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Using Benthic Macroinvertebrates to Assess Ecological Status of Lakes Current Knowledge and Way Forward to Support WFD Implementation

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Key conclusions

1. While practical and WFD-compliant assessment tools using macroinvertebrates are already in use to assess the ecological quality of rivers, in many European countries there are currently no working macroinvertebrate assessment systems for lakes.
2. The lack of WFD compliant macroinvertebrate assessment tools was identified as one of the major ecological ‘knowledge gaps’ impeding the full assessment of ecological quality of lakes in a literature review carried out within the EU project REBECCA.
3. The current lack of knowledge is also limiting the fulfillment of the EU-wide intercalibration of the lake ecological quality assessment systems in Europe, and thus compromising the basis for setting the environmental objectives as required by the WFD.
4. In the food webs of lakes, benthic invertebrates have an intermediate position between primary producers and destruents on one side, and higher trophic levels (as fish) on the other side. Hence, they play an essential role in key ecosystem processes (food chain dynamics, productivity, nutrient cycling and decomposition). Rare species, which partially indicate reference conditions, form a link to the biodiversity objectives of the EU Habitat Directive.
5. As benthic invertebrates respond sensitively not only to pollution, but also to a number of other human impacts (hydrological, climatological, morphological, navigational, recreational, and others), they could potentially be used for a holistic indication system for lake ecosystem health. Their ubiquitous presence and their relative longevity may be seen as strong points recomennding them for use in an indication system.
6. Benthic invertebrates show considerable spatial variation with lake depth, across habitats, and across lakes. In order to distiguish human pressures from natural variability, the habitat factors driving natural variability have to be understood. As littoral, sub-littoral and profundal invertebrate communities are driven by different governing factors, they probably indicate different human disturbances.
7. In order to analyze the response of benthic invertebrates to human degradation of habitat conditions, there is an urgent need of developing European-level research able to place macroinvertebrate assessment into the framework of lake ecosystem functioning. Such European-level research could partially use knowledge from regional research on the indication of lake acidification by littoral invertebrates, or on the indication of eutrophication by profundal invertebrates.
8. As practical tools with high predictive power have to be derived, research should be designed to both improve ecological understanding and to lead to better ecological assessment methods.
9. Key elements under consideration should include: the role of invertebrates in the material cycling, the functional role of littoral invertebrates within the ecosystem in different lake types, their response to watershed and shoreline alterations, the importance of spatial and temporal factors on assemblage dynamics and relative bioindicator behaviour, their influence on reference conditions, habitat constraints on species traits, taxonomic and methodological limitations.
10. The uneven geographical distribution of lakes in Europe, the peculiarity of Mediterranean lakes, the impacts of climate change on lakes, and the great number of reservoirs in some Mediterranean regions should be taken into account.

1 Introduction and scope of the report

Background

Directive 2000/60/EC, commonly known as the Water Framework Directive (WFD) has initiated a change in both the concept of water quality and its assessment throughout Europe. It has started a shift from the mindset of Europe's water resource being a product which may be monitored chemically to ensure its suitability for human use to one that regards water as a heritage. A more holistic assessment by member states of functioning and structure of aquatic ecosystems is now legally required, which include the following biological elements: fish, phytoplankton, macrophytes and phytobenthos and benthic macroinvertebrates (Irvine et al. 2002; Heiskanen et al. 2004). For member-states this represents a highly complex task, as traditionally national and regional monitoring programs included only a subset of these elements.

Benthic invertebrates play an essential role in key processes within lake ecosystems (food chain dynamics, productivity, nutrient cycling and decomposition: Reice & Wohlenberg 1993). Benthic invertebrates form an important link between primary producers, detrital deposits and higher trophic levels in aquatic food webs (Brinkhurst 1974, Stoffels et al 2005). Hence,, any environmental changes in lakes, for example in nutrient concentrations, would be reflected by changes in the structure of the benthic invertebrate community (Carvalho et al. 2002). This means that benthic invertebrates may potentially indicate eutrophication, as planktonic communities, but in addition several other modes of lake degradation (see chapter 2). In consequence, the more holistic assessment based on benthic invertebrates is expected to result in different classifications than that based on planktonic communities, especially for lakes subjected to multiple impacts. The study of benthic invertebrates in lakes is traditionally segregated by depth zone: littoral, sub-littoral and profundal, see table 1, as these zones are generally colonized by distinct communities, which also respond in different ways to specific impacts on lakes (see next chapter)..

Recent extensive reviews of the current state-of-the-art of ecological water quality assessment systems in Europe have revealed that, while practical (and WFD-compliant) assessment tools using macroinvertebrate parameters are already in use to assess the ecological quality of rivers, in many European countries there are currently no working macroinvertebrate assessment systems for lakes (Cardoso et al. 2005, Nõges et al. 2005). Indeed, this has been recently identified as one of the major ecological 'knowledge gaps' impeding the full assessment of ecological quality of lakes as required by the WFD in a literature review carried out within the EU project REBECCA[1] (Heiskanen and Solimini 2005). The current lack of knowledge is also limiting the fulfillment of the EU-wide intercalibration of the lake ecological quality assessment systems in Europe, and thus compromising the basis for setting the environmental objectives as required by the WFD, particularly concerning quantification of the ecosystem impacts of nutrient loading pressures (i.e. eutrophication), which is the most wide-spread pressure on surface water ecological quality in Europe (EEA, 2003).

Document purpose

Despite their key role in aquatic ecosystems, macroinvertebrates are a neglected element in the development of an assessment system in lakes. Factors that may be largely responsible for this include their complex biotic structure, high temporal variability and the high substrate heterogeneity found in lakes. A solution needs to be found to understand natural variability so that anthropogenic impact may be identified and extracted from other sources of variation.

This report focuses on the ecological assessment of lakes using benthic macroinvertebrates. The report's purpose is to review the current knowledge, current challenges, identifying the research needs in the use of benthic invertebrates as indicators of lake ecological stautus. The way forward on the realisation of a lake holistic assessment tool based on benthic invertebrates is discussed in the light of the WFD requirements and implementation timetable.

Document structure

The review is structured by the major anthropogenic pressures affecting lakes: eutrophication, acidification and hydromorphological alterations. Current knowledge and examples of use are presented in the context of the required understanding needed to use benthic macroinvertebrates in lake assessment as required by Directive 2000/60/EC (see Table 2).

[1] Specific Targeted Research Project under the EC 6th Framework programme (SSPI-CT-2003-502158) http://www.environment.fi/syke/rebecca.

The current methodological perspectives and limitations in the use of benthic macroinvertebrates are discussed. The report ends with a succinct section summarising the main knowledge gaps and marking the way forward through key research needs.

Intended audience

This report is of interest to persons involved in the implementation of the Water Framework Directive, ecological assessment, freshwater ecologists and policy makers.

Table 1 Hypotheses for response of macroinvertebrate assemblages to pressures in different lake zones. Estimates are mostly based on scattered results, only the cells 'acidification – littoral' and 'eutrophication – profundal' are based on more extensive databases, but which are regionally restricted. *** = Sensitive response, ** = major response, * = minor response, 0 = no response. Question marks indicate especially high uncertainty of the respective hypothesis. Additionally to the pressures shown here, benthic invertebrate communities in lakes may be strongly altered by other pressures, too (see also table 8), as the immigration of invasive species, and by the input of toxic pollutants.

	Eutrophication	**Hydromorphological**	**Acidification**	**Combined**
Littoral	*	***	***	***?
Sub-littoral	**?	*?	**?	**?
Profundal	***	0	?	**?

Table 2 Definitions for high, good and moderate ecological status in lakes for the biological element benthic invertebrate fauna (Directive 2000/60/EC).

High status	**Good status**	**Moderate status**
The taxonomic composition and abundance correspond totally or nearly totally to the undisturbed conditions. The ratio of disturbance sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed levels. The level of diversity of invertebrate taxa shows no sign of alteration from undisturbed levels.	There are slight changes in the composition and abundance of invertebrate taxa compared to the type-specific communities. The ratio of disturbance sensitive taxa to insensitive taxa shows slight signs of alteration from type-specific levels. The level of diversity of invertebrate taxa shows slight signs of alteration from type-specific levels.	The composition and abundance of invertebrate taxa differ moderately from the type-specific conditions. Major taxonomic groups of the type-specific community are absent. The ratio of disturbance sensitive to insensitive taxa, and the level of diversity, are substantially lower than the type-specific level and significantly lower than for good status.

2 Review of work relevant to the assessment of lake ecological status using benthic macroinvertebrates

According to the WFD, the assessment of a water body based on specific biological element requires the identification of reference state for each type of water body. This typology should be based on physical and biological characteristics. Hence, for each member state or ecoregion a typology of lakes should be elaborated. These typologies are mostly based on physical (climatological, geological, morphological, or hydrological, including conductivity) criteria (Moss et al. 2003), with regionally differing resolution. Typologies based on the community composition of benthic invertebrates hardly exist, and are restricted to the regional scope. As biological lake types based on benthic invertebrates are mostly lacking, neither reference states can be defined today, nor degradation levels that can be assigned to a specific ecological status.

Thereby, the reference states are to contain type-specific biological communities that have been subject to very minor anthropogenic disturbance. Reference state should be considered equal to high ecological status. The Reference state is, subsequently, the benchmark used to calculate Ecological Quality Ratios. Guidance from the EU Common Implementation Strategy Working Group (Anonymous 2003) considered that the reference value for EQR calculation should be the most robust statistical parameter (e.g. median, arithmetic mean) most appropriate for the quality element chosen. The boundaries high-good and good-moderate are set in an intercalibration exercise required by the WFD. Reference state of biotic elements need to be supported by minimally disturbed hydromorphological and chemical conditions. The Reference State is, therefore, the benchmark from which all ecological classifications are to be made.

The identification of the reference status for a specific lake type based on benthic invertebrates is complicated by the facts that the composition of benthic invertebrate communities exhibits natural variation due to season, lake depth, meso-scale habitat structure, and also due to biotic effects (competition and predation).

Clear seasonal changes in community structure can be observed, which are primarily due to the life cycles of aquatic insects, but may be influenced by seasonal changes in habitat conditions, too. As most univoltine insects emerge in summer, these are hardly present in summer samples of benthic invertebrates, and the early larval stages present in autumnal samples can often not be determined taxonomically. However, the same phenology applies to benthic macroinvertebrates in streams and rivers, which have been very successfully used as bioindicators for more than a century. Moreover, the planktonic communities of lakes exhibit seasonal dynamics which is at least similar in strength.

Lake morphometry affects community structure of both macrophytes and macroinvertebrates (Duarte and Kalff, 1986; Rasmussen, 1988). While the terminology related to physical structure of lakes is large and and varies to some extent (Ruttner, 1953; Hutchinson, 1957; Wetzel, 2001) the depth profile of lakes can most simply be divided into the littoral, sub-littoral and profundal. Generally, the benthic zone of lakes can be divided along the depth profile into the littoral, sub-littoral and profundal zones. The littoral zone is defined as the nearshore lake bottom areas where emerged macrophytes grow. The sub-littoral zone is defined as the bottom area covered by submerged macrophyte or algal vegetation. Often, empty shells of molluscs are accumulated at its lower end (littoriprofundal) and thus form a specific sediment type. The lake bottom area extending deeper is called profundal zone, which consists of exposed fine sediment free of vegetation.

It would be expected that nutrient enrichment affects those zones in different ways (see table 1). It is known for a long time that profundal invertebrate communities are strongly influenced by the trophic state of a lake (Naumann 1921, Lenz 1925, Lundbeck 1936, Thienemann 1954, Brundin 1956, Sæther, 1979, Wiederholm, 1981; Aagaard, 1986). From recent research it may be hypothesized that eutrophication affects the sub-littoral zone to a generally less extent that the profundal, and the littoral zone even less (Brauns et al. subm. b). In contrast, hydromorphological alterations will affect most strongly the littoral zone, but the sub-littoral to a much lower extent (Brauns et al. subm. a). The profundal is probably hardly affected. Similarly, acidification probably mostly affects the upper zones of the lake (see ch. 2.4 on acidification). Due to these differences in the importance of specific pressures, the following chapters are mostly focused on one of these zones.

The concept of lake assessment based on benthic invertebrates, as delineated above, would be basically applicable to reservoirs, too. According to the European Water Framework Directive, these heavily modified water bodies should be assessed similarly as lakes, but the best 'ecological potential' should be used as a reference instead. In some Mediterranean member states, the majority of standing waters are reservoirs. Often, they are of great economical importance for the water supply for agriculture and urban centres. As most reservoirs are formed by damming up a stream or river, they generally exhibit a large catchment. This makes reservoirs highly sensitive for anthropogenic impacts originating in the catchment, as the input of nutrients, heavy metals or organic toxic pollutants from agriculture, industry, or settlements. According to the WFD, it should be examined if benthic invertebrates could serve as effective indicators for such impacts on reservoirs.

To date, there are only few data available on the colonization of reservoirs by benthic invertebrates, especially from Poland. It is known that the sediments in reservoirs show a distinct gradient along its main axis, with the coarsest sediments near its inflow (s) and the finest sediments near the dam. It may be assumed that this gradient is paralleled by a gradient in profundal invertebrate colonization. The littoral and sublittoral zones in typical reservoirs mostly lack any macrophytes or riparian trees, as the shores are steep and the water level changes frequently. Only the shores near the inflow(s) often exhibit more gentle slopes, which may allow the growth of higher vegetation. Density of littoral benthic invertebrates is very low under these conditions, and is largely determined by the dynamics of water level fluctuations. Also, the littoral zone offers scarce carbon resources for benthic invertebrates under these conditions (Black et al. 2003). Generally, the role if the littoral zone for the diversity and functioning of a lake ecosystem is probably strongly compromised by extensive and fast water level fluctuations.

2.1 Eutrophication and littoral assemblages

Nutrient enrichment is the most widespread pressure affecting European lakes. It is a significant challenge for the implementation of the WFD that the understanding of natural distribution patterns of littoral invertebrates, reliable determination of reference conditions, and response to increased nutrients have yet to be developed for littoral invertebrates. The limited investigations of littoral macroinvertebrate distribution and response to anthropogenic pressures (e.g. Tolonen et al., 2001; Brodersen, Dall and Lindegraad, 1998; Willén; Andersson and Söderbäck, 1997; (White and Irvine, 2003) in lakes is in marked contrast to their extensive use as indicators of water quality in rivers (Hellawell 1986; Rosenberg and Resh, 1993; Mason, 1996; Maitland, 1997); although even the use of invertebrates in flowing waters generates debate regarding the most appropriate techniques (Walley and Fontama, 1998a b; Barbour and Yoder, 2000; Wright 2000). Furthermore, the structural heterogeneity of lake littoral areas has been suggested to negate the feasibility of littoral macroinvertebrates for ecological assessment (Rasmussen, 1988; Harrison and Hildrew, 1998; Koskenniemi, 2000; Moss et al 2003). The use of littoral invertebrates in lakes assessment requires that ecological dose-response relationship to defined pressures are identified with sufficient reliability against the background of spatial and temporal heterogeneity (White and Irvine, 2003; Jones et al., 2006). For WFD application there needs to be reliable estimation of reference condition either averaged across lakes types (Baily et al., 2004), or modelled as a site-specific estimates (e.g. Wright 2000). Either approach presents particular challenges.

Separating effects of habitat structure, including depth, from that of nutrients per se is difficult, as these are often interrelated. Distinct invertebrate communities are often associated with particular sediment types (Cole and Wiegmann, 1983; Kangur et al., 1998; White and Irvine, 2003; Peeters et al., 2004; Stoffels et al.,2005) or macrophytes (Rooke, 1984; Cyr and Downing, 1988ab; Hanson, 1990; Tolonen et al., 2001; Van den Berg et al. 1997; Hinden et al., 2005; Krecker, 1939; Gerking, 1957; Gerrish and Bristow, 1979; Cheruvelil et al., 2000; Czachorowski and Kornijow (1993). Macrophyte and invertebrate communites also vary with depth (Kajak and Dusoge, 1975ab; Czachorski, 1989; Brodersen et al. 1998; Kornijow (1988, 1989b); with significant interaction with trophic state (Jupp and Spence 1977; Pieczyńska et al. (1999). Effects of nutrients on littoral invertebrate taxa richness may be confounded by the generally well established unimodel response of macrophytes to phosphorus (Sand-Jensen and Borum, 1991; Scheffer, 1998; Jeppesen et al., 2000).

Macrophytes can also provide protection from predation, mainly by fish (Crowder and Cooper, 1982; Gilinsky, 1984; Hanson, 1990; Diehl, 1992; Svensson et al., 1999) and lakes with fish may contain different invertebrate communities than those without (Hinden et al., 2005). Although fish species and year-classes vary in their ability to forage for macroinvertebrates among plants (Persson, 1987; Kornijów et al., 2005), submerged plants and reed beds can also act as a cover and source of food for, particularly, small fish (Diehl and Kornijów, 1998; Okun and Mehner, 2005). While macrophyte cover generally supports greater diversity and abundance of invertebrates than open silty areas or those dominated by gravel and stones (Watkins et al., 1983), and removal of submerged vegetation generally reduces macroinvertebrate taxa richness (Rabe & Gibson, 1984; Tolonen et al., 2003), this is not always the case (Kuflikowski, 1974).

High standing biomass of invertebrate grazers among plant beds implicates them as major conduits of energy along trophic pathways, prompting suggestions that grazers on epiphytes are important symbionts for macrophytes by cleaning epiphytes from stems and leaves (Phillips et al., 1978; Underwood, 1991; Daldorph and Thomas, 1995). Changes in standing biomass, of filtering molluscs can also relect and effect shifts in lake trophic state (Lewandowski, 1991; Dobrowolski, 1994; Krzyzanek and Kaska, 1995; Dusoge et al., 1999). Patterns of abundance and prevalence of oligochaetes and chironomids are often very notable within plant beds (Soszka, 1975; Gerrish and Bristow, 1979; Kornijow, 1989a,b; Van den Berg (1997). Overall, littoral macroinvertebrate communities may have an important role in sequestration and recycling of minerals (Kolodziejczyk, 1984a,b; Underwood, 1991; Schindler and Scheuerell, 2002).

There is, however, increasing evidence that spatial pattern may be nested within lakes and sites (White and Irvine, 2003; Jones et al., 2006; R.Little, Irish EPA, unpublished data) and is scale dependent (Lassen, 1975; Tolonen, 2004; Kansanen et al., 1984; Hawkins et al., 2000; Schindler and Scheuerell, 2002; Stoffels et al., 2005). Stoffels et al. (2005) remark on the ‘surprising differences in structure and function that are often found between lakes with very similar features and physical settings’ (citing Webster et al., 1996; Kratz et al., 1997; Sorrano et al., 1999; Donahue et al., 2003). This is a thought-provoking comment with regard to identifying type-specific reference conditions under the WFD. Understanding of the effects of scale, from the site to the ecoregion, requires a great deal of further work. This is important for monitoring and management (Hawkins et al., 2000; Johnson et al., 2004). In general, the strength of the relationship between landscape and site-specific biota is poorly understood (Hawkins et al. 2000).

Attempts to classify lakes based in invertebrates and across nutrient gradients has revealed the difficulty of disentangling highly multivariate data (Kansanen et al., 1984; Brodersen et al., 1998; Hämäläinen et al., 2003) . Work on individual taxa groups has shown trends with nutrient enrichment but these are often associated with high variance and, hence, low predictive power, which limits their use in relating a pressure to impact (Håkanson, 2001). Specific indicator species and/or taxa ratios may not be universally reliable because of e.g. the nature of sediments, trophic gradient, extent of eutrophication, physical structure and biogeography (Dermott, 1987; Godfrey 1978; Wisniewski and Dusoge, 1983; Dobrowoloski, 1987).

Examples of taxa groups that have been reported to provide a response to nutrient state are summarised in Table 3. It was, however, not possible from a review of the cited literature to provide clear links with particular nutrient concentrations. While rhetoric relating to trophic state is common, quoted nutrient concentrations are not. Even if a value relating to a nutrient concentration is stated, this may refer to a single or low number of samples. Table 3, therefore, relates taxa to general nutrient state, denoted by the terms oligotrophic, mesotrophic and eutrophic. It is also uncertain if these terms relate exactly, or closely, to the boundaries used in lake classification schemes following OECD (1982). Overall, quoted associations need to be treated with caution, and it is notable that some taxa span the trophic scale. In preparation for the WFD, a consortium of Dutch scientists has suggested expected dominance of taxa that represent impact and a scoring system for parameters of community structure (van der Molen, 2004). Positive and negative indicator taxa proposed are shown in Table 4. However, it has to be kept in mind that the composition of littoral benthic invertebrates is heavily influenced by other factors other than trophic state, as habitat availability, which depends on ecoregion, lake type, and human shoreline alterations (Brauns et al. subm. b). Concerning the applicability of the information given in the Tables 3 and 4, it thus has to be noted that

- the validity of the associations has still to be checked for other lake types than studied by the cited authors,
- the presence or absence of species marked in the tables cannot be interpreted absolutely, but refer to high or low abundances in a specific littoral habitat type, as the populations of many of the listed species depend on the availability of specific food resources,
- for use of such information for lake assessment, the faunistic data have to be related to the ecoregion, lake type and mesohabitat where they were recorded.

A general understanding of the ecological mechanisms by which suggested indicator taxa respond to pressures is frequently lacking

(Savage, 1982; Hawkins and Vinson, 2000; Irvine et al. 2001). Where there is uncertainty about the mechanism that drives biological change, the justification for including those elements in classifying lakes, and particularly their use as ecological indicators, may be weak. Additionally, use of indicator taxa is, inevitably, dependent on taxa occurrence. Sensitive taxa may be rare and detection can vary with sampling effort. Indeed, it is striking that most taxa found in the lake littoral are rare (Brodersen et al., 1998; Irvine et al., 2001; Nijboer and Verdonschot, 2004) and low impacted sites tend to have greater taxa richness and more rare species than impacted ones (Doberstein et al., 2000; Fairchild et al., 2000; Chase and Liebold, 2002). For this reason a number of authors (Lyons et al., 1995; Cao et al., 1998) have argued that rare species are critically important indicators of ecosystem health. It may also be that the search for simple relationships between single taxa or assemblages may fail because it will not encompass a sufficient range of conditions and scales. Nijboer et al. (2005) concluded that single or sub-sets of taxa provided high classification error, and recommended using all taxa for community characterisation, especially where habitat diversity was high.

The use of taxa scores that relate to trophic state, rather than taxa descriptions per se overcomes the influence of occurrence of particular taxa in lake assessment but is dependent on accumulation of large and reliable data sets in order to provide reliable scores in the first place. The application of community metrics developed in rivers may, however, be of limited value for the assessment of lakes. This is not surprising, as these metrics were developed largely from the response of invertebrates inhabiting riffle zones of rivers to a depletion in oxygen resulting from organic pollutants, although there remains hopeful investigation of the usefulness of metrics from e.g the AQEM (see Donohue et al., 2006) for application in lakes. There is no general agreement on which metrics or taxa group(s) offer(s) the best option for use in cost-effective monitoring. Current work using metrics derived from multivariate analysis, based on Canonical Correspondence Analysis (CCA; ter Braak, 1986, 1990; ter Braak and Verdonshot, 1995) may provide a more reliable basis for classification (Dodkins et al. 2005). However, data sets amenable to either multimetric or multivariate analysis may be limited by few data across lakes types and nutrient gradients (e.g. Timm et al. 1999; White, 2001; Brodersen et al., 1998; Kashian and Burton, 2000). Multivariate approaches provided by Artificial Intelligence models, so far developed largely for freshwaters from use of datasets collected from U.K. rivers (Walley & Fontama, 2000) may produce alternative valuable diagnostic and prognostic tools for lake assessment. Under conditions of uncertainty, methods of 'inexact' or plausible reasoning, such as Bayesian inference provide a powerful tool that enable a) the ability to reason bidirectionally (i.e. from cause to effect and from effect to cause as required); b) the ability to modify the dependencies between variables whenever new evidence is introduced; and c) the ability to change one’s mind when new evidence ‘explains away’ earlier evidence. Further discussion on the development of Bayesian modelling for predicting ecological communities is found in Ter Braak et al. (2003).

Because a high amount of unexplained variance in community assessment techniques may limit use of individual or taxonomically similar organisms for biomonitoring, analysis based on functional groups or trophic guilds offers an alternative approach (Merritt & Cummins 1996; Pinel-Alloul, 1996; Kornijów and Ścibior, 1999; Johnson et al. 2004; Menetry et al., 2005). Heino (2000), however, found slight differences in response of functional groups to environmental factors across 21 lakes in NE Finland. Distributions of body size may provide a simple, though largely untested, assessment of trophic state (Basset el al., 2004). However, previous work on streams have shown a very limited variation of size spectra of macroinvertebrate assemblages when tested against phosphorus gradients (Bourassa and Morin, 1995; Solimini et al., 2001). Another interesting scientific development is the use of species traits of macroinvertebrates in biomonitoring that was recently suggested for streams (Usseglio-Polatera et al., 2000). Species traits may provide a useful way of understanding the role of biological communities into ecosystem functioning and the response of assemblages to changes of environmental conditions. Using mulriple indicators of species traits may also increases the probability of detecting impacts. Unfortunately, no data are available on the species traits of lake macroinvertebrates.

In conclusion, while the Water Framewok Directive has spawned research on littoral macroinvertebrates in lakes (e.g. Irvine et al., 2001; Ruse, 2002; Tolonen, 2004) there remains a limited understanding to guide monitoring and classification. While water chemistry, sediment type and vegetation have major effects on macroinvertebrate community structure, the interactions between these are, however, varied, often complex and frequently mediated by trophic relationships. Littoral invertebrate distribution is affected by habitat structure, depth and season. Biotic effects of competition and predation may also be important. Further work is required on whether variation within lakes is greater than that among lakes within similar lake types, and on scalar effects in the interpretation of data. A number of studies have indicated the usefulness of individual taxa or taxa groups for lake classification. Multimetric and multivariate models have also been used. There is, however, no current consensus on which approach is the best or most cost-effective. The use of Artificial Intelligence, functional groups or body-size distributions offer alternative approaches. Finally, many assessment programmes may not be designed with sufficient regard to the confidence that can be placed in the results. While these problems can be overcome with careful thought to sampling strategy there is, nevertheless, the uncomfortable recognition that the theoretically required sampling effort (numbers of samples needed to provide reliable estimate of site condition) is restricted because of financial costs. Most monitoring, therefore, provides results that are "best estimates" of site condition that may not stand up that well to highly critical evaluation. Sampling programmes that are designed for well defined objectives, therefore, provide a number of inherent challenges. These include operational decisions on frequency and spatial distribution of sampling, whether to collect replicates and, if so, to pool them, and choices about equipment and its use. The classification process that has been driven by the WFD is empirical, but the difficulties that challenge cost-effective sampling and assessment of littoral invertebrates in lakes will only be solved through testing of ideas and extensive sampling. If littoral macroinvertebrates are to provide a meaningful contribution to lake assessment there is, a need for considerable increase in understanding the response of those communities to nutrient, and other, anthropogenic pressures.

Table 3 Association of littoral invertebrates with nutrient state. For notes on the applicability of this list see text. References: 1 Saether, 1979; 2, Savage, 1982; 3, Stańczkowska et al., 1983; 4 Kansanen et al., 1984 ; 5 Biesiadka and Szczepaniak 1987; 6 Kornijów, 1988 ; 7 Kuklińska, 1989; 8 Radwan et al. 1991; 9 Petridis, 1993; 10, Pinel-Alloul, 1996; 11 Brodersen et al., 1998 ; 12 Kangur et al., 1998; 13 Pieczyńska et al. 1999; 14 Irvine et al., 2001; 14 Edsall et. al., 2001.

Taxa	Oligotrophic	Mesotrophic	Eutrophic	Reference
Chironomidae				
Glyptotendipes gripekloveni	X	X		6
Monodiamesa bathyphila	X	X		1
Stictochironomus psammophilus	X			8
Tanytarsus ex gr. *mancus*	X			8
Paracladopelma camptolabis	X			8
Psectrocladius sp.	X	X		4, 6, 9, 11
Microtendipes sp.	X	X		4, 6, 9, 11
Pseudochironomus	X	X		4, 6, 9, 11
Corynoneura	X	X		4, 6, 9, 11
Chironomus sp.			X	4, 6, 9; 10, 11
Chironomus plumosus			X	12
Cricotopus sp.			X	4, 6, 9 11
Tribelos sp.		X	X	10
Oligochaeta				
Limnodrilus sp.			X	7
Limnodrilus hoffmeisteri				10
Potomothrix moldaviensis			X	7
Potamothrix hammoniensis			X	12
Tubifex tubifex blanchardi			X	7
Spirosperma ferox	X	X		10
Aulodrilus limnobius	X	X		10
Sytlodrilus heringianus	X	X		10
Corixidae				
Sigara falleni			X	5, 14
Sigara concinna			X	2
Siagara praeusta			X	5
Micronectinae	X			5

Sigara striata	X			5
Sigara concinna			X	5
Mollusca				
Theodoxus fluviatilis	X	X		3
Valvata cristata	X	X		3
Pisidium sp.	X	X		3
Musculium lacustrae	X	X		3
Marstoniopsis scholtzi	X	X		3
Amnicola limnosa	X	X		10
Probythinella lacutris	X	X		10
Valvata carinata	X	X		10
Bithynia tentaculata	X	X	X	10, 12
Gyraulus deflectus		X	X	10
Valvata tricarinata		X	X	10
Physa gyrina gyrina		X	X	10
Gyraulus albus,	X			13
Physa fontinalis	X			13
Planorbis carinatus	X			13
Lymnaea sp.			X	
Amphipoda				
Gammarus fasciatus		X	X	10
Ephemeroptera				
Hexagenia sp.	X			15
Plecoptera				
Siphonoperla torrentium			X	
Trichoptera				
Mystacides	X	X		10
Ceralcea	X	X		10
Necropsyche	X	X		10
Heliopsyche	X	X		10
Polycentropus	X	X		10

Table 4 Positive and negative dominant indicator taxa associated with quality of Dutch standing waters (van der Molen, 2004). For notes on the applicability of this list see text.

Negative associations (poor quality)	Positive associations (high quality)
Asellus aquaticus	*Arrenurus robustus*
Cricotopus gr sylvestris	*Caenis horaria*
Dero digitata	*Caenis luctuosa*
Dicrotendipes nervosus	*Cladotanytarsus*
Limnodrilus variegatus	*Cloeon dipteran*
Polypedilum nubeculosum	*Cloeon simile*
Procladius sp.	*Endochironomus albipennis*
Procladius choreus	*Gammarus pulex*
Procladius lugens	*Mesovelia furcata*
Procladius rufovittatus	*Micronecta minutissima*
Procladius sagittalis	*Micronecta scholtzi*
Psectrotanypus varius	*Microtendipes chloris agg.*
Radix ovata	*Piona nodata nodata*
Valvata piscinalis	*Pisidium* sp.
	Pseudochironomus prasinatus
	Stylaria lacustris
	Tanytarsus sp.

2.2 Eutrophication and the profundal assemblage

Enhanced supply of nutrients to lakes often results in increased littoral and pelagic productivity with subsequent increase in the organic matter input to sediments. The increase of respiration of microorganisms associated with the input of organic matter, can results in oxygen depletion in the hypolimnion of lakes. This indirect effect of eutrophication on oxygen condition has a direct effect on the bottom fauna. Consequently the assemblage of organisms living in the profundal zone can provide an indication of past and current disturbances and may be used in the assessment of the ecological conditions of a given lake (Brinkhurst 1974; Rosenberg & Resh 1993).

Different macroinvertebrate species prevail depending on lake trophic status, which affects food quality and quantity, and oxygen status. Food is the main factor changing community composition when environmental conditions are not too severe. However, when organic pollution is more intense, it is oxygen concentration rather than food that limits the species survival and determines the community composition. Biological communities are highly influenced by site-specific conditions, and often these effects can be well synthesized by the seasonal dynamics and the depth distribution of benthic organisms (Bazzanti and Seminara 1987a,b). Therefore, even though some generalisations are possible, each lake has its own history that must be understood before benthic macroinvertebrates can be used for biological assessment. The lake type is an important factor in determining invertebrate species composition. For example, different response of benthic species to interactions between maximum depth, conductivity, dissolved oxygen and nutrient concentration were observed in different lake typologies (Rossaro et al., 2006).

Lake biomonitoring using species assemblages of benthic macroinvertebrates dates back to the beginning of the 20th century (Cairns and Pratt, 1993); Naumann (1921), Lenz (1925), Lundbeck (1936), Thienemann (1954) and Brundin (1956) observed a distribution of different chironomid species according to trophic condition, oxygen saturation and depth in lakes. Brundin (1974) revised the state of knowledge about the indicator value of chironomids, discussing bio-geographical problems. Benthic macroinvertebrates have been extensively used in lake classification (Wiederholm, 1981; Kansanen et al., 1984; Aagaard, 1986) and are acknowledged indicators of lake quality (Johnson et al., 1993; Bazzanti et al. 1995, 1998). Profundal benthic macroinvertebrates can reveal short and long-term changes in ecological quality of lakes involving both worsening or recovery action towards the original conditions (Bazzanti & Seminara 1987b; Lang & Lods-Crozet 1997). Geographically, the scientific knowledge on profundal lake invertebrates is more developed in the Nordic Countries and UK than in Southern Europe. Few data from Mediterranean countries are available and limited to Italy (Rossaro et al., 2006 and references therein), France (Verneaux et al., 2004; but many of France lakes are not strictly in the Mediterranean), Spanish reservoirs (Real et al., 2000), and Turkey (Arlsan & Šaìn, 2006).

In the profundal zone, oligochaetes and chironomids are considered the most useful indicators of oxygen condition (Brundin, 1949) and trophic status (Sæther, 1979). Both oxygen levels and sediment granulometry have been related to oligochetes species distribution (Verdonschot, 1996). The drawback of Oligochaeta based indexes is that only a component of macrobenthos is considered and the identification of species of oligochaetes is necessary, which requires mature specimens with developed genitalia. Chironomids are probably the most useful profundal indicator group of trophic status. They have high species richness compared to other benthic invertebrate groups, they occur over the whole spectrum of nutrient conditions and individual species have specific environmental tolerances. Therefore species composition changes with changing lake trophic status. Contributions attempting to relate environmental factors with Chironomids species composition are common in litterature (Ruse 2002a, 2002b) and include paleolimnological studies (Little & Smol, 2001). The response of the genus Chironomus to environmental factors was analysed in Spanish reservoirs (Real et al., 2000). Different species could be separated according to their response to oxygen concentrations. High depth, water temperature and sulphide content were inversely correlated with Chironomus density, alkalinity and particulate nitrogen were positively correlated. Spatial and temporal factors usually account for the largest source of variation of the chironomid assemblage (Franquet et al., 1995; Ruse & Davison, 2000). Unfortunately, the environmental variables are often correlated to each other, so it is difficult to separate the influence of single factors in determining the assemblage composition like water temperature, oxygen and nutrients (Larocque et al., 2001). Moreover, the large number of species and the problems arising when dealing with their taxonomic identification left many unresolved questions.

Following the strict limitation of phosphorus inputs, many European lakes are recovering from cultural eutrophication (Sas, 1989). The recolonization of profundal sediments by the benthic species which prevailed before the onset of eutrophication can be used to monitor the progress of recovery (Lang, 1990). For this purpose, oligochate and chironomid species are complementary indicators because of differences in their autoecological traits. Chironomids larvae are more mobile and can migrate at different depths, being less dependent on the quality of the sediment than the oligochaetes. Moreover, chironomid larvae feed on “fresh” detritus deposited on the surface of sediments, whereas oligochaetes feed on bacteria associated with organic matter in an advanced state of decomposition. Therefore, chironomids are preadapted to react more rapidly than oligochaetes to changes in environmental conditions, it is expected that they react more quickly to the improvement of water quality in lakes (Dinsmore and Prepas, 1997; Lang and Lods-Crozet, 1997).

The annex V of the WFD specifically outlines benthic invertebrate fauna composition and abundance, the ratio of sensitive taxa to insensitive taxa and the diversity of invertebrate communities as criteria that need to be defined for type-specific ecological assessment of lakes. Several indices and classification systems have been developed using chironomid and oligochaete assemblages. Those indices, most of which were developed for lakes in Northern Europe, rely on relative abundances of chironomid species, the ratio of tolerant to intolerant tubificid oligochaetes, or the ratio of oligochaetes to chironomids (Wiederholm, 1980). Two other frequently used measures in assessment studies are the Shannon and the Margalef diversity index. For example, Wiederholm (1980) showed a good correlation between taxa richness (the total number of taxa, adjusted per sampling depth) and chlorophyll-a (e.g. trophic state). In general, low values of diversity are associated with more eutrophic lakes.

Wiederholm (1980) developed a benthic quality index (BQI) using chironomids alone, proposing six different scores (Table 5). However, this index cannot be applied in Southern Europe because the proposed species are rarely recorded in the Mediterranean. A more sophisticated index can be developed by accounting for differential response of single species using multivariate statistics approach and considering the whole benthic assemblage. This approach was developed recently for Italian lakes (Rossaro et al., 2006) where different invertebrate taxa could be related to the environmetal conditions, indicative of lake trophic state. The advantage of this approach is that it considers (e.g. gives “weights”) all the taxa collected in a given lake, without the need to select few species (as in chironomidae or oligichaete based indexes).

Also Sæther (1979) developed a lake trophic classification identifying 15 lake types using profundal chironomid assemblages from Nearctic and Palaearctic lakes. Community structure was examined against the ratio between phosphorus concentration and depth. Unfortunately, the 15 chironomid assemblages proposed by Sæther (1979) also include many species never recorded in Southern Europe. Lang (1985) proposed a list of oligochaeta species identified as indicator species (Table 6), giving a low score to species indicative of oligotrophic conditions and a high score to species preferring eutrophic conditions. Lang also developed three indices of trophy for lake Geneva based on the structure of the Tubificidae and Lumbriculidae communities (Lang, 1998).

More recently Irvine and coworkers put together a list of macroinvertebrate taxa indicative of trophic state based on an ectensive literature review (Irvine et al., 2006). The European project Rebecca (www.rbm-toolbox.net) also tried to identify potential indicators of lakes minimally impacted in different European regions using expert judgement (see Lyche Solhaim et al., 2005). The validity of those taxa as indicator of lake trophic staus was then validated using a large dataset by Donohue et al., (2006; Table 7).

Table 5 Species of chironomidae and relative scores (Wiederholm 1980).

Species	Score
Heterotrissocladius subpilosus	5
Micropsectra spp.	4
Paracladopelma nigritula	4
Sergentia coracina	3
Stictochironomus rosenschoeldi	3
Chironomus anthracinus	2
Chironomus plumosus	1

Table 6 Oligochaetes as indicator species. O =oligotrophic; M =mesotrophic; E =eutrophic (from Lang, 1985).

Species	Lake trophy
Stylodrilus lemani	O
Stylodrilus heringianus	O
Peloscolex velutinus	O
Potamothrix vejdovskyi	M
Peloscolex ferox	M
Psammoryctides barbatus	M
Aulodrilus pluriseta	M
Limnodrilus profundicola	E
Limnodrilus hoffmeisteri	E
Limnodrilus sp.	E
Potamothrix bedoti	E
Potamothrix heuscheri	E
Potamothrix hammoniensis	E
Tubifex tubifex	E

Table 7 Profundal macroinvertebrates displaying consistent nutrient association in literature review and validated using the Rebecca data set for lakes in different trophic categories. TP: Total phosporus; chl a: chlorophyll a; oligotrophic (<10 μg/L TP and/or <2.5 μg/L Chl a), oligo to mesotrophic, mesotrophic (10-35 μg/L TP and/ or 2.5-8 μg/L Chl) and eutrophic (35-100 μg/L TP and/or 8-25 μg/L Chl). From Donohue et al. (2006).

Trophic category	Taxon	TP	Chl a
Oligotrophic	*Heterotanytarsus apicalis*	X	-
	Paracladopelma nigritula	X	-
	Stylodrilus heringianus	X	-
	Heterotrissocladius marcidus	X	-
Oligotrophic to Mesotrophic	*Pisidium* sp.	X	X
Mesotrophic	*Sergentia coracina*	X	X
	Aulodrilus pluriseta	X	X
	Tanytarsus sp.	X	X
	Spirosperma ferox	-	X
	Stictochironomus rosenschoeldi	-	X
	Monodiamesa bathyphila	-	X
Eutrophic	*Chironomus plumosus*	-	X

2.3 Hydromorphological alteration and the littoral/sublittoral assemblages

While the discharge of wastewater to lakes has been reduced extensively in many countries (Kraemer et al. 2001, Brönark & Hansson, 2002), hydromorphological alterations of lake shores represent a relatively recent anthropogenic pressure to riparian zones (Engel & Pederson, 1998). Climatic change will likely increase unintended or intended fluctuations of lake water levels, in particular in subarid regions of Europe (Brauns et al, subm. c). Further, the intensity of shoreline development is expected to increase in the future (Walz et al, 2002; Schmieder 2004). Both hydrological and morphological alterations clearly affect littoral benthic invertebrate assemblages (Table 8). In addition, most alterations also affect the sublittoral, although less severely. A recent study (Rowen et al., 2006) has documented a methodology for Lake Habitat Assessment (LHS) that has been designed to support the WFD. The relationship between LHS quality scoprews for habitat and ecological metrics related to biological elements would form an important development of this work.

Anthropogenic water level fluctuations

Most lakes are subjected to seasonal fluctuations of water level following seasonal patterns of stream, rain and groundwater discharge to the lake. The extent of these fluctuations depend largely on lake type (groundwater fed or connected to stream system) and the regional climatic pattern. Lakes in subalpine, subarid and arid regions exhibit the most pronounced water level fluctuations. Natural seasonal patterns of water level fluctuations have, however, often been altered anthropogenically. Alterations either aimed at a seasonal stabilisation of water levels, as in navigable lakes in subalpine regions, or resulting in an amplification of water level fluctuations, as is the case in lakes used for drinking water or as hydropower reservoirs. Either way, this leads to a specific seasonal alteration of shoreline structure. As substrate particle size often decreases with increasing distance from the high water shoreline, artificial changes in seasonal water levels may cause a potential mismatch between habitat availability and life histories of species. Moreover, as submerged and emergent macrophytes are known to respond sensitively to water level fluctuations, the availability of complex invertebrate habitats provided by plants is probably controlled strongly by the actual regime of water level fluctuations.

Most existing knowledge on the ecological effects of human-altered hydrological regimes comes from studies on reservoirs or regulated lakes, where generally seasonal water level amplitudes of up to 30 m occur (e.g. Smith et al., 1987). Water level fluctuations were, in general, demonstrated to affect the shore zone of reservoirs directly by desiccation (Hynes, 1961) and bottom freezing (Palomaki & Koskenniemi, 1993). Water level fluctuations were also shown to reduce the diversity, or alter the composition of, littoral habitats (Baxter, 1977; Hellsten et al., 1996; Hill & Keddy, 1992), and affect the littoral food chain through the loss of macrophytes as a food resource (Hill et al., 1998; Wilcox & Meeker, 1991, 1992). Benthic invertebrates are the biotic component of lake shores that are affected most severely by these alterations, since their low mobility restricts their ability to follow the receding water. Thus, in reservoirs and regulated lakes, invertebrate richness and abundance was demonstrated to be lowest in the eulittoral zone (Smith et al., 1987) and highest within the sublittoral zone below the drawdown limit (Koskenniemi, 1994; Palomaki, 1994). Detrimental impacts on littoral macroinvertebrate community structure have also been quantified (Giziński & Wolnomiejski 1982; Jurkiewicz-Karnkowska 1989). Some macroinvertebrates have, however, been shown to be able to move with moderate rates of water level alteration to the order of 0.5 cm hour-1 (Winter 1964). Some macroinvertebrate taxa have been shown to recolonise habitats within weeks of rewetting, while others may take over 3 months (James et al. 2002).

The effects of anthropogenically increased water level fluctuations on natural lakes have been studied in a number of north German lowland lakes (Brauns et al. subm. c). Here, roots of riparian trees form an important eulittoral habitat exhibiting the highest invertebrate diversity among eulittoral habitat types. These root habitats would become inaccessible for benthic invertebrates with receding lake water level. It was demonstrated that this would affect primarily Coleoptera and Odonata, which rely upon the 3-dimensional structure of root habitats most likely as a refuge against predation by fish. It was shown, however, that unimpacted and dense reed stands in the infralittoral zone can substitute for the loss of riparian root habitats (Fig. 1).

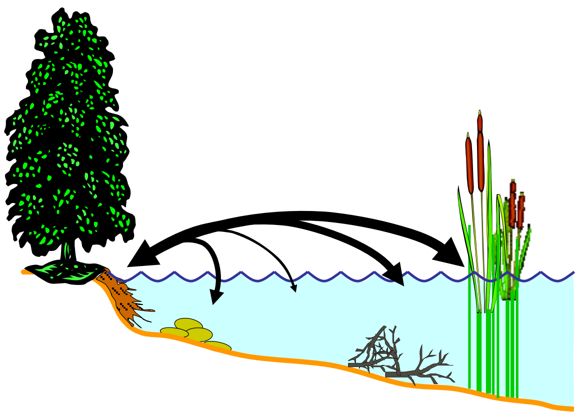

Figure 1 Primary habitats in the littoral zone of a lowland lake (from left to right: roots, stones, sand, coarse woody debris, reeds). The width of the arrows indicates the degree of similarity between the invertebrate community in the root habitat with the communities of the other habitats (From Brauns et al., subm. c).

Anthropogenically increased wave action

Wind-induced waves may affect the composition and biomass of littoral macroinvertebrate assemblages both in the short- and long-term. Over extended time periods, wind can determine littoral habitat characteristics, through affecting sediment particle size distribution (Brodersen 1995; Rasmussen & Rowan 1997; James et al. 1998; Tolonen et al. 2001; Weatherhead & James 2001). In the short term, extreme wind exposure has been shown to reduce total abundance and species richness of macroinvertebrate communities on wave exposed shores (Barton & Carter 1981; Mac Isaac 1996).

In contrast, ship-induced waves may substantially exceed the impacts of natural waves because of their increased height (up to 60 cm; Bhowmick et al. 1991), higher frequency, and stronger increase of flow velocity (up to 80 cm s^{-1}; Arlinghaus et al. 2002). Further, ship-induced waves may affect shore zones that are not usually exposed to waves. Clear impacts of ship-induced waves on the distribution, fitness and survival of fish larvae , eggs and juveniles have been documented (e.g. Morgan et al. 1976; Holland 1986; Arlinghaus et al. 2002; Wolter & Arlinghaus 2003; Wolter et al. 2004), while impacts on macroinvertebrate assemblages have also been found (Bishop 2003, 2004; Bishop & Chapman 2004). In navigable rivers, quantification of hydraulic forces caused by the passage of ships on littoral areas led to the conclusion that ship-induced waves constitute a major hydrodynamic stress for macroinvertebrates communities (Brunke et al. 2002; Garcia et al. 2005). To resist against wave-induced disturbances, invertebrates mainly depend on species-specific morphological and behavioural adaptations to high flow conditions, and on the availability of refuges in their immediate environment. In particular, high habitat complexity may be a key factor in reducing wave-induced disturbance on benthic invertebrates (cf. Borchardt 1993). It has been demonstrated recently that the various habitat types occurring in the littoral zone of lowland lakes differ in their efficiency to provide refuges against wave disturbance for benthic invertebrates (Gabel et al. submitted). The extent to which different habitat types acted as a refuge against wave disturbance was dependent primarily on their structural complexity, which in turn was related to the dissipation of the kinetic energy of the waves. Hence, adverse effects of ship and boat traffic on littoral invertebrate assemblages are increased substantially if complex littoral habitats like tree roots or dense reed belts are absent (Fig. 2).

Morphological shoreline alteration

Shores are often protected by rip-rap (stone surfacing), usually to avoid bank erosion due to intense navigation, or even by retaining walls (vertical walls) in urbanized areas. In general, shoreline development is considered to have detrimental impacts on the littoral zone through the alteration or loss of littoral habitats. Shoreline developments have, for example, been shown to cause a reduction of both submerged and emerged macrophyte stands (Radomski & Goeman 2001; Elias & Meyer 2003), and to alter sediment particle size composition (Jennings et al. 2003). Concomitantly, littoral fish communities, in particular their spatial aggregation (Scheuerell & Schindler 2004), species richness (Jennings et al. 1999), and production (Schindler, Geib & Williams 2000; Radomski & Goeman 2001), were affected adversely. As shoreline development is often accompanied by clear-cutting of the adjacent riparian vegetation, the amount of coarse woody debris (CWD) supplied to the littoral zone can be reduced substantially (Christensen et al. 1996).

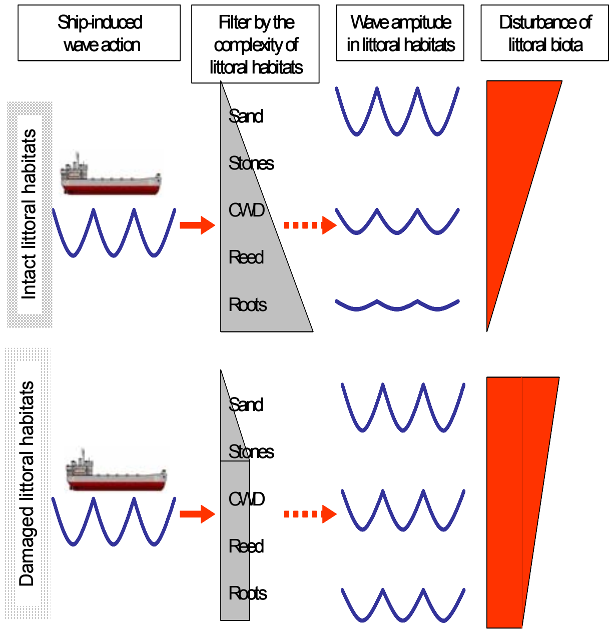

Figure 2 Conceptual diagram on factors influencing the disturbance of littoral biota by navigation. Ship-induced wave disturbance is mitigated in complex habitat types, but which may be damaged by intense navigation on the long-term.

Since CWD constitutes an important habitat type for littoral fish (Newbrey 2002; Lewin, Okun & Mehner 2004; Barwick 2004), the impacts of shoreline development and riparian clear-cutting can be doubly severe. Although few studies have been done, impacts of shoreline development on macroinvertebrate communities have also been found. Bänzinger (1995) compared the macroinvertebrate communities of three different types of erosion control structures with five types of natural shorelines in Lake Geneva (Switzerland), and demonstrated species diversity and abundance to be lowest at modified shorelines. In addition, work done by Watson (2005) on a number of Irish lakes also found a negative relationship between lake shoreline habitat modification and the richness of macroinvertebrate taxa. Recreational use of lake shores has – to our knowledge – never been examined regarding its effects on littoral macroinvertebrates.

As benthic invertebrates are much less mobile than fish, and exhibit a much higher dependence on littoral habitat types, shoreline developments would be expected to have considerably more severe impacts on invertebrate communities. A recent study of habitat-specific benthic invertebrate assemblages in German lowland lakes exhibiting several types of shoreline modification (Brauns et al. subm. a) found lowest species richness, total abundance, and abundances of predators, shredders and xylophagous species on recreational beaches and retaining walls in the eulittoral zone. Macroinvertebrate community composition differed significantly between beaches, retaining walls and natural shorelines. Community structure in rip-raps did not, however, differ significantly from natural shorelines. In the infralittoral zone, neither rip-raps nor retaining walls differed significantly from natural shorelines in any of the examined community parameters. Conversely, recreational beaches showed a significantly lower species richness, total abundance, and abundances of piercers, predators and scrapers. This work suggests that shore protection by rip-raps had only minor effects on the macroinvertebrate communities in the eulittoral and infralittoral zones of lowland lakes. In contrast,, retaining walls caused a substantial reduction in the complexity of habitat structures in the eulittoral zone which was accompanied by an impoverishment of the invertebrate community, but these ecological effects did not extend to the infralittoral zone. Recreational beaches affected habitat quality within the entire upper littoral zone most profoundly.

It can be derived from that study that management efforts to preserve or restore lake shores should focus on the structural complexity of the littoral zone. In the studied lowland lakes, structural complexity is especially provided by roots of riparian trees, woody debris, and reed stands.

Conclusion

It has been shown that there are a number of hydromorphological alterations that may impair the ecological status of lakes. The extent to which lake ecosystems are affected should be assessed preferentially using benthic macroinvertebrates, as these are much less mobile than fish. There is some indication that distinct species may be identified that react most sensitively to specific pressures, and thus may serve as specific indicator species. However, empirical data on the relationship between lake hydromorphology and lake zoobenthos exist only for selected ecoregions and lake types so far, and some pressure types have probably never been assessed. Hence, the estimation of the ecological effects of hydromorphological alterations on European lakes would need a field survey on the most prominent impacts of each ecoregion, and a database on the biological-ecological species traits of lake zoobenthos. Such knowledge would enable a holistic assessment of the ecological status of lakes, and to identify effective restoration measures for lake shores..

Table 8 Hydromorphological alterations of lake shores and associated ecological effects on littoral benthic invertebrate assemblages.

Human activities	Hydromorphological alterations	Ecological effects
Modification of water level dynamics	Alteration of the seasonally available shorelines and including substrate types Increased sedimentation through degradation of tributary streams	Potential mismatch between habitat availability and life histories of species
Navigation / boating	Increased wave action even in sheltered shorelines Resuspension of fine sediments Abrasion of submerged macrophytes	Increased frequency of hydraulic disturbance Habitat degradation Possibly favours the immigration of invasive species
Artificial shoreline stabilisation	Loss of many natural shoreline structures Increase of wave action by wave reflection	Habitat loss, e.g. tree roots Reduced aquatic-terrestrial connectivity Increased frequency of hydraulic disturbance
Shoreline deforestation	Lack of input of leaves (esp. in autumn) and of coarse woody debris (CWD)	Habitat loss Decrease of food supply for shredders
Recreation (bathing, angling)	Removal of emergent and submerged macrophytes Frequent disturbance by trampling	Habitat loss Decrease of food supply for shredders Increased frequency of hydraulic disturbance

2.4 Acidification and the littoral assemblage

Acidification of rivers and lakes is a major ecological problem in the Nordic countries Sweden, Norway and Finland. In 1995 the critical load for S were exceeded in 9% of the Finnish lakes (3000 lakes), 9% of the Swedish lakes (6000 lakes) and 27% of Norwegian lakes (10000 lakes) (Skjelvåle et al. 2001). In a study by Henriksen et al. (1998) 16.1% of the investigated lakes in Wales, 12.6% of the lakes in Russian Kola, 8.3% of the lakes in Norway, 3.9% of the lakes in Scotland and 3.6% of the lakes in Sweden had a pH below 5.0 (Table 9). Recent calculations from Sweden suggest that about 5% of the Swedish lake population is affected by acidification and another 3% would have been acidified if not limed for protection (J. Fölster pers comm.). In the early 1990s, it was estimated that some 14,000 or 15% of Swedish lakes with a surface area < 1 km^2 and about one-fifth of all watercourses could be regarded as being adversely affected by acidification (Bernes, 1991). The emission of S has been reduced by 50-60% since 1980 in Europe and thus this has lead to marked decrease in S deposition in the Nordic countries since that time (about 60%) (Kulmala et al. 1998). Sulphate is the major driving force in changes in lake water chemistry in the Nordic countries from 1990 – 1999 (Skjelvåle et al. 2001). In the study by Skjelvåle et al. (2001) 344 lakes included in the national monitoring programme in Finland, Norway, and Sweden were analysed for trends in water chemistry between the years 1990-1999. These authors report that Sulphate has decreased in 69% of the lakes, concentrations of base cations decreased in 26%, chloride concentrations decreased in 23% of the lakes, ANC increased in 32% of the lakes, pH increased in 23% of the lakes, whereas total organic carbon concentrations increased in 12% of the lakes. Natural recovery of water chemistry has been documented in a number of lake ecosystems in Sweden (Wilander, 1997) and across Europe (Stoddard et al., 1999), and recent studies in Norwegian lakes have attributed recovery of lake biology to decreased deposition of acidifying compounds (e.g. Halvorsen et al., 2003).

Acidification is still considered as a serious threat to the biodiversity and functioning of Swedish inland surface waters and also in Finland and Norway. For fish for example, about 470 fish populations have been lost in Finland, about 5100 populations in Norway, and in Sweden about 1200 populations of brown trout, 1100 of roach and 1100 of perch population are estimated to be lost (Rask et al., 2000). It also costs considerable amounts of money, the Swedish national liming programme e.g. has spent approximately 2 billion Swedish crowns (ca 216 million Euros) since the beginning of the 1990´s on liming surface waters (Bishop et al. 2001). Organism response to acidification can be complex, reflecting both the direct physiological effect of pH as well as the effects of associated metals and indirect effects mediated through bottom-up processes (e.g. food availability), and a number of studies have shown that macroinvertebrates, in particular mayflies, are affected directly by low pH and high concentrations of aluminium (e.g. Ormerod et al., 1987). Rosseland et al. (1990) found that aquatic organisms were affected by inorganic Al concentrations > 25 µg/L and these authors suggested this value as a lower threshold below which biological effects are negligible and a second concentration of 75 µg/L was suggested as an upper threshold where strong effects were predicted. Herrman et al. (1993) summarized the mechanistic effects of acid stress on benthic macroinvertebrates: i) H+ affects the osmoregulation negatively thus less energy is left for growth and reproduction, ii) Al3+ is more soluble at low pH which also affects the osmoregulation negatively, iii) some heavy metals known to affect invertebrates negatively such as Cd, Fe, Pb, Zn, and Cucan be more soluble at lower pH, iv) higher levels of Cd, Fe, and Pb decreases the escape behaviour and activity in general for e.g. L. marginata (active transport) but increases their drift behavior (passive transport), v) increased Fe concentration decreases feeding activity and search for e.g. L. marginata, deposits of Fe such as humic-oxides decrases nutrient uptake and also decrases the O2 uptake through the gills, vi) both molting and emergence of the insects is affected by lower pH, e.g. final metamorphosis to adult insects was only one third at pH 5 compared to pH 7, vii) the composition of functional feeding groups changes with changing food availability e.g. decreased numbers of scrapers while the number of shredders increased with increasing amount of course detritus, viii) th quality and quantity of microalgae changes e.g. green algae increases at lower pH, ix) the species numbers and abundances decreases generally of benthic macroinvertebrates with decreasing pH.

Lowered pH and/or increased metal concentrations of stream water are two of the most important factor associated with changes in benthic macroinvertebrate community structure of freshwaters (e.g., Townsend et al., 1983; Raddum & Fjellheim, 1984; Herrmann et al., 1993 Larsen et al., 1996). Hildrew et al. (1984) found that the pool of occurring species was limited at more acidic sites compared to neutral ones, and that the available food resources was lower in acid streams than in neutral ones. The rather straightforward relationship between acid conditions and the presence/absence of certain benthic macroinvertebratespecies have therefore been used to assess the effects of acid stress on stream ecosystems (e.g., Henrikson & Medin, 1986; Raddum et al., 1988; Bækken & Aanes, 1990; Degerman et al., 1994). Generally biological assessment systems for acid stress effects have been developed for running waters rather than lakes. In Sweden and Norway, a number of indices have been developed to assess the effects of acid stress on running water ecosystems. Six such indices were evaluated in a paper by Sandin, Dahl & Johnson (2004). These are referred to as the Norwegian acidity index I (N I) (Raddum et al., 1988), Norwegian acidity index II (N II) (Bækken & Aanes, 1995), Norwegian acidity index III (N III) (Bækken & Kjellberg, 1999), Swedish acid index I (S I) (Henrikson & Medin, 1986), Swedish acid index II (S II) (Degerman et al., 1994), and the Swedish acid index III (S III) (Lingdell & Engblom, 2002). The two most commonly used indices to evaluate acid stress in Swedish streams are S I (used mainly in southern Sweden) and S II (used mainly in northern Sweden), whereas the third index (S III) has recently been proposed to evaluate the mitigating effects of liming on Swedish streams. All of the indices referred to above, with the exception of the S I, are based on an extensive taxa list, where each taxon is classified according to its tolerance or sensitivity to acid stress. The stream is then classified as being of the same quality as the most acid sensitive taxon found at the site, even if only a single specimen of this taxon is found. In contrast, S I is based on five criteria (presence of taxa with different acid sensitivity, presence of *Gammarus* spp., presence of certain acid sensitive macroinvertebrate groups, the ratio of *Baetis* spp. and Plecoptera individuals, and total number of taxa). This index is also part of the Swedish Ecological Quality Criteria (EQC) for running waters (Swedish Environmental Protection Agency, 2000). The main conclusion from the Sandin, Dahl & Johnson, 2004 paper was that changes in acid index S I which is a kind of multimetric index, but does not include a normalisation of the parameters is seemingly more conservative and responding slower when indicating an improvement from stress than the other indices. This attribute is of special importance when evaluating acid stress, since sampling in the spring (when the worst acid conditions generally occurs) is difficult for logistic reasons. Thus an autumn sample using index S1 can still detect the effects of acid stress from an episode acidification effect in the spring. On the other hand, when a real improvement occurs, it might be more difficult to detect this change using a more slow changing index compared to the more variable, but thus also more sensitive indices (as indices S II, S III and the Norwegian indices). If the ecological quality of a stream is evaluated using the more variable indices, then the precautionary principle should be adopted and only data from the season where the organisms are exposed to the highest level of stress (i.e., spring) should be used. Even tough the index value changes from spring to autumn does not necessarily mean that the community structure recovers, since these indices are based on the presence of a single sensitive individual. Recently Johnson & Goedkoop (2006) have suggested a multimetric index to assess acid stress in Swedish lakes; the MILA index [Macroinvertebrate Index for Lake Acidity] which contains six parts; i) % ephemeroptera abundance, ii) % diptera abundance, iii) no of gastropoda taxa, iv) no of ephemeroptera taxa, v) the UK AWIC index [Dawy-Bowker et al. 2005], and vi) % abundance of predators. This is to my knowledge the only such system developed for littoral benthic macroinvertebrates in lakes.

In a study by Johnson et al. (submitted) some 126 Swedish lakes that are monitored annually for surface water chemistry and biology (e.g. benthic macroinvertebrates) as part of the national lake monitoring program was evaluated for the relationship between acidification variables and water chemistry. In this study the strongest relationship was found between water chemistry variables indicative of acidity and the benthic invertebrates, a second gradient was related to productivity of the system, where e.g. chlorophyll a and total phosphorus were positively, while altitude and latitude (x coordinates) were negatively associated with the gradient. There is thus a strong relationship between lake benthic macroinvertebrate community structure and acidification in Scandinavia. There is thus great potential for developing further the assessment systems based on the ecological quality of lakes for the Water Framework Directive purposes with thess stressors. Further, future climate change can also influence acidification trends and status in lakes. Recovery of biological components of lake ecosystems will lag behind the recovery of water chemistry in lake ecosystems. Climatic extremes can thus result in extremely low ANC values, which will affect ongoing biological recovery negatively. Continued monitoring of acid deposition, water chemistry, and biological elements is therefore of high importance to follow to what extent the biological recovery has indeed followed from the large-scale decrease in acid deposition in Europe over the last twenty years (Skjelvåle et al. 2001).

Table 9 Data on lake acidification from Henriksen et al. (1998).

Country	No of lakes investigated	Total no of lakes	% acid lakes (pH <5)
Finland:	873	29515	0.9
Norway:	1006	38845	8.3
Sweden:	3075	60264	3.6
Denmark:	19	709	-
Russian Kola	460	20320	12.6
Russian Karelia:	29		-
Scotland:	136	5054	3.9
Wales:	52	255	16.1

3 Current status on the use of benthic macroinvertebrates for ecological assessment of lakes at EU level

Water Framework Directive (WFD) requirements

The term 'ecological status' is defined in the WFD as: "…an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters, classified in accordance with Annex V" (Article 2.21). This implies that classification systems should reflect changes taking place in the structure of the biological communities and in the overall ecosystem functioning as response to anthropogenic pressures (e.g. nutrient loading, acidification, toxics).

To ensure the completeness of Member States assessment and classification systems, the Directive includes instructions on the elements to be measured, their assessment and how to integrate the information from the individual elements in a final classification score (ECOSTAT Working Group 2A, 2003). A 5 class classification of ecological status need to be derived for each biological element based on the Ecological Quality Ratio (EQR), defined as the ratio between type specific reference conditions and observed values of the relevant biological quality elements. The principle of one out all out is applied to obtain the final classification score.

One of the biological elements which need to be measured in assessing ecological status of lakes is benthic macroinvertebrates, for which the Directive requires the collection of data informing on the communities taxonomic composition, abundance, diversity and sensitive taxa.

Several are still the gaps to a complete application of the WFD requirement in relation to ecological status classification, in particular, the identification of reference conditions and development of indicators for lake benthic macroinvertebrates as evidenced by the WFD Common Implementation Strategy (CIS) Harmonisation Activity, the Intercalibration exercise and FP6 research projects.

WFD-CIS Harmonisation activity and the intercalibration exercise

Biological lake research and the use of biota in water quality classification have long traditions in Europe (e.g. the saprobic system by Kolkwitz and Marson, 1908; the trophic paradigm by Naumann, 1919). Also, freshwater biomonitoring using species assemblages of benthic macroinvertebrates goes back to the beginning of the 20th century, as benthic macroinvertebrates were extensively used in the lake categorization and as lake quality indicators (e.g. Naumann, 1921; Lenz, 1925; Lundbeck 1936; Thienemann, 1953; Brundin,1956; Wiederholm, 1981; Kansanen et al., 1984; Aaagard, 1986; Johnson et al., 1993).

The WFD requires (Annex V 1.3.6) that standards methods are used for monitoring of water quality elements. Thus, during 2004 and 2005 an activity on Harmonisation of biological methods within the WFD-CIS ECOSTAT[2] Working Group was started and gathered information on the Member States biological monitoring systems revealing that for all biological elements but phytoplankton there was rather little data collected by national lake monitoring programs.

Thus, benthic macroinvertebrates have not been until the present date consistently included in national monitoring systems for lakes, and there is a wide geographic discrepancy in data availability and use of macroinvertebrates in lake classification. The number of monitored lakes for which benthic invertebrates are also sampled is below 40% of total monitored lakes in most member states, with exception of Austria and Germany (Cardoso et al., 2005). These are particularly deficient in the Alpine and Mediterranean Geographic Intercalibration Groups (GIGs, for an overview of the GIGs see Van de Bund et al., 2004) countries.

There are several sampling designs adopted by the Member States differing in sampling frequency, devises used to collect samples and metrics calculated. As for example, the collection of samples is performed by several different devices: Ekman grab, sediment corer, triangle bottom dredge and hand net; the mesh size varies widely from 100µm to 670µm; kick sampling is performed with different duration (1-3 minutes), and habitat sampled are different (littoral in general or stones only); some countries sampled in lake littoral, other in the profundal and some in both lake zones. Probably these different approaches result from different information needs, e.g. assessing change as resulting from impact of different pressure either acidification or eutrophication but also results from the current lack of common/ harmonized sampling procedures as also required by the WFD. Sampling frequency is also variable between Member States, the most common frequencies

[2] Common implementation strategy: working group on the ecological status of surface waters

being 1 or 2 times per year, often in spring and summer.
A number of metrics/parameters are used in the assessments using benthic macroinvertebrates but no Member State is fully compliant with the requirements in the WFD, meaning that not all parameters are covered and type-specific reference conditions are missing. Also, the country monitoring lakes by benthic macroinvertebrates do it by making use of a different combination of metrics, the following metrics are either used alone or in combination: abundance and relative abundance, diversity indicators, indicator species lists, frequency of occurrence of individual taxa, number of taxa, group ratios, average score per taxa, biotic score, biotic integrity index, saprobic index, average score per taxon and ratio of littoral to profundal taxa. Again it clearly shows lack of harmonization/ standardization of methods.

Further evidence of need for development of benthic macroinvertebrate methods for lake ecological status assessment comes from the WFD Intercalibration of Member States biological monitoring systems, where benthic macroinvertebrate methods are not being intercalibrated in any of the GIGs due to lack of methods and data (Table 10).

FP6 Strep Project REBECCA

REBECCA was designed to provide relevant scientific support for the implementation of the WFD. The two specific aims of the project are, first, to establish links between ecological status of surface waters and physico-chemical quality elements / pressures from different sources, and, second, to develop and validate tools that Member States can use in the process of classification, in the design of their monitoring programs, and in the design of measures in accordance with the requirements of the WFD.
These objectives were followed by collecting existing knowledge and analyzing knowledge gaps, and using this information as a basis for analyzing the dose-response relationships between pressures and chemical/biological quality elements based on existing data.
It was found that the use of benthic macroinvertebrates in the assessment of lake ecological status and for management purposes is currently hampered by the knowledge gaps detailed below.
Taxonomic composition, abundance, diversity and sensitive taxa of profundal invertebrates have been widely used as reliable quality elements for monitoring deep lakes in relation to eutrophication. Further studies are needed to test applicability of identified metrics and their relationship to nutrients in other regions/lake types and determination of reference conditions.
More recently littoral benthic invertebrates have been investigated with differing results in different regions. Further investigation is needed to understand the distribution of littoral invertebrates and their relationship to nutrients and hydromorphological modifications, and to identify reference conditions.
The understanding of how invertebrates respond to coupled pressures (e.g. eutrophication and toxics) is at an early stage and no specific invertebrate indicators have been selected for lakes. The identification if indicator species that react specifically to single pressures would enable an additional diagnostic dimension to the assessment of lakes.
Further, the data collation process for REBECCA also highlighted some important issues and decisions that must be made when collating macroinvertebrate data from a number of sources, including difficulties with the incorporation of data sampled with differing techniques (e.g. various mesh sizes, lengths of time taken for sampling, areas sampled); differing taxonomic resolution among datasets, which necessitates considerable loss of information for database-wide analyses, and variable specificity of information on habitat structure or substrate sampled.

Conclusions

The WFD requires that all Member States have compliant and comparable monitoring systems in operation before the end of 2006. Evidence from different European projects show that for benthic macroinvertebrates in lakes that would require a major effort from several Member States. Hence, the co-ordinated development of a monitoring system would be desirable.
Lakes are unevenly distributed in Europe. Generally, the percentage of lakes in the land covered decreases from north to south and remains below 0.5% in several southern countries in Europe like Bulgaria, Slovenia, Macedonia, France, Spain, and Portugal. This distribution is also reflected in the current knowledge of the southern lakes ecology, and in the effort needed in southern countries to meet the WFD deadlines for the assessment of ecological status.
Current knowledge of benthic macroinvertebrates ecology and population dynamics is mostly based on data gathered from lakes in northern and central areas of Europe, and further studies need to be carried out to understand the macroinvertebrate communities in southern European lakes. Moreover, there is a general need to understand/determine the benthic macroinvertebrate communities at lake reference conditions.

Attention should be given to the Mediterranean lakes that depart considerable from the contemporary limnological paradigm and of which there is still a limited kwowledge of their flora, fauna and little understanding of the biologically-mediated ecological processes (see Alvarez Cobelas et al., 2005). These are highly ecologically complex systems, in general very small, with a catchment area much larger than their size. They experience both a longer vegetation period and a stronger seasonality in water supply which occurs outside the hot season, often from groundwater sources.

There are a number of macroinvertebrate indicators sensitive to mostly eutrophication or acidification pressures. Again, these were developed for northern and central European lakes, and their applicability to other geographic areas than those for which they were developed for needs to be validated.

In general there is a need for further investigation of the benthic macroinvertebrate indicator value for lake-types (including consideration of indicator variability) of benthic macroinvertebrate communities from different lake zones (littoral, sub-littoral and profundal) and to understand their sensitivity to various pressures. While there is already some regional knowledge on the sensitivity to eutrophication and acidification, other pressures (i.e. hydromorphological alteration, toxics, coupled pressures) have rarely been studied.

For the identification of indicators sensitive to specific pressures, research on functional relationships is needed, too. As a first step, a database on the ecological/biological species traits has to be compiled, which would enable further functional analyses. Research in lotic systems (streams and rivers) has shown that species traits remained stable for native river assemblages along environmental gradients across Europe (Charvet et al. 2000, Statzner et al. 2001) and thus may reliably indicate human impacts in different regional settings (Charvet et al. 1998, Doledec et al. 1999, Statzner et al. 2001).

Finally, the WFD 'sponsors' harmonized assessments through its requirement for intercalibration of biological monitoring systems and foresees the use of standard methodologies (national and international) to ensure the quality and comparability of the biological data collected by the Member States. Currently CEN standard methods do not cover these standardization needs (see table 11 for lake benthic macroinvertebrate standard methods). An effort is being made to prioritise standardization of WFD relevant methods, in areas such as lake macroinvertebrates. The process would benefit from input from targeted research in the area.

Table 10 Biological quality elements used in the WFD Intercalibration exercise for lakes.

Biological element	Use of biological element/ GIG
Phytoplankton chlorophyll	All GIGs
Phytoplankton – taxonomic composition	All GIGs have started
Macrophytes	All GIGs have started (excluding Med)
Benthic fauna	Only Nordic GIG
Fish fauna	Not yet started for any GIG

Table 11 Macroinvertebrate standard methods applicable to lakes, a) currently under review by CEN to include WFD related requirements, b) a CEN new proposal.

a)	Water quality – Methods of biological sampling - Guidance on handnet sampling of aquatic benthic macro-invertebrates (ISO 7828:1985)
	Water quality – Methods of biological sampling – Guidance on the design and use of quantitative samplers for benthic macro-invertebrates on stony substrata in shallow freshwaters (ISO 8265:1988)
	Water Quality – Sampling in deep waters for macro-invertebrates – Guidance on the use of colonization, qualitative and quantitative samples (ISO 9391:1993)
b)	Guidance on field & laboratory procedures for processing samples of benthic macroinvertebrate from surface freshwaters.

4 Macroinvertebrates and lake ecological assessment, synthesis and way forward

The importance of macroinvertebrates in lake ecological assessment

Macroinvertebrates are one of the key components of lake ecosystems. Whereas other biological elements might be considered to be more sensitive to a given pressure, or simply respond in a way more readily understood, macroinvertebrates remain as a surveillance monitoring requirement of the WFD. They are essential to any whole lake assessment and their inclusion in Annex V of the WFD is cognoscent of their key role in the structure and functioning of aquatic ecosystems the expression of which is defined as ecological status in Article 2.21 of the WFD:

"Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters, classified in accordance with Annex V."

In the implementation of the WFD for lakes, the use of benthic invertebrates has been widely neglected so far. Obviously, there is a widespread misconception on the potential additional value of an assessment tool based on benthic invertebrates. The assumption that benthic invertebrates mostly respond to eutrophication pressure, as phytoplankton, neglects the fact that many lakes are subject to other significant human impacts. This means that the ecological integrity of lakes cannot be assessed by solely surveying phytoplankton. Hence, an assessment system based on benthic invertebrates could effectively indicate multiple pressures.

In summary, assessment by benthic invertebrates could be useful by the following reasons.

- Diversity and ecological functions of a lake ecosystem are not solely based in the pelagic water body. Benthic invertebrates include several trophic guilds and consumer levels, hence they are closely interrelated to ecosystem processes (see below), and thus may well reflect ecosystem health.
- Pressures are different in the pelagic and littoral zones of lakes. Littoral benthic invertebrates thus indicate different pressures (e.g. land use or non-point inputs in the catchment, hydromorphological changes) than what can be indicated by plankton.
- Benthic macroinvertebrates integrate combined and varying pressures better than plankton, as they exhibit life cycles ranging between several months and several years.
- Benthic macroinvertebrates can be found in most lakes, and during most of the year. This is a substantial practical advantage over other biological quality elements, e.g. phytoplankton (highly variable in time, present only during vegetation period) or macrophytes (not present in some lake types and degradation levels, fully developed only in summer, slow reaction to pressures due to hysteresis effect).

The littoral zone of lakes plays a crucial and dynamic role in regulating the flows of nutrients and materials from the watershed. In this lake zone, benthic invertebrates take an intermediary position between primary producers and microbial decomposers on one side and vertebrate predators (mainly fish) on the other side. Hence, the energy flow to the pelagic depends also on a well functioning benthic energy channel from the littoral. Notably, changes in any ecosystem process are potentially reflected by related changes in the structure (abundance and species composition) of the benthic invertebrate assemblages.
Hence, lake assessment based on benthic invertebrates conveys crucial information on the ecological status of a lake ecosystem which is not reflected by planktonic organisms. Thereby, the information content of benthic invertebrates varies depending on the depth zone sampled. Hence, a benthic invertebrates assessment tool seems to be essential if a holistic assessment of lake ecosystems is aimed at. Thus, lake macroinvertebrates are likely to prove especially useful when dealing with the effects of combined pressures and lake shore modifications and hydromorphological alterations which would have a direct impact on the habitat of littoral macroinvertebrates. The community change of macroinvertebrates with eutrophication is also likely to convey key information, summarising the magnitude of the alteration of functional processes resulting from fundamental changes in the communities of primary producers (macrophytes, phytoplankton and phytobentos).

To date, little work has been done on lake macroinvertebrates, especially in the eulittoral, in contrast to the large volume of research carried out on running waters (Figure 1). It is therefore clear that much more specific work on lake macroinvertebrates is required which should be done in the context of achieving a broader understanding of lake system functioning and its relationship with ecological status to achieve the aims of the Water Framework Directive.

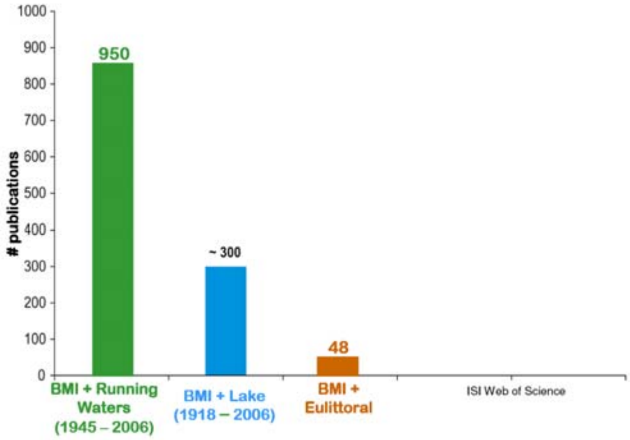

Figure 1. Published knowledge on benthic macroinvertebrates (BMI).

Current issues and questions to be addressed

The current substantial knowledge gaps on lake macroinvertebrates (Figure 1) were discussed at an expert meeting held at the Joint Research Centre on the 29-30th of June (see Annex 1). Although many of the issues listed below are interrelated and a comprehensive research approach should be targeted, it is instructive to divide the main knowledge gaps and questions into basic and applied research needs. Other issues are relevant if addressed at the continental level (e.g. Europe).

Basic research needs: identifying the drivers that govern spatial variation of benthic invertebrates within a lake and across lakes; defining the role of macorinvertebrates within the lake material cycling and within the lake food web

- Spatial and temporal scales issues
- Habitat constrains and species traits
- Response to watershed alterations
- Role of the littoral zone processes for the whole lake organic carbon dynamics
- Importance of littoral macroinvertebrate production for fish production
- Influence and interactions strengths with other components of the food web (macrophytes, phytobenthos and fish).

Applied research needs: defining bioindicators to assess lake ecosystem functionality and their response to system alterations

- Definition of the scale and lake zone to be targeted into monitoring programmes
- How to deal with the variation of the macroinvertebrate assemblage among different lake sites to develop lake specific and/or site specific assessment tools
- Response to different pressure intensity and relative relevance for different lake zones
- Interaction among morphological factors and pressure intensity to quantify the risks and effects
- How to cope with strong seasonality
- Relevance of habitat/spatial issues for the reference condition concept
- Sampling design (comparison of consistent methods, including analysis)
- What metrics to develop, including indicators recovery and restoration
- Habitat issues: the influence of sediment, depth and macrophytes, and response of different habitats to watershed pressures and their reference conditions.

Research issues relevant at the European scale

- Importance of climate change and ecoregion on lake processes and function assessment and reference condition definition
- Management of lakes especially in the Mediterranean region under the influence of climate changes, which affects keyprocesses in these ecosystems
- Assessment and management of reservoirs, which are of great economic value in Mediterranean countries.
- Strategies to limit the spread of invasive species
- Biodiversity issues (taxonomic problems, rare species, defining key species for ecosystem function, biogeographical diffrences), which give a link to the EU Habitat Directive.

Key actions proposed

The report hilights the urgent need of developing European-level research able to place macroinvertebrate assessment into the framework of lake ecosystem functioning. Such European-level research could partially use knowledge from regional research on the indication of lake acidification by littoral invertebrates, or on the indication of eutrophication by profundal invertebrates. Key elements under consideration should include: the role of invertebrate in the material cycling and the functional role of littoral invertebrates within the ecosystem in different lake types, their response to watershed and shoreline alterations, the importance of spatial and temporal factors on assemblage dynamics and relative bioindicator behaviour, their influence on reference conditions, habitat constraints on species traits, taxonomic and methodological limitations, Geographically, the uneven distribution of lakes in Europe and the peculiarity of Mediterranean lakes, the impacts of climate change on lakes, and the great number of reservoirs in some Mediterranean regions should be taken into account.

One of the key conclusions was the need to steer research to place more emphasis on lake ecosystem function and its relationship to pressures rather than the traditional overemphasis on community structure.This reflects the necessity to assess the status of European lakes based on parameters which are less variable due to climatological or biogeographical gradients.Research on the role of macroinvertebrates in lake ecosystem functioning needs to be designed to both improve ecological understanding and to lead to better ecological assessment methods. This needs to be accomplished aquiring information with both experiments and new field data.

The design and execution of field experiments is necessary:

- To identify the functional role of littoral benthic invertebrates in lakes types that differ by natural reasons in the structural complexity of their litoral zones.
- To relate, including the use of experimental techniques the relationship between ecological function and community structure.
- To examine the response of the invertebrate community structure with pressure in key habitats and lake zones.
- To assess the influence and interactions strengths with other components of the food web (macrophytes, phytobenthos and fish).

The acquisition of new data from field sampling campaigns is necessary:

- To examine key strategic lake types across Europe using standardised methods in key habitats and lake zones. This should include appropriate identification, definition and validation of typologies and reference conditions.
- To use the knowledge acquired from the field experiments to guide effective ecological assessment techniques and biodiversity and habitat assessment.

Other issues

Training

In a number of member states there are hardly scientists working on benthic invertebrates in lakes, or on its functional aspects. Hence, some workshops or training elements on the ecology of the benthic zone of lakes seem to be necessary in order to achieve a common methodological level that enables conducting joint research following the same protocol.

Compilation and analysis of existing data

As there is only limited knowledge on the ecology of the benthic and littoral zones of lakes in Europe, existing databases should be compiled in order to support research approaches.

- Compilation of existing long-term datasets on benthic invertebrates. With such data, an analysis for interannual variability could be performed, which would be important basic information for further research.
- Compilation of major human pressures in specific ecoregions to lakes and reservoirs, e.g. from water management and navigation agencies. This is especially important to obtain estimates for human pressures independent from benthic invertebrate analyses.
- An analysis on the relevance of littoral zones for rare species (listed in the annexes of the EU Habitat Directive).

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Chapter 3

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6 APPENDIX

Workshop for developing tools for the ecological classification of lakes using macroinvertebrates

Held at the Joint Research Centre, Ispra, Thursday 29th – Friday 30th June 2006

Agenda

Thursday 29 June:

08:45		**Bus Departs Hotel Lido**
09:30	Wouter van de Bund	Welcome and update on WFD implementation process
10:00	Ken Irvine	Introduction: - Objectives of the meeting - What do we know and need to know about benthic communities to support policy?
11:00	**Coffee**	
11:15	Martin Pusch	Habitat structure and ecosystem processes in the littoral zone of lakes
12:00	Henn Timm	Macroinvertebrates as a tool for typology and classification of lakes in Estonia
12:30	**Lunch**	
14:00	Ian Donohue	The development of ecological classification tools using macroinvertebrates for lakes in Ireland and the UK
14:30	Jeremy Biggs	Macroinvertebrates in UK ponds and lakes: a review of Pond Conservation's work on benthic macroinvertebrate assemblages
15:00	*Group discussion*: Ken Irvine chair	Identification of specific gaps in our knowledge *(ranking of priorities)*
15:45	**Coffee**	
16:00	*Group discussion* Ken chair	Adding value and the development of sound methods suited to lake classification (*why a coordinated effort among European research groups is/ or not needed*)
17:15	Ian Donohue	Sum up of discussion
17:30	*End*	
19:30	*Social dinner*	Please assemble in the lobby of Hotel Lido for 19:10

Friday 30 June:

08:30		**Bus Departs Hotel Lido**
09:15	Leonard Sandin	A suggestion for a lake AQEM project (LAQEM)
09:45	Brigitte Lods-Crozet	Use of oligochaetes and chironomid communities in biomonitoring programs in Swiss deep lakes
10:15	Bruno Rossaro	Development of a BQI for Italian lakes
10:45	**Coffee**	
11:00	*Group Work* Leonard Sandin chair	Future work necessary to fill the gaps: what basic ecological concepts should we consider (*building the conceptual framework*)? Towards a common integrated approach
12:30	**Lunch**	
14:00	*Group work* Leonard Sandin chair	Discussion of a way to promote a dynamic exchange of information between research groups and how best to coordinate it (*building the critical mass*)
15:45	**Coffee**	
16:00	Ken Irvine	Sum up and conclusions
16:30	**Meeting ends**	Transportation to hotel / airport

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Abstract

Recent extensive reviews of the current state-of-the-art of ecological water quality assessment systems in Europe have revealed that, while practical (and WFD-compliant) assessment tools using macroinvertebrate parameters are already in use to assess the ecological quality of rivers, in many European countries there are currently no working macroinvertebrate assessment systems for lakes. Indeed, this has been recently identified as one of the major ecological ‘knowledge gaps’ impeding the full assessment of ecological quality of lakes as required by the WFD in a literature review carried out within the EU project REBECCA.

The current lack of knowledge is also limiting the fulfillment of the EU-wide intercalibration of the lake ecological quality assessment systems in Europe, and thus compromising the basis for setting the environmental objectives as required by the WFD, particularly concerning quantification of the ecosystem impacts of nutrient loading pressures (i.e. eutrophication), which is the most wide-spread pressure on surface water ecological quality in Europe.

This report focuses on the ecological assessment of lakes using benthic macroinvertebrates. The review is structured by the major anthropogenic pressures affecting lakes: eutrophication, acidification and hydromorphological alterations. Current knowledge and examples of use are presented in the context of the required understanding needed to use benthic macroinvertebrates in lake assessment as required by Directive 2000/60/EC. The current methodological perspectives and limitations in the use of benthic macroinvertebrates are discussed.

The report highlights the urgent need of developing European-level research able to place macroinvertebrate assessment into the framework of lake ecosystem functioning. Key elements under consideration should include: the role of invertebrate in the material cycling, the functional role of littoral invertebrates within the ecosystem in different lake types, their response to watershed and shoreline alterations, the importance of spatial and temporal factors on assemblage dynamics and relative bioindicator behaviour, their influence on reference conditions, habitat constrains on species traits, taxonomic and methodological limitations. The uneven geographical distribution of lakes in Europe and the peculiarity of Mediterranean lakes, the impacts of climate change on lakes, and the great number of reservoirs in some Mediterranean regions should be taken into account.

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An ecological assessment system for sub-alpine lakes using macroinvertebrates –

The development of a parsimonious tool for assessing ecological health of European lakes

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