switzerland) indicated by oligochaete communities. *Hydrobiologia* **115**, 131-138.

Lang, C. 2010. Indices basés sur les oligochètes et les chironomides indiquant la restauration écologique des sédiments du Léman. *Bull. Soc. vaud. Sc. nat.* **91**, 283-300.

Lenar-Matyas, A., Lafont, M., Lapuszek, M., Poulard, C. 2009. Alleviating effects of the technical training of mountainous rivers and streams in the urban areas. *Infrastructure and Ecology of Rural Areas* **9**, 241-251 [in Polish].

Macan, T.T. 1974. Running water. *Mitt. Internat. Verein. Limnol.* **20**, 301-321.

Mermillod-Blondin, F., Creuzé des Châtelliers, M., Gérino, M. 2003. Effects of the interaction between tubificid worms on the functioning of hyporheic sediments: an experimental study in sediment columns. *Archiv für Hydrobiologie* **156**, 203-223.

Milbrink, G., Timm, T., Lundberg, S. 2002. Indicative profundal oligochaete assemblages in selected small Swedish lakes. *Hydrobiologia* **468**, 53-61.

Nagell, B., Landhal, B.C.C., Lann, F.H. 1977. Quantitative and qualitative composition of bottom fauna in polluted parts of lake Vänern and some aspects of water quality indices. *Vatten* **4**, 434-441.

Nogaro, G., Mermillod-Blondin, F., François-Carcaillet, F., Gaudet, J.P., Lafont, M., Gibert J. 2006. Invertebrate bioturbation can reduce the clogging of sediment: an experimental study using infiltration sediment columns. *Freshwater Biology* **51**, 1458-1473.

Nogaro, G., Mermillod-Blondin, F., Valett, M.H., François-Carcaillet, F., Gaudet, J.-P., Lafont, M., Gibert, J. 2009. Ecosystem engineering at the sediment-water interface: bioturbation and consumer-substrate interaction. *Oecologia* **161**, 125-138.

Piguet, E., Bretscher, K. 1913. Oligochètes. *Catalogue des invertébrés de la Suisse* **7**, 1-214.

Poulard, C., Lafont, M., Lenar-Matyas, A., Lapuszek, M. 2010. Flood mitigation designs with respect to river ecosystem functions. A problem oriented conceptual approach. *Ecological Engineering* **36**, 69-77.

Resh, V.H., Unzicker, J.D. 1975. Water quality monitoring and aquatic organisms: the importance of species identification. *Journal of Water Pollution Control Federation* **47**, 9-19.

Rossaro, B., Marziali, L., Cardoso, A.C., Solimini, A., Free, G., Giacchini, R. 2007. A biotic index using macroinvertebrates for Italian lakes. *Ecological Indicators* **7**, 412-429.

Sánchez-Montoya, M.M., Vidal-Abarca, M.R., Suárez, M.L. 2010. Comparing the sensitivity of diverse macroinvertebrate metrics to a multiple stressor gradient in Mediterranean streams and its influence on the assessment of ecological status. *Ecological Indicators* **10**, 896-904.

Särkkä, J. 1972. The bottom macrofauna of the oligotrophic lake Konnevesi, Finland. *Ann. Zool. Fennici* **9**, 141-146.

Schmitt, L., Lafont, M., Trémolières, M., Jézéquel, C., Vivier, A., Breil, P., Namour, P., Valin, K., Valette, L. 2010. Using hydro-geomorphological typologies in functional ecology: preliminary results in contrasted hydrosystems. *Physics and Chemistry of the Earth* **36**, 539-548.

Sládeček, V. 1973. System of water quality from the biological point of view. *Arch. Hydrobiol. Beih.* **7**, 1-218.

Tixier, G., Rochfort, Q., Grapentine, L., Marsalek, J., Lafont, M. 2011a. In search of effective bioassessment of urban stormwater pond sediments: enhancing the "sediment quality triad" approach with oligochaete metrics. *Water Science and Technology* **64**, 1503-1510.

Tixier, G., Lafont, M., Grapentine, L., Rochfort, Q., Marsalek, J. 2011b. Ecological risk assessment of urban stormwater ponds: Literature review and proposal of a new conceptual approach providing ecological quality goals and the associated bioassessment tools. *Ecological Indicators* **11**, 1497-1506.

Torrisi, M., Scurri, S., Dell'Uomo, A., Cocchioni, M. 2010. Comparative monitoring by means of diatoms, macroinvertebrates and chemical parameters of an Apennine watercourse of central Italy: The river Tenna. *Ecological Indicators* **10**, 910-913.

Verneaux, J., Tufféry, G. 1967. Une méthode zoologique pratique de détermination de la qualité biologique des eaux courantes. Indices biotiques. *Ann. Sci. Univ. Besançon, Zool.* **3**, 79-90.

Verneaux, V., Verneaux, J., Schmitt, A., Lovy, C., Lambert, J.C. 2004. The Lake Biotic Index (LBI): an applied method for assessing the biological quality of lakes using macrobenthos; the Lake Châlain (French Jura) as an example. *Ann. Limnol. – Int. J. Limnol.* **40**, 1-9.

Vivier, A. 2006. *Effets écologiques de rejets urbains de temps de pluie sur deux cours d'eau périurbains de l'ouest lyonnais et un ruisseau phréatique en plaine d'Alsace*. Thesis, L.P. University, Strasbourg, France.

Western, D. 1992. The biodiversity crisis: a challenge for biology. *Oikos* **63**, 29-38.

Wiederholm, T. 1980. Use of benthos in lake monitoring. *JWPCF* **52**, 537-547.

Woodivis, F.S. 1964. The biological system of stream classification used by the Trent River Board. *Chem. Ind.* **14**, 443-447.

Zalewski, M. 2006. Ecohydrology – an interdisciplinary tool for integrated protection and management of water bodies. *Archiv für Hydrobiologie, Supplementum* **158**/4, 613-622.

Zalewski, M., Wagner, I. 2008. Ecohydrology of urban aquatic ecosystems for healthy cities. In: Wagner, I., Marsalek, J., Breil, P. [Eds.] *Aquatic habitats in integrated urban water management*, Chapter 6. Urban water Series – UNESCO-IHP. Taylor and Francis Group, The Netherlands, pp. 95-106.