COST Action 626

Data collection concerning macrobenthos

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PREFACE

The aim of this report is to present the state-of-the-art on monitoring macrobenthos and related stream characteristics. On top of this, issues on database set-up and management are also discussed. In this way we hope to allow model developers and users to get some insight on the critical aspects of collecting data on macrobenthos.

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CONTENTS

1 OBJECTIVES

The objectives of this overview were to present major issues on macrobenthos data collection. Most of the topics focus on streams, but are often also applicable for stagnant waters.

The first part deals with sampling sites and protocols. In this chapter, an overview is given on sampling methods. A comparison is made between the methods applied in different European member states. Scaling aspects are also a central issue in this part.

The second part is focussing on the identification of the macroinvertebrates. Different identification keys are presented and the efforts for several identification levels are compared. In the Appendix, a list of valuable references on the identification of macroinvertebrates is provided.

In the third part, the relations between physical and biological components are discussed. An overview of additional measurements is presented, as well as their relation with the macroinvertebrate communities.

The last part is dealing with quality assurance and quality control of the generated data. This chapter is of major concern, because the data quality determines the reliability of the derived models and their validation. Data base set-up, meta-data bases and the maintenance of data bases are also discussed in this chapter.

2 SAMPLING SITES AND PROTOCOLS

2.1 Sampling design

Often the argument is made that comprehensive understanding of ecological phenomena requires long-term monitoring of salient patterns and processes in adequately replicated control and experimental units at appropriate spatial and temporal scales using sound samping design and statistical analysis. Although this line of reasoning is based on fundamental ecological principles and existing knowledge base, this theoretical optimum in research design can rarely be achieved (Michener & Brunt, 2000).

A standardized sampling desing for macrobenthos is described in Standard Methods (1997). The practical selection of sampling sites depends on the project objectives and has often to be adapted to the particular needs of the study. When defining the locations and number of sampling sites, the following factors should be considered (Nguyen *et al*., 2000):

- study objectives;
- historical data and other available information;
- size of the sampling area;
- characteristics of the sampling substrate and overlying water;
- availability of funds and estimated cost of the study.

2.2 Sampling methods

A lot of methods exist to collect macroinvertebrates in aquatic systems. Nguyen *et al*. (2000) gives an overview on the most convenient techniques to sample macrobenthos (Table 2.1). This CD also gives a detailed description of the equipment and its use is illustrated by means of a video. Details can be found in: Nguyen, L.T.H., Duong, D.T., Colles, A., Van Eyck, A.S., De Pauw, N., Gravendeel, R., Ollevier, F., Vo, X.T., Sorgeloos, P., Van Heddegem, J., Ooghe, B., Duong, L.E. & Le, H.N. (2000). Sampling of shallow still waters: lakes, ponds and wetlands. (Water quality: from sample to quality assessment. CD 1)., Katholieke Universiteit Leuven, Belgium & Can Tho University, Vietnam. ISBN 90-76978-07-7.

Table 2.1: Macrobenthos sampling equipment (Nguyen et al., 2000).

Net samplers	**Corers or cylindrical samplers**
Handnet Scraper	Kajak-Brinkhurst corer Borge corer Tube corer
Grab samplers	**Colonisation samplers**
Ponar grab Wildco box corer Birge-Ekman grab	Standard colonisation unit Multi-plate or modified Hester-Dendy sampler

Van Veen grab	Artificial substrate – Bag sampler Artificial substrate – Box sampler

In Standard Methods (1997), also a detailed description is formulated for the collection of bethic macroinvertebrates by means of various methods (Biological examination (10000)).

2.3 Scale (spatial/temporal resolution)

2.3.1 General introduction

Natural environments are inherently variable in space and time, and this heterogeneity makes ecological patterns and processes scale-dependent: what seems apparent at one scale may collapse to noise when viewed from other scales. Yet, variability is often regarded as uninteresting noise that interferes with our ability to detect ecological patterns. Only rarely is the spatial scale consciously chosen to be appropriate to the problems studied, implying that results from small-scale studies cannot be reliably 'scaled up' to more relevant spatial and temporal scales (Thrush et al. 1997, Rafaelli & Moller 2000).

Streams are notoriously heterogeneous environments at scales ranging from a few millimetres to several kilometres. Furthermore, stream systems are organized as natural spatio-temporal hierarchies (microhabitat - pool-riffle sequence - stream section - drainage network), thus calling for a multi-scale approach to stream research (Frissell et al. 1986; Hawkins et al. 1993). At the scale of local communities, however, present knowledge is based on controlled experiments conducted over very limited spatial scales. Recently, worries have been expressed about the relevance of such small-scale experiments in explaining patterns at the whole-system scale (e.g. Englund 1997).

A hierarchically-based approach to stream ecology has indicated that some features of stream biota are locally controlled, while others are under a larger-scale, regional control (Palmer et al. 1996; Wiley et al. 1997). For example, variation in stream discharge is controlled by catchment properties, like soil type, topography and vegetation. Therefore, species responsive to flood disturbance are under regional control. On the other hand, riparian vegetation regulates the input of terrestrial leaf litter into a stream reach, and retention of leaves onto the stream bed is related to the physical structure of the substrate. Thus, species dependent on leaf litter as their major food source should be under local control. These spatial scales can be viewed as a set of environmental filters (sensu Poff 1997), selecting species with suitable traits (individual size, life history, feeding group, etc.) from the regional species pool. Studies including both local and large-scale variables are badly needed to better understand the relative roles of factors operating at different scales. In one such study, Sandin & Johnson (unpubl.) used Partial Canonical Correspondence analysis on an extensive data set (Swedish national stream survey) to partition total variance in macroinvertebrate species data into local, catchment- and large-scale factors. They showed that local scale physical and chemical factors were most important, explaining 45 % of the among-site variation. Clearly, this example is restricted to boreal streams, and more similar studies in different environmental settings are needed. The heterogeneity of the in-stream physical environment has been the focus of much research (e.g. Hildrew and Giller, 1994). Streams offer excellent opportunities for studying heterogeneity in both time and space. A wide range of patterns and processes in streams may be influenced by physical and biological heterogeneity, including spatial distribution and persistence of in-stream biota (Palmer and Poff, 1997). Multi-scale studies of physical and biological heterogeneity have been used to analyse the influence of predator-prey interactions on macroinvertebrate distributions (Crowl et al., 1997) and habitat utilisation by macroinvertebrates during spates (Lancaster and Beleya, 1997).

The majority of studies have mainly focused on two spatial scales: micro- and macro-scale. A number of studies have investigated flow patterns around macroinvertebrates (e.g. Statzner and Holm, 1986; Statzner et al., 1988) and flow velocity preference for single species or groups of species (e.g. Fonseca and Hart, 1996). These studies have all been conducted under laboratory conditions and have provided valuable information of flow preferences for a number species. Some studies have showed that interactions between species (e.g. predator-prey) occur where their flow preference curves overlap (Peckarsky, et al., 1990). The drawback of this type of studies is that they are not readily transferred to natural, larger scale systems. However, they have emphasised the importance of physical features for stream invertebrates.

2.3.2 Micro-habitats (size of sample)

Most approaches to stream bioassessment (e.g. RIVPACS; AUSRIVAS; BEAST) work on individual stream reaches, i.e. samples collected at a stream site are assumed to reflect the condition of the whole stream. Such a relationship has rarely been tested, however, although there is no a priori reason to expect that benthic samples from neighbouring riffles are similar in species composition (but see Hawkins & Vinson 2000). The few direct tests available thus far do not lend support to this contention (Downes et al. 1993, 1995; Li et al. 2001). For example, Downes et al. (1995) showed that adjacent riffles varied considerably in both faunal densities and near-bed flows. Similarly, our own studies (J. Heino et al., unpubl.) in river Kiiminkijoki, northern Finland, have shown that riffles separated by no more than a few hundred meters may support widely different macroinvertebrate assemblages.

At the within-riffle scale, studies on microhabitat distribution of lotic invertebrates have largely employed conventional sampling devices defining a predetermined area of the stream bottom. Such studies are, however, plagued by the fact that stream organisms display patchy distributions in a spatially and temporally variable arena. Unfortunately, the scale of conventional field techniques is often quite different from that perceived by benthic organisms. For example, in a colonization study employing substrate baskets of 25 x 25 x 10 cm, Reice (1981) documented generally weak spatial interactions among members of the invertebrate community of a woodland stream. One of the species pairs studied by him was the grazing caddis *Leucotrichia* and *Eukiefferiella* midge larvae. In an experimental study, McAuliffe (1984) later showed, however, that Reice's result was an artefact of the study plot used. In fact, these two species are heavily competing for limited resources (periphytic algae), but they interact at very small spatial scales (up to few centimetres), and this interaction could only be detected if a plotless sampling design was used. Indeed, it has been recommended that if the appropriate spatial scale for a study cannot be determined a priori, ecological field studies, regardless of their specific objectives, should be conducted across a variety of spatial scales (Ives et al. 1993). Use of plotless designs and related spatial statistics have rarely been pursued by benthic ecologists, although the technical and statistical machinery needed are widely available (e.g. Muotka & Penttinen 1994; Cooper et al. 1998).

Stream communities are known to be highly resilient to environmental disturbances, i.e. they recover rapidly after a disturbance (e.g. Resh et al. 1988). A mechanism suggested to explain the high resilience of stream macroinvertebrate communities is the presence of hydraulic refugia within the stream substratum (Townsend 1989). These refugia are small patches where conditions remain essentially unaltered even during the highest discharges. There is some evidence for the existence of such refugia in small streams (Lancaster & Hildrew 1993a), and benthic animals may indeed aggregate into these refugia during floods (Lancaster & Hildrew 1993b). As discharge recedes to pre-disturbance level, animals that survived disturbance in these benign microhabitats may rapidly recolonize vacant patches in the denuded stream habitat. At the community level, this translates into a fast recovery rate, even after the most severe disturbances. The presence of flow refugia has typically been tested using standardized FST-hemispheres, a series of hemispheres with a constant shape and size and a known distribution of specific gravity (Statzner & Müller 1989). Although this method has many advantages over more traditional approaches to describing near-bed flow environments, it is also somewhat questionable, because data on near-bed flows thus obtained cannot be reliably interpolated to smaller scales. Yet, it is this small-scale variability that is often most important for benthic organisms. For example, using a time series of highly sophisticated microflow measurements, Hart & Fonseca (1997) showed that, at the onset of storm flows (natural or manipulated), larval blackflies moved from their feeding sites on stone tops to more safe sites on other stone surfaces. After the storm flows receded, the larvae returned to their original attachment sites within a few hours. Based on these results, Hart & Fonseca (1997) suggested that the ability to track temporal fluctuations in flow is an important selective factor for stream organisms.

2.3.3 Meso-habitats (morphological entities: pools, riffles, …)

Relatively few studies have investigated the linkage between physical features and macroinvertebrates on the meso-scale. Meso-scale is particularly interesting as it roughly corresponds to what is normally perceived as the "functional habitat" scale (Harper et al., 1995). The meso-scale has had particular interest in England, where meso-scale habitat characteristics and associated macroinvertebrate communities have been studied in detail in groundwater-dominated chalk streams (Wood et al., 1997). The majority of meso-habitats studies outside Great Britain have been carried out in connection with stream restoration projects (Friberg et al., 1994; Friberg et al., 1998).

2.3.4 Macro-habitats (river stretch)

Effects of physical conditions on the macro-scale, typically a stream or a stream reach, have also been studied fairly intensively. Especially the effect of stability on macroinvertebrate communities has been widely studied, especially in New Zealand (e.g. Death and Winterbourn, 1994; Death and Winterbourn,1995; Townsend et al., 1997). These studies have shown that both structure, function and diversity are impacted by stability. Unstable systems will have fewer individuals, fewer species and less specialised feeding groups compared with stable systems (e.g. Winterbourn 1995). One problem with this type of studies is that they often cover large gradients in disturbance, which is not common in e.g. lowland areas in Europe. Furthermore, the ability to resist disturbance is theoretically related to species diversity (e.g. Elton, 1958; Tilman, 1996; Doak et al., 1998) and consequently the results from some of the studies performed in e.g. New Zealand may not be applicable in Europe where the regional species pool is larger.

2.3.5 Practical experience from a case-study in Denmark

Stream restoration projects provide excellent opportunities for studying the relationships between physical and biological conditions the reason being that the extent and composition of the physical environment is fully controlled.

As an example, results from a project where lowland streams are re-meandered are discussed. In this project a new meandering course is created and filled with physical structures such as gravel riffles, riprap structures made by stones etc. This re-meandering project has been followed in the Danish river Gelså since its restoration in 1989 (Friberg *et al*., 1994; Friberg *et al*., 1998). In 1995, all meso-habitats were mapped in a restored reach and compared with an not restored, control reach. The mapping was undertaken by measuring several physical features (e.g. substrates, velocity, shear-stress) and subsequently performing a cluster analysis (Kronvang et al., 2000). Five distinct physical habitats at meso-scale level were classified of which only one was restricted to the restored reach - gravel riffles were actively added to the reach as part of the restoration six years prior. Macroinvertebrate samples were collected within each habitat, and the community composition of the 5 habitats and the two reaches (restored and control) was compared. Overall, there was a high degree of overlap between habitats when data were analysed using (DCA) Detrended Correspondance Analysis (Friberg et al., 2000). Only the gravel riffle habitat had a community composition that was different from the other habitats. In general, the un-estored, control reach had more species and individuals in any habitat compared with the restored reach, probably reflecting much denser macrophyte growth in the control reach habitats. In conclusion, this study showed only a weak linkage between the physical features on the meso-habitat level and macroinvertebrate composition. However, the study indicates that macrophytes may play a key role in lowland streams, a contention which is also supported by several studies showing how macrophytes alter physical conditions in streams (Sand-Jensen, 1997). The macrophytes alter the current velocity pattern and generally reduce the current velocities (Sand Jensen and Pedersen, 1999; Sand-Jensen and Mebus, 1996), thereby enhancing the deposition of fine sediments (Sand-Jensen, 1998).

2.4 Comparison of standard methods between different European countries

The study of macroinvertebrates to assess the water quality of rivers is not a new task and in Europe it started in the very beginning of the 20th century with the Saprobien system (Kolwitz & Marsson, 1902). Since then two major groups of methods derivated. The saprobic and the biotic indices (see Fig.) Moreover the structure of the community measured by different indices of diversity it has been used with the problems of the different results obtained depending of the taxonomical level of identification (i.e.: Guerold, 2000).

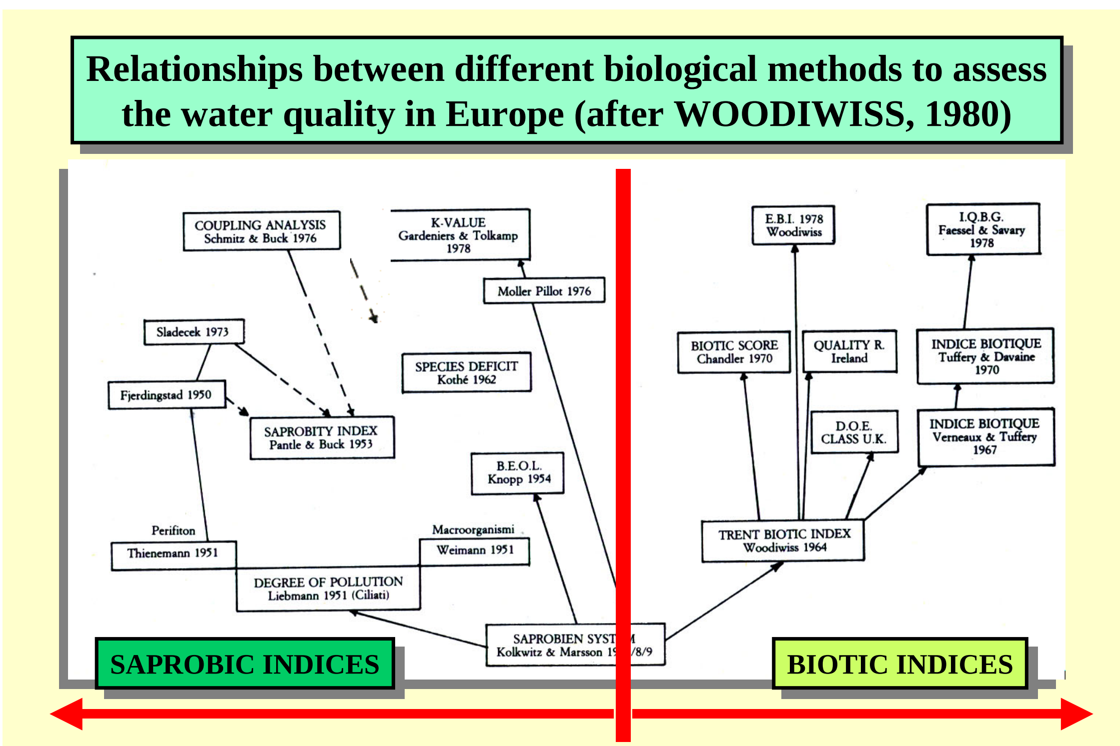

Fig. 2.1: The Saprobic and Biotic indices are the two major group of methodologies used in Europe to assess the water quality.

Despite the great amount of procedures based on different level of the community looking at different hierarchical levels: Biochemical, Physiological, Individual, Populations, Community, etc... (see: Johnson *et al*, 1993, Rosenberg & Resh, 1996), in Europe most of the stress have been pointed out in the development of different biotic indices rejecting saprobic and diversity indices (Metcalfe, 1989). In fact the Saprobic

System is still used only in Germany, The Netherlands, Austria and some countries from the ex Yugoslavia. Thus, Austria invested a lot of effort in develop taxonomy of aquatic organisms to be able to implement and update this methodologies (Moog, 1995).

An overview of the status of the art was updated in a publication of the Commission of the European Communities (Newman *et al.*, 1992), and most of the methods can be consulted in: Ghetti & Bonazzi (1981), Hellawell (1986), Metcalfe (1989) and Ghetti (1997)

A recent and thorough overview on different macroinvertebrate assessment wast made within the European AQEM-project (www.aqam.de). Annex 6 in the 'Manual for the application of the AQEM system', describes the most common European metrics used for calculating the ecological quality of individual stream types.

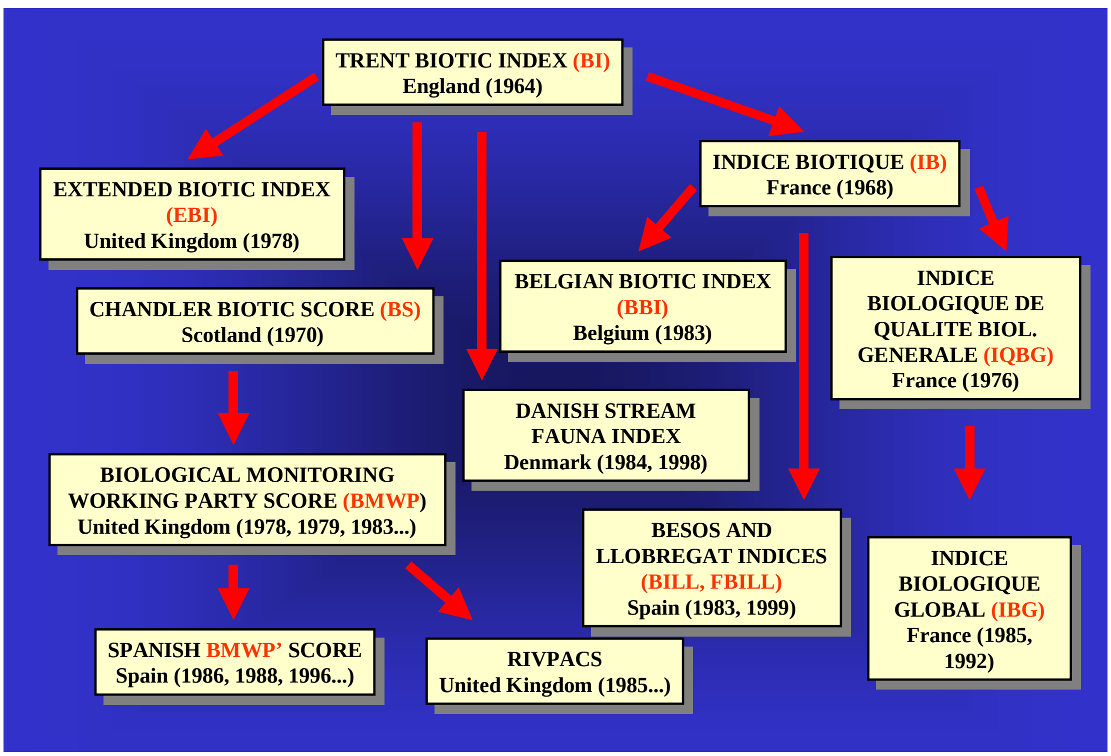

Fig. 2.2: Development of the most extended biotic and score systems existing in Europe

Those methods intend to study the structure of the community assigning values according with natural status versus stress. The RIVPACS approach represents a different point of view because this is a predictive model that offers a prediction of expected fauna at a given site. Thus by comparing the existing fauna with the potential (predicted) one it is possible to know the degree of deviation and thereafter to establish the degree of alteration and/or goals for restoration (see: Wright, 2000 and Wright *et al.*, 2000).

2.5 Protocols and standardisation

The great diversity of methodologies existing in the different countries implies the applications of different sampling protocols as well different identification level and elaboration of samples and thereafter this could implies a great difficulty to compare this "universe" of methods that could difficult an European Standardisation. This is why along different meetings of the European Commission for the Normalization (CEN/TC230/WG2/TG1) this topic appeared. Concluding that no matter what methods use each EC country because exist correlations between the different methods and thereafter it is possible to interrelate the different results. This consideration was possible because there were an European pilot experience (Ghetti & Bonazzi, 1980) were it was demonstrate the existing correlations between results obtained by different methods. Later on Rico *et al.* (1992) and the CEN/TC230/WG2/TG1 in 1993 did a similar exercise including the BMWP and the Spanish BMWP', and recently in July 1998 in Trento (Italy) were analysed together four European methodologies from Italy (IBE), Belgium (BBI) Spain (BMWP') and England (RIVPACS) (Siligardi *et al.*, 2000; Mancini & Spaggiari, 2000).

3 IDENTIFICATION LEVEL

3.1 Identification keys

Taxonomic resolution has always been considered to be a critical aspect of limnological studies and particularly when aquatic organisms such as invertebrates are used for the bio-assessment of water quality. In a recent review of 45 lotic studies using macroinvertebrates, Resh and McElravy (1993) observed that the taxonomic levels of identification were very variable according to the groups of selected organisms. Indeed, insect groups (except Odonata) were identified most commonly to the genus level, although the Ephemeroptera, Plecoptera and Simulidae (Diptera), as well as two non insect groups (Platyhelminthes and Crustacea) were identified to species level in about one-third or more of the lotic studies. Nematodes, annelids and water mites were identified most often to family level or above. These differences reflect probably the current state of the taxonomic knowledge of these organisms.

Several authors using invertebrates as water quality indicators or working with multivariate analysis to classify the various types of rivers so as to establish models of prediction based on macroinvertebrates assemblages, obtain similar results with family or finer (species and genus) level of identification (Furse 1984, Wright et al. 1988, Marchant et al. 1995, Zamura-Muñoz & Alba-Tercedor 1996). Other investigations, on the contrary, show a better sensitivity when the taxonomic resolution is more precise (Resh & Unzicker 1975, Marchant 1990, Marchant *et al*. 1997, Hawkins *et al*. 2000). Nevertheless, the concept of the species as the basic biological unit is widely accepted and everyone recognizes that the finest level of identification is needed to detect the exact biological response to an environmental disturbance, and that the lack of species-level information can decrease sensitivity and reduce the ability of a study to detect more subtle changes. The key issue here is clearly sensitivity. Authors who recommend use of levels above species recognize that such use represents a compromise between a desire for the increased information content and either its unavailability (i.e. absence of larval keys) and/or the cost (time, required expertise) of obtaining it (Resh & McElravy 1993).
Furthermore, because different species are found in different eco-regions, the use of specific indicators is often limited to specific geographical regions. Thus, the use of a family level of identification instead of the species level should guarantee an application to a wide geographical area since the same families of aquatic organisms are often found in different geographical regions.

Based on these assumptions, several "score systems" that combine general patterns of organism tolerance with elements of diversity have been developed. Metcalfe (1989) gives an exhaustive review on the use and development of macroinvertebrates' indices in Europe. For instance, score systems such as the biological Monitoring Working party (BMWP) score (Armitage et al. 1983) and the Belgian Biotic Index (BBI, De Pauw & Vanhooren 1983) assign scores to different families of aquatic organisms based on their sensitivity to particular perturbations (i.e. organic pollution, habitat disturbance,…). This scoring system provides an excellent early warning of deteriorating water quality. There are however problems associated with the use of this technique for other purposes such as to examine natural structure and variability of aquatic communities or to analyze relationships between invertebrates and complex environmental gradients or to detect subtle impairment in water quality assessment. As aquatic organisms are only identified to family level serious shortcomings should be expected in extrapolating the "biotic indices" technique beyond the purposes of its original intended use.

Indeed, it's rather obvious that assigning entire taxonomic groups (usually family level or above) to a tolerant or intolerant designation is particularly arbitrary. Among macroinvertebrates for instance, not all chironomids or oligochaetes are limited to strictly polluted conditions. Inversely, not all stone flies or caddis flies are restricted to unpolluted or reference conditions. In a wide survey of headwater streams in Luxembourg (Dohet 2001) we observed that numerous species belonging to the same family or even to the same genera (e.g. *Hydropsyche* for Trichoptera and *Baetis* for Ephemeroptera) showed the whole range of sensitivities to organic pollution. Some species within each family were tolerant, others intolerant and yet others were considered facultative as regards pollution tolerance. In this case, the family-level identification tells us nothing about ecological indicators of water quality and it seems equally true that generic-level identifications will not yield a great deal of additional information and may not be worth the time and effort.

3.2 Identification costs

Unfortunately, the need of a high taxonomic resolution required for the increased information content can't always be achieved. Actually, in numerous groups of aquatic insects, identification of immature stages cannot currently be made below the generic level. Furthermore, it is the immature stage in the life cycle of an aquatic insect that is most commonly encountered by hydrobiologists. This problem is particularly obvious for some groups like water mites, midges, black flies,… Even if many improved keys are now available for some other aquatic insects (stone flies, may flies, caddis flies,…) they are often confined to the fauna of a small region. An exception to this is the recent key for the identification to species of caddis flies immature stages, published in Austria and valid for a relative large area equivalent to central Europe (Waringer & Graf 1997). Even if most available keys allow nonspecialists to produce reliable identifications at the family and genus levels, the species level of identification is still frequently a task for specialists. Consequently, without the help of these specialists, it is almost impossible for a biologist alone to identify at the species level more than the dozen of different groups of macroinvertebrates that we can usually find in limnological studies. Interestingly, in their review of 45 lotic studies, Resh and McElravy (1993) point out that less than 25% of the papers examined in this survey reported consulting a taxonomist, and only 4% reported sending specimen to a recognized depository (i.e. established university department, government museum,…). In such conditions, the risk of misidentification is particularly high and the eventual benefit of the higher level of identification is lost in reason of the inaccuracy of the species list. Consequently, the arbitrary nature of the ecological prediction is at least as high than if a generic or a family level of identification was used. Moreover, even if experienced personnel and adequate literature are available, the cost and time consuming of species identification can be too high especially if special preparation is required (e.g. clearing and mounting water mites or midges on slides) and if such species occurred frequently in the samples.

As a first conclusion, we can state that there is still a critical research need to develop accurate identification keys to the species level and to resolve basic life history problems of numerous aquatic insects. This can be considered as a long-term objective. In the meantime, the use of higher-level categories such as genus, family or even order of identification may be considered as an acceptable compromise to predict ecological quality, particularly when the assessment concerns eco-regions characterized by few genera and species per family. In fact, it depends on the purpose of the study, the level of sensitivity required, the type of index or analysis being used, and the particular group of organisms of primary interest.

Another approach which has some merit, would be to focus attention on a selected order or family of aquatic insects. As a first advantage, this strategy implies that if a predetermined group is targeted, then specifically designed collecting techniques can be used and these will ensure the maximal coverage of the group being studied. Actually, collecting techniques and effort are not always equally efficient for each group when all groups of invertebrates are being sampled. As a result, certain taxa will be underrepresented or may even not be collected in general surveys (de Moor 1998). These targeted orders or family of aquatic insects should be selected among organisms that have an obligatory aquatic life cycle stage. It's important to choose such a group as they would be confined to the aquatic environment for at least a part of their life cycle making the need for a healthy aquatic ecosystem important. Furthermore and ideally, indicator organisms would be those species that have narrow and specific environmental tolerances (Johnson et al. 1993). Applied to species assemblages or communities, it is important that these narrow and specific environmental tolerances are distributed among different species and for different levels of pollution, so that indicators in the whole range of pollutants are available. Taxonomic soundness and easy recognition (ideally also by the non-specialist), cosmopolitan distribution, numerical abundance, low genetic and ecological variability, large body size, limited mobility and relatively long life history, knowledge of ecological requirements and suitability for use in laboratory studies are among other characteristics that should have an "ideal" indicator (Rosenberg & Wiens 1976, Hellawell 1986, *in* Johnson et al. 1993).

In conclusion, the selection of ideal indicators, based on these different features, should provide an improvement (due to the accurate species level of identification) in the precision of the prediction of ecological quality of running waters together with a reasonable cost- and time-consuming needed for the identification of target organisms since they are numerically limited and selected from those which are easiest to identify.

4 RELATIONS BETWEEN PHYSICAL AND BIOLOGICAL COMPONENTS

4.1 Introduction

During recent decades substantial research in lotic ecology has been carried out to elucidate the link between physical features and macroinvertebrate communities (e.g. Statzner and Higler, 1986; Brown and Brussock, 1991). Studies have mainly focused on the effects of current velocity and shear stress (Erdington, 1968; Bouchardt and Statzner, 1990), and substrate composition (Percival and Whitehead, 1929; Minshall, 1984; Reice, 1980) on the structure, function and diversity of macroinvertebrate communities. In addition, the dynamic aspect of changes in the physical environment has received considerable attention, and several studies have investigated the effect of temporal variation in physical features on macroinvertebrate communities. Special attention has been directed towards an understanding of the role of disturbance in structuring the macroinvertebrate community (Resh et al., 1988). Work has primarily concentrated on analysing the influence of high discharges on unstable and erosion-dominated environments in upland streams (Matthaei, et al., 2000; Death, 1996; Matthaei and Townsend, 2000; Townsend et al., 1997). Studies of stability and disturbance in groundwater-dominated streams have not been conducted until recently (Wood et al., 2000; Wood et al., 2001). Effects of erosion and deposition on the macroinvertebrates have been studied in riffles and pools (Scarsbrook and Townsend, 1993). In groundwater-fed streams, effects of small-scale erosion and deposition during baseflow condition have recently received some attention (e.g. Miyake and Nakano, 2002). Furthermore, several investigations have addressed the relative importance of physical features compared to biotic interactions in determining e.g. the realised niche of stream invertebrates. In other words, analyses have addressed a very central question: "What is the dominant factor determining the distribution of macroinvertebrates – the physical structure of the habitats or the biotic interactions or both?" (Feminella and Resh, 1990; Lancaster, 1990; Townsend, 1996).

The decisive role played by the physical environment in the composition and dynamics of fluvial benthonic macroinvertebrate communities is well known and has been described in numerous studies. There are different ways of approaching this relationship, either from the perspective of individuals or of the entire communities, or assessing the implication of an isolated physical factor or of more complex structures (habitat, mesohabitat, etc.). In this overview, we opt for a mixed approach involving aspects of both perspectives.

4.2 Individual physical factors

4.2.1 Introduction

Four factors are usually included in this approach: current, substrate, temperature and dissolved oxygen. The last mentioned factor will not be analysed individually since it is not generally considered limiting for fluvial macroinvertebrate communities in natural conditions (Ward, 1992). Indeed, in situations where it may be an important factor for the populations concerned, it is always due to conditions which embrace other abiotic and biotic factors of the ecosystem (Allan, 1995).

4.2.2 Current

Current velocity is perhaps the most important factor affecting fluvial macroinvertebrates. Allen (1995) writes " Biologists have long believed that water as a médium, and current as a force, strongly determine ecological distributions and shape anatomical and behavioral adaptations ". We might also add that current velocity also determines substrate distribution and composition, the second most important factor to influence macroinvertebrates. It also influences the transport and composition of much of the available food supply. The tiny variations in current velocity at microscale level partly due to the presence of interstice and chink or differences in rugosity may permit the adaptation and use of the riverbed by different groups of organisms. Its importance has even led to the establishment of different criteria for classifying flow environments (Davis & Barmuta, 1989) and systems for assessing and measuring the main components/descriptors which help us define more precisely the hydraulic stream ecology (Starzner et al., 1988; Statzner & Müller, 1989). It must be borne in mind that remaining in given microenvironments exposed to high velocity currents implies a high energetic cost for many macroinvertebrate species. For this reason, macroinvertebrate tend to use these microenvironments to feed or for moving during very small time periods. The morphological and physiological adaptation of the larvae of different aquatic insects (Rhithrogena, Elmis, Blephariceridae) or the net construction behaviour of Hydropsyche (Cudney & Wallace, 1980; Petersen et al., 1984) to gain maximum benefit from lotic environments is the norm followed by fluvial benthonic macroinvertebrates (Vogel, 1981; Williams & Feltmate, 1992). At this point, it should be remembered that most invertebrates are nocturnal by choice and so most studies which look at the use they make of flow microenvironments only analyse and provide information on the use of shelters during the least active phases of daily activity. Changes in the spatial distribution of different mayfly, stonefly, caddisfly, freshwater shrimp and leeche species are clearly documented (Allan et al., 1986; Statzner et al., 1988; Elliot, 2002).

It is important to underline the importance of the flow regime of different hydric regions, since the existence of annual discharge patterns may greatly alter the current velocity in the same stretch of river. Consequently, the use of given areas of the riverbed will be limited as the velocity acting on them changes. Some microenvironments may even disappear during periods of low water and drought. In such situations, there will occur a greater aggregation of populations of species like Hydropsyche lobata, which suffer reductions in the spatial availability of the optimum range of velocity conditions in these circumstances (Soler & Puig, 1999). On the other hand, some species, such as Caenis

luctuosa, prefer a slower current and in such conditions they disperse more widely and show a practically homogeneous spatial distribution (Malo, 1993).
In general, it has been observed that relatively large discharge fluctuations may not result in significant changes in a complex hydraulic key characteristic within a stream reach (Statzner et al., 1988). In this respect, it has been shown that macroinvertebrate communities have evolved to the point that they can integrate into their life history traits very wide flow ranges, as long as they are predictable (Resh et al., 1988) or, in other words, if they follow clear seasonal patterns, such as autumn or spring flooding. However, changes in the natural flow regime due to different uses to which the water is put (hydropower generation, flow regulation, inter-basin transfers etc.) can seriously affect the structure of fluvial invertebrate populations and communities (Ward & Stanford, 1983 a and b; Cerghino & Lavandier, 1996; Gibbins et al., 1996).

4.2.3 Substrate

Very few species or groups of macroinvertebrates live in one type of substrate, although many have a clear preference for one type or another. Such cases are generally due to the specific requirements associated with feeding habits, respiratory needs or shelter (Minshall, 1984; Ward, 1992). Some of the species with marked preferences are those showing specific morphological adaptations. This is the case with fauna considered as lithopholous, characterised by different species of Blephariceridae, Dixidae, Heptageniidae, Rhyacophila, Glossosomatidae, Simuliidae, Perla and Dinocras, or species of Taeniopteryx y Gripopterygidae, which live amongst mosses, or burrowers which live in gravel (many species of Leuctridae), in sand (Ephemera danica, Ephoron virgo, Choroterpes sp., Thraulus sp. and many Capniidae) or in mud (Caenis rivolorum) (Elliot, 2002).

In theory, at community level and considering only inorganic substrates, maximum species diversity should occur in zones where medium-sized substrates predominate since it is generally true that as the median particle size of sediments increases, their physical complexity also increases (Hynes, 1970; Minshall, 1984). However, two concept must now be introduced: substrate stability and heterogeneity. The former understood as the temporal persistence of one type or size of substrate on the riverbed and the latter as relative substrate diversity of different particle size ideally covering different surface textures, especially of coarse inorganic substrates. Both these concepts are essential for evaluating the importance of substrates in lothic ecosystems at community level, whether species richness, diversity, abundance or biomass is being assessed.

Although it has not been possible to demonstrate in experimental conditions that some community descriptors such as species richness show higher values in substrates of mixed size than in those of a uniform size unless variations in associated organic components (detritus or leaves, for example) are introduced (Wise & Moller, 1979; Erman & Erman, 1984). In normal conditions, rivers do not lack these organic components since greatest substrate heterogeneity is usually associated with great diversity of the organic substrate retained by or among inorganic substrates. In such

diverse conditions, the abundance and biomass of macroinvertebrate communities is greater (Ward & Dufford, 1979; Dudgeon, 1982).
However, it has been shown that the relatively more stable substrates of a reach or area are usually inhabited by a greater number of species (Townsend et al., 1997) and can bear a greater biomass per unit of area (Benke et al., 1984).

In the case of large organic substrates, whether filamentous algae like Cladophora sp. , mosses, macrophytes or tree trunks, we know that they are used by macroinvertebrates as substrates rather than food sources. It has also been demonstrated that such relatively large use as substrate by macroinvertebrates when are the most stable structures in riverbed (Benke et al., 1984) or act as shelter thanks to their damping effect on the current (Dodds & Biggs, 2002).

When considering whether a given species prefers one type of substrate or another, it must be borne in mind that it may change its preference according to its stage of development. For the same insect which shows an aerial stage, one must consider the substrate used by the aquatic larvae, the pupae (when applicable) and the egg-laying zones used by the adults. Much information is available to support such changes in preference in the same species. For example, the pupae of many Pyrenean species of Rhyacophila choose larger substrates than the larvae of the same species, and it is frequent to find substantial aggregations of pupae on boulders, while larvae tend to be found on cobbles.

Temporal changes, both daily and seasonal, may also occur. Let's not forget that many macroinvertebrate species are nocturnal and the substrate used as shelter during the day may well differ from that used for night time feeding.

4.2.4 Temperature

River systems are characterised by changes in temperature, usually daily and seasonal, with clear differences according to the climatic conditions of the drainage basin concerned. Different reaches may have a temperature that depends on the degree of cover offered by riverbank vegetation. Daily variations may be minimal and seasonal changes may be mitigated by water supplied from groundwaters, especially in karstic areas.

The regularity of temperature patterns in temperate rivers has led to some generalisations being made, such as "the extent of diel temperature variation usually is greatest in streams of intermediate size" (Allan, 1995), an affirmation which treats the distribution of maximum temperature ranges as a function of stream order (Vannote & Sweeney, 1980). Temperatures may range from 0 to 25ºC in temperate rivers, -2 to 18ºC in cold temperate rivers and 4 - 36ºC in the rivers of semiarid Mediterranean regions.

There is a longitudinal variation associated with altitude in most rivers, whereby the macroinvertebrates living in the higher reaches are adapted to the colder temperatures.

Such is the case with a large number of stonefly species, species of caddisflies comprising the family of Brachycentridae, Drusinae and Goeridae, and some genera of mayflies (Acentrella, Nigrobaetis, Rhithrogena). Other species show a clear thermal barrier in their upper height distribution associated with relatively low limiting temperatures; in other words they are stenothermal warm water species. An example of such species are the mayflies of the genera Thraulus and Choroterpes. In general, we have a good overall knowledge of the macroinvertebrate altitudinal distributions associated with temperature in different geographical areas (Berthelemy, 1966; Ward & Berner, 1980; Ward, 1982) and their classification as regards the degree of tolerance of temperature ranges (Tachet, 2000).

The importance of the temperature regime in the composition, structure and functioning of communities of fluvial benthonic macroinvertebrates is evident when changes in the general pattern by, for example, flow regulation. The way in which a reservoir is managed has a strong influence on the temperatures of the water downstream. In many cases a river will receive hypolimnetic water from a reservoir, raising the temperature range in winter and lowering it in summer (Ward & Stanford, 1979; Petts, 1988). In such cases the community composition may change with the appearance of cold water stenothermal species which normally live in the higher parts of the basin (Cereghino & Lavandier, 1996), the disappearance of summer developing species and a reduction in the number of annual cycles of many species of mayflies and caddisflies. Furthermore, the populations of the new species colonising these stretches may show different life cycles from those they normally show in their natural habitats in the same basin (Cereghino & Lavandier, 1996), with a consequent isolation of the populations. To such situations, one must bear in mind the case of reservoirs destined for hydroelectric power production, when there may be a complementary weekly change in downstream temperatures if power is only produced on weekdays but not at the weekend. When this happens the minimal flows which occur at the weekend mean that temperatures will be lower than the naturally expected ones and the winter temperature range will be broader, with a corresponding decrease in the number of species, population density and biomass.

4.3 Habitat

As we have already mentioned in the case of the river bed current velocity and the relative particle size of the substrate, the interdependence of some factors is evident. However, it is also necessary to introduce some structural factors, such as depth, when explaining macroinvertebrate distribution. For example, in some zones with an identical substrate type composition and velocity, the presence of some filtering species is dependent on depth. A clear example of this is the distribution of the larvae of Simuliidae, which live very near to the surface limit layer, that is in very shallow areas. It is therefore preferable to analyse species distribution at the habitat level, or even microhabitat level in the case of macroinvertebrates if the study concerns only few species as opposed to the community as a whole.

Several revisions have been made on the application of models of habitat use in population ecology, relating diversity or the heterogeneity of the habitat with species distribution (Hart & Horwitz, 1991; Palmer & Poff, 1997). Such models frequently include the effect of different types of biological interactions, such as interspecies competition (Hart, 1983) to discriminate criteria of habitat use as a function of the population densities analysed.

However, the regulation of river flows, a common practice world-wide but particularly so in our countries, has led to the need to design a more applied approach to understand, and predict the use of the microhabitat by populations, which, at least, permits us to conserve the structure and functioning of a basic biological community proper to each stretch. Such a perspective has take on a new importance since the EU Water Framework Directive has come into operation.

From this last applied perspective, an integrated approximation which initially only contemplates physical and structural variables as descriptors of changes in species, whether considered at the level of density or biomass (Stalnaker et al, 1994). For the application of the múltiple attribute standard-setting methods different types of tools can be used. The most common in the USA is the Physical Habitat Simulation System (PHABSIM)" (Bovee, 1982), which relates open channel hydraulics with known elements of the fish and macroinvertebrate environment. This knowledge leads initially to the construction of response surfaces for pairs of variables, such as velocity and depth, turbulence and substrate, for example, (Gore, 1987) and habitat preference curves (Gore & Judy, 1981; Gore, 1987). From these descriptors it is possible to estimate "the weighted usable area" (WUA) for a species in a stretch as a function of flow (Gore, 1987).

Another option are the complex hydraulic models that consist of using methods to estimate complex hydraulic key characteristics and their relation with the distribution and densities of macroinvertebrates (Statzner et al., 1988). Within this type, models for estimating hydraulic habitat suitability curves for macroinvertebrate species have been developed, making it possible to evaluate hydraulic habitat quality and its availability for a given species (CASIMIR model, Jorde 1996).

The initial problem with this type of method lies in the fact that estimates and predictions that affect the whole community are made from analysing the optimal curves of very few species, in some cases one only. This has led to two different options being considered. The first, if conservation of the river ecosystem is given weight, resides in working with sensitive species, whose populations decrease rapidly with small changes in habitat quality or availability (Gore, 1987). The second chooses characteristic species or indicators of the community present in a given stretch, which also show optimal densities in the same area (Statzner et al., 1988). Usually, for both strategies, we find ourselves with a wide variety of species which fulfil the first requirement and so the final selection will give preference to the most easily identifiable in all their instars and which inhabit upper layers of the river bed (Resh, 1979). However, the process of selecting species can be made more objective from the outset if different statistical treatments are used to determine groups of species which characterize the stretch (Gibbins et al., 1996).

The large number of samples necessary to apply with a minimal degree of predictability any of the above methods have meant that models of macroinvertebrate habitat have been little used in river management. Furthermore, the habitat preference curves estimated for one species in a given stretch cannot be extrapolated to other communities of which this species forms a part. However, it has been demonstrated that they should be included in certain systems to complete the information provided by fish populations (Gore et al., 2001), especially in warm-water rivers and those of arid and semi-arid areas, where fish normally occupy pools but whose principal source of food, the macroinvertebrates, inhabit riffles. In these systems, at least, the habitat preferences of macroinvertebrates and how changes in the availability associated with variations in flow may affect their densities and biomass, should be studied. Moreover, such preference curves are now being used at more generic levels for groups of taxa which indicate water quality, such as the EPTs (Ephemeroptera, Plecoptera and Trichoptera), or which use community diversity (Gore et al., 2001).

4.4 Relation between physical and biological indicators

The obvious question to be asked in an applied context is whether e.g. physical degradation is manifested in the macroinvertebrate indices commonly used in water quality assessment. The majority of indices have been developed mainly to detect the effects of organic pollution (e.g. Metcalfe-Smith, 1994) but are widely used to detect general degradation of streams and rivers. However, very few studies have tried to single out the effect of habitat degradation on commonly used macroinvertebrate indices. Olsen & Friberg (1999) investigated 6 small Danish streams to assess the importance of physical features for the Danish Stream Fauna Index (DSFI) value. DSFI is the national biomonitoring standard in Denmark (Skriver et al. 2000). In each stream, two stream reaches that differed with respect to physical heterogeneity were sampled. The reaches did not differ with respect to chemical features, e.g. BOD, nor were there any dispersal barriers for the macroinvertebrates. In 5 out of 6 streams investigated, the DSFI value was higher (indicating better ecological quality) in the heterogeneous reach compared with the homogeneous reach. This study indicates that physical features alone can influence the assessment of ecological quality using macroinvertebrate indices.

Several countries have developed specific indices to assess the physical (habitat) quality of streams and rivers. The most internationally well-known example is probably the River Habitat Survey (RHS) which is the national physical stream monitoring method in the UK (Raven et al., 1998). In future, hydromorphological quality elements are going to be an integrated part of the Water Framework Directive, which will be the basis for a Pan-European legislation on water. Standardisation of habitat assessment methods is currently undertaken by CEN and a draft standard has been developed (CEN, WG2, TG5). Evaluation of physical quality has a dual purpose, namely to directly quantify the physical quality for its own sake as well as being an integrated component of ecological quality assessment. The latter is especially important in the context of this paper as it can provide causality between physical degradation and changes in macroinvertebrate communities. Nevertheless, published results using the RHS and a macroinvertebrate biotic index (BMWP/ASPT) have failed to show a clear relationship. It is therefore alarming that a number of methods may be developed and implemented that are only vaguely related to the invertebrate communities. This concern also applies to the rest of the biota such as fish and macrophytes. The failures of establishing the link between physical features and the biota could limit the ability of determining the main factors responsible for an unsatisfactory ecological quality in a given stream reach. This "missing link" will potentially fail to provide the necessary and relevant information to support decision-makers.

Another possibility is to develop macroinvertebrate metrics directly targeting physical impacts. The US EPA (Barbour et al., 1999) have in their multimetric approach one metric which directly indicates physical degradation. Another approach is multivariate methods, such as canonical correspondence analysis, which relate community structure to e.g. the strength of various physical vectors (e.g. TerBrak and Verdonschot, 1995). These methods are widely used both in assessment studies and less applied investigations.

There is an apparent need to develop more metrics that target physical degradation, and/or to test existing metrics (or various multimetric combinations) for their ability to detect physical changes. However, two projects financed by EU 5th Framework Programme are addressing the question of how macroinvertebrates indicate various stressors, including habitat degradation (www.STAR.au; www.AQEM.de)

4.5 Future perspectives

From the state-of-the-art knowledge it is obvious that several essential questions are presently unresolved. These questions need to be addressed to support the future development of aquatic models. The questions include:

- The importance of physical features for the biological structure in lowland streams – are lowland system less structured by physical factors and do biotic interactions consequently play a key role?
- The interactions between biota (including macroinvertebrate communities) and the physical environment across scales - what scale is most important for understanding physical-biological interactions and how valuable are small-scale experiments for our ability to predict interactions at a higher scale?
- The use of restoration projects in our understanding of the relationships between habitat features and the biota – what lessons have been learned from the large number of restoration projects conducted? General trends across temporal and spatial scales.
- How do we integrate the assessment of physical features into the monitoring of ecological quality and what data should be the input for future models? Can we provide a direct and predictable (using models) link between the quality of the physical environment and e.g. macroinvertebrate community composition? End

users need to be able to quantify the ecological consequences of e.g. habitat degradation to make sustainable decisions.

5. QUALITY ASSURANCE AND QUALITY CONTROL (QA/QC)

5.1 Training and accreditation

Every time that an environmental problem is detected some sort of click mechanism start, and stakeholders claim for environmental assessment to find solutions. In other cases it is not necessary the existence of a special problem, and by law it is obligatory do periodical environmental assessment. In both cases there are two boxes or interactive parts that intervene in the process: box A represented by scientist and environmental technician and box B represented by environmental managers (see Fig.5.1).

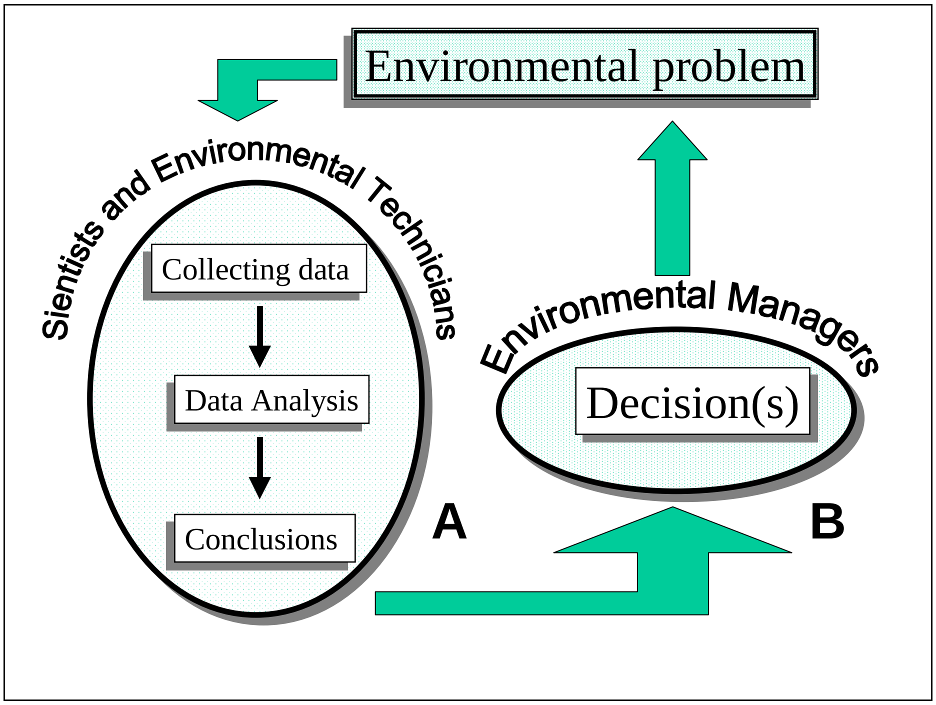

Fig. 5.1: A simplified illustration of an environmental assessment process and implied elements.

Once an assessment strategy has been decided the first steep is to collect data, mostly applying sampling protocols (i.e.: collecting macroinvertebrates, assessing the habitat quality, etc...). The elaboration of samples generates the data, and the data analysis is the base to elaborate "good" conclusions in which to support "good" managing decisions. Thereafter, if box boxes work properly, everything will run properly and the effort and the results worthwhile. However, more often than desired it doesn't occur due to a bad quality of the data that invalidates or makes less efficient all the process. This is a non new but very important topic as was pointed out by Stevenson *et al.* (in press).

To be able to have reliable data it is necessary to collect them properly. For instance if along the assessment procedure macroinvertebrates have to be collected it is important that the operator(s) do it properly (following the appropriate protocol) so that later on the lack of some taxa, or the difference in abundance should respond to a real situation and not to the expertise of the operators, or to sampling effort. This is particularly important when different sites are to be compared. An example of this appears represented in Fig.5.2. This corresponds to an exercise conducted along a stream of the Sierra Nevada Mountains in south of Spain, participating students with different backgrounds expertise when they were attending a course on macroinvertebrates and pollution. Their student profiles were as follow: a biologist with a year experience working on biomonitoring by using macroinvetebrates (Operator 1), a biologist with experience in Limnology, actually doing a PhD with lakes but not familiar with watercourses (Operator 2), a terrestrial entomologist (Operator 3), a biologist dealing with toxicology not used to work outside the lab (Operator 4), and a chemist without experience on field work (Operator 5).

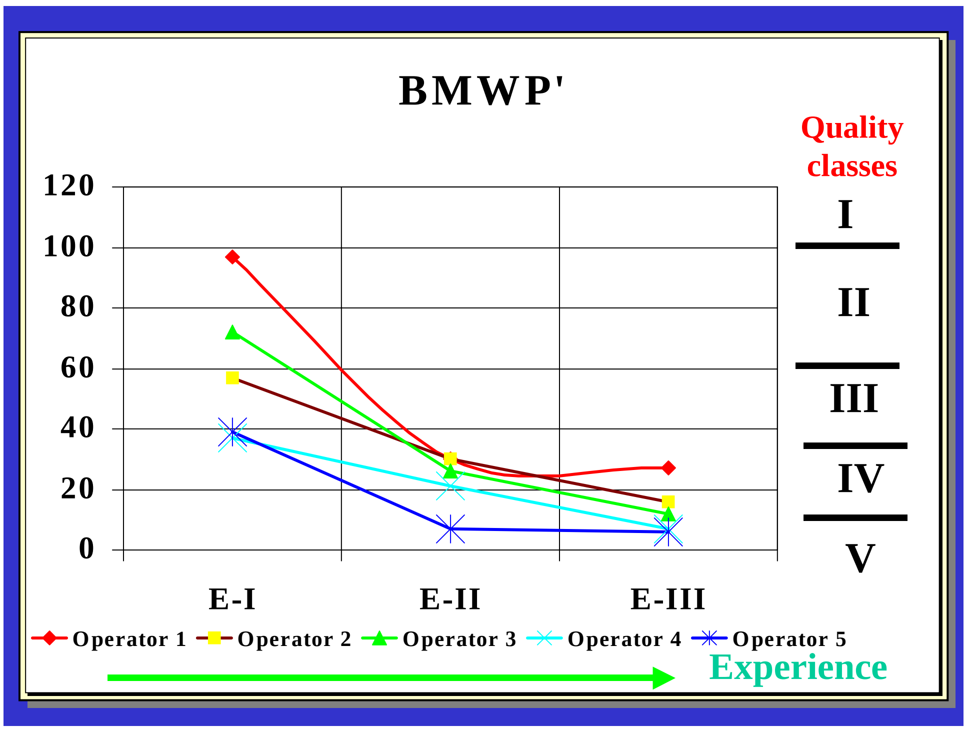

Fig. 5.2: Example of results obtained by different experienced operators assessing water quality along a stream (for more information see inside text).

Previously to the experience they have received an explanation of how to sample macroinvertebrates to follow the protocol to apply a biotic index termed BMWP’ (Alba-Tercedor & Sánchez-Ortega, 1988), a Spanish adaptation of the British BMWP (Armitage *et a*l., 1983), following the protocol described by Alba-Tercedor (1996, 2000). It is clear the different results obtained in accordance with the experience. All of them detected the same trend in a losing of water quality downstream, but with very different scores, so that clear dysfunctions appeared when scores were translated to water quality classes’ significance.

To avoid these dysfunctions it is very important that in any assessment design should be included training part to ensure the appropriate application of methodologies, permitting the comparison of data. The importance of training to reduce operator variability al training but especially practical training along a enough period of time to permit operators to get reliable data. The importance of training to reduce operator variability is not a new task and has been reported in different studies (i.e.: Hannaford et al. 1997, Rankin 1995, Resh 1995). On this respect we agree with Resh (1995) when pointed out that special training of personnel is needed in a few key area including: (1) how to chose study sites and controls and (2) how to conduct standardized sampling. And the later includes training on: (a) criteria to sampling, (b) how to keep samples (include labelling, preserving and transport), (c) elaboration and analyses of samples, and (d) safety regulations to follow during the field procedure and laboratory. Many of these topics and field quality assurance protocols can be found in the recent publication of the EPA (Barbour et al. 1999).

Training in safety worthwhile and avoid many risk factors. Thus, for instance, it is important for the fieldwork train in: (1) know how to conduct in fast sweep water, (2) to use gloves (and/or protective cream), not only to manipulate preservative products, but also for sampling, (3) to be sure that operator don't touch their mouth or faces after they have been manipulating into non pristine waters. For the laboratory it is important that operators follow the normal safety rules to work in the lab, especially these interested in manipulations of the chemical that they use and how to conduct in case of an accident. Both for the field and the lab purposes it is important a training on first aid and CDR, and to contact with local health group when prospecting non-familiar areas.

Once it is ensured that operators have been trained ad have the necessary expertise to follow appropriate procedures, the next important step will be to test all the assessment process by an external Quality Assurance/Quality Control program (QA/QC). Its purpose is to ensure that methods for data collection are standardised, that are of consistent and high quality, and that the quality is maintained throughout the duration of the assessment project. There are numerous examples o quality controls programs to refer to in the USA (i.e.: Barbour et al 1999, Cuffney et al. 1993, Drouse et al. 1986, Moulton et al. 2000), and Europe as well (i.e.: van Dijk 1994, Dines and Murray-Bligh 2000).

5.2 Data base set-up and maintenance

Data management can be viewed as a process that begins with the conception and design of the research project, continues through data capture and analysis, and culminates with publication, data archiving and data sharing with a broader public (Michener & Brunt, 2000).

The design of an effective data management system depends on considerable forethought and planning to meet and balance several fundamental requirements or objectives. The primary goal of a data management system is to provide data of a requested quality (data reliability, amount of missing values, …) within a reasonable budget. A second system requirement is facilitating access to data by investigators. An important related issue is providing short-term and long-term security for data through data archiving. Archival storage involves various activities that are designed to protect the data against information fuzzyfication and loss. A data management system may therefore have the following components or activities (Michener & Brunt, 2000):

- an inventory of existing data and resources will have to be compiled and priorities for implementation be set;
- data will have to be designed and organized by establishing a logical structure within and among data sets that will facilitate their storage, retrieval and manipulation;
- procedures will be required for data acquisition and quality insurance and quality control;
- data set documentation protocols, including the adoption or creation of metadata content standards and procedures for recording metadata, will need to be developed;
- procedures for data archival storage as well as maintenance of printed and electronic data will have to be developed;
- an administrative structure and procedures will have to be developed so responsibilities are clearly delineated.

Ecologists can avoid many potential difficulties in field sampling and subsequent data analyses if sufficient thought is given to designing data sets prior to collecting data. Therefore a preliminary sampling and/or information research is often useful to set up a definitive intensive monitoring campaign. Some decisions about data design are necessary before data are collected in order to produce field and laboratory data sheets. The completed design can be transferred directly to data entry tools to aid in data collection, to facilitate analysis by statistical software and to support metadata development and structure the data set for archiving (Goethals, 2002). Fig. 5.3 illustrates a simplified database system (Elmasri & Navathe, 2000).

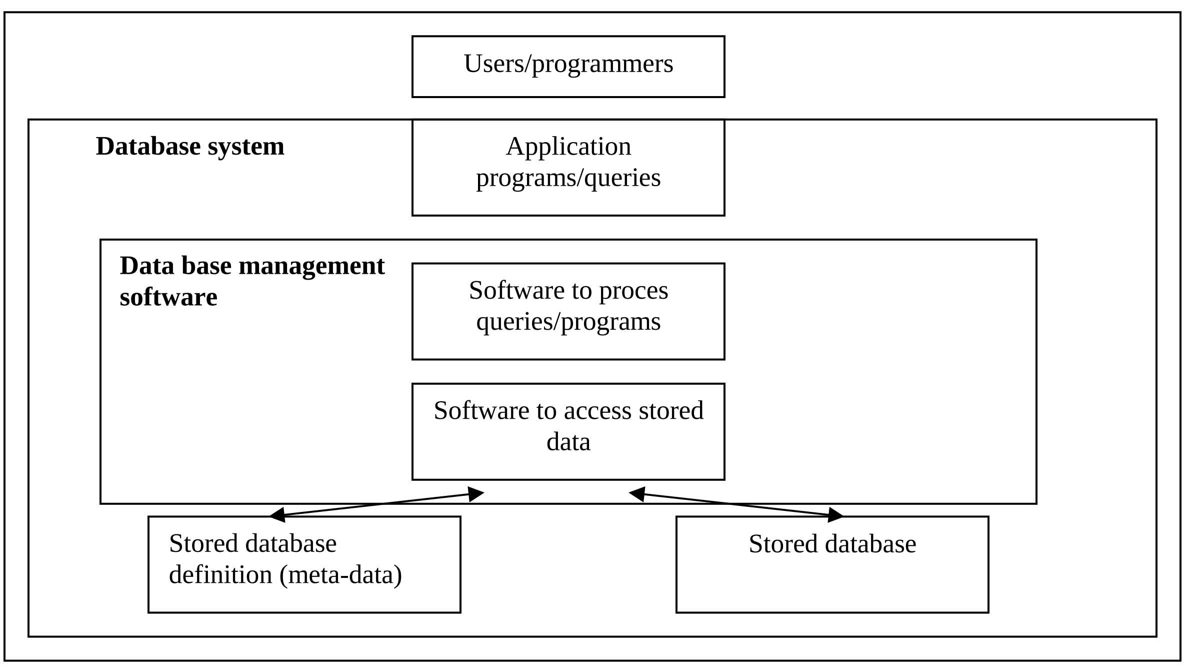

Fig. 5.3: A simplified database system environment (Elmasri & Navathe, 2000).

6 CONCLUSIONS

Within the European Community, a lot of different methods and standards exist to collect macrobenthos. This often causes difficulties to develop, train and validate models describing macroinvertebrate communities in different countries. Further research is therefore needed which methods are most convenient to collect data for national and international purposes, and which methods are most convenient for a particular scale of use.

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8 APPENDIX

8.1 Identification literature

8.1.1 General literature

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8.1.3.13 Lepidoptera

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8.2 European projects

8.2.1 AQEM

The Development and Testing of an Integrated Assessment System for the Ecological Quality of Streams and Rivers throughout Europe using Benthic Macroinvertebrates.
Acronym: AQEM
Contract No. EVK1-CT-1999-00027
Duration: March 2000 to February 2002
AQEM has been a research project under the 5th Framework Programme of the European Union.
The AQEM consortium has developed an assessment system for the Ecological Quality of European streams based on benthic invertebrates.
http://www.aqem.de

Aims of the AQEM system are:

- to classify a stream stretch in a quality class from 5 (high) to 1 (bad) based on a macroinvertebrate taxa list
- to give information about the cause of a possible degradation to help direct future management practices.

In contrast to many other comparable projects, the development of the AQEM system has been based on
a new dataset covering both the fauna and general stream characteristics of 28 common European stream types.

As demanded by the Water Framework Directive AQEM applies a stream type-specific approach. Particularly at the European scale this is inevitable, since e.g. an highland stream in Sweden and a lowland stream in Italy are inhabited by very different macroinvertebrate communities. Therefore, for each stream type different calculation methods are applied based on the comparison with different reference conditions. However, the system always follows the same evaluation scheme and each stream-type specific method fits into the general assessment framework. This framework can be defined as follows:

Stressor-specific approach; for each stream type the "main" degradation factor presently affecting the stream is assessed. This might be acidification (e.g. in Northern Sweden), degradation in stream morphology (e.g. in Central Europe) or organic pollution (e.g. in Southern Europe). In some cases more than one stressor is separately assessed and the results of the individual steps are then combined to a final assessment result or the assessment is addressing the "general degradation".

Multimetric system; for each stream type those calculation methods have been identified, which are best at indicating a sites' state of degradation. The results of the individual calculation methods are then combined in a "multimetric formula", which is always the same.

The multimetric result is converted into the final score ranging from 5 (high quality) to 1 (bad quality).

8.2.2 STAR

Standardisation of River Classifications: Framework method for calibrating different biological survey results against ecological quality classifications to be developed for the Water Framework Directive
The Star project is a research project supported by the European Commission under the Fifth Framework Programme and contributing to the implementation of the
Key Action "Sustainable Management and Quality of Water" within the
Energy, Environment and Sustainable Development Programme.
Contract No: EVK1-CT 2001-00089
http://www.eu-star.at/frameset.htm

Data will be used to answer the following questions, which form the basis of a conceptual model:

- How can data resulting from different assessment methods be compared and standardised ?
- Which methods/taxonomic groups are most capable of indicating particular individual stressors ?
- Which method can be used on which scale ?
- Which method is suited for early and late warnings ?
- How are different assessment methods affected by errors ?
- What can be standardised and what should be standardised ?

For the purposes of this project two '**core streams types**' are recognised: small, shallow, upland streams and medium-sized, deeper lowland streams. Besides the evaluation of existing data, a completely new data set is sampled to gain comparable data on **macroinvertebrates**, **phytobenthos**, **fish** and **stream morphology** taken with a set of different methods from sites representing different stages of degradation. This will be the main source of data for cross-comparisons and the preparation of standards. A number of '**additional stream types**' will be investigated in order to extend the range of sites at which field methods and assessment procedures are compared. The participants will be trained in sampling workshops and quality assurance will be implemented through an audit. Using the project database, assessment methods based on benthic macroinvertebrates will be compared and inter-calibrated, particularly in terms of errors, precision, relation to reference conditions and possible class boundaries. The discriminatory power of different organism groups to detect ecological change will be tested through various statistical procedures.

8.2.3 PAEQANN

Predicting Aquatic Ecosystems Quality using Artificial Neural Networks: impact of environmental characteristics on the structure of aquatic communities (algae, macrobenthos and fish)

The PAEQANN is a research project supported by the European Commission under the Fifth Framework Programme and contributing to the implementation of the Key Action "Sustainable Management and Quality of Water" within the Energy, Environment and Sustainable Development.
Contract n°: EVK1-CT1999-00026

Duration: March 2000 to February 2003

http://www-cesac.ecolog.cnrs.fr/~paeqann/index2.htm

The **aim** of this project is to develop methodologies which allow: i) to provide predictive tools that can be easily applied to define the most effective policies and institutional arrangements for resource management; ii) to apply the most effective and innovative techniques (mainly goal function and artificial neural networks) to identify problems in ecosystem functioning, resulting in ecosystem degradation from human impact, and to model relevant biological resources; iii) to fully exploit existing information, reducing the amount of field work (that is both expensive and time consuming) that is needed in order to assess freshwater ecosystems health; iv) to explore specific actions to be taken for restoration of ecosystem integrity; and v) to promote collaboration among scientists of different interested countries and research fields, encouraging collaboration and dissemination of results and techniques.

The main **applied objective** is to propose a set of tools for water management and water policies in order to allow to easily assess ecological quality and perturbations of stream ecosystems. These tools will provide information about running water quality as well as community structure. The assessment tools will allow to identify measures which should be taken to restore biological integrity in running waters. Hopefully, the study can be considered as a first step toward linking the improvement of water quality through specific management measures (e.g. waste water treatment, habitat restoration, etc.) with the expected improvement in ecological and biological value of running water systems.

The main **scientific objectives** will be i) setting up a standardised methodological approach (we have defined a set of technical procedures which will be used in a common framework to analyse or to predict community structure of studied ecosystems; each reference site is sampled in a standardised way and this will allow to compare the different sites for regional conservation priorities); ii) linking the environmental characteristics and the community structure at each reference site by using a defined set of parameters and a combination of target groups representing the main functional levels of the ecosystems (rapid assessment procedures will be implemented on these hypotheses that regulative and functional factors, the resources describe ecosystem functioning in a unifying way); iii) evaluating at a functional level the sensitivity of the studied ecosystems and their response to disturbance through implementation of sensitivity indices and modelling (the main threats on living communities and on local endangered species will be identified as we shall build predictive models of community structure for a set of critical habitats); and iv) investigating the effects of human impacts on the functioning of the ecosystem, i.e. on the composition and change in structural and functional organism groups in comparison to nearby natural reference conditions. Special attention is directed to summarising ecosystem functioning by exploring the chance of community restoration in selected sites submitted to the most common types of disturbance.

8.3 Information and databases

Websites with databases on:

- Measurements
- Relations (autecological and synecological relations)
- Geographical distribution (maps)

8.4 Results of the COST 626 questionnaire on macroinvertebrates

8.4.1 Aim of the questionnaire

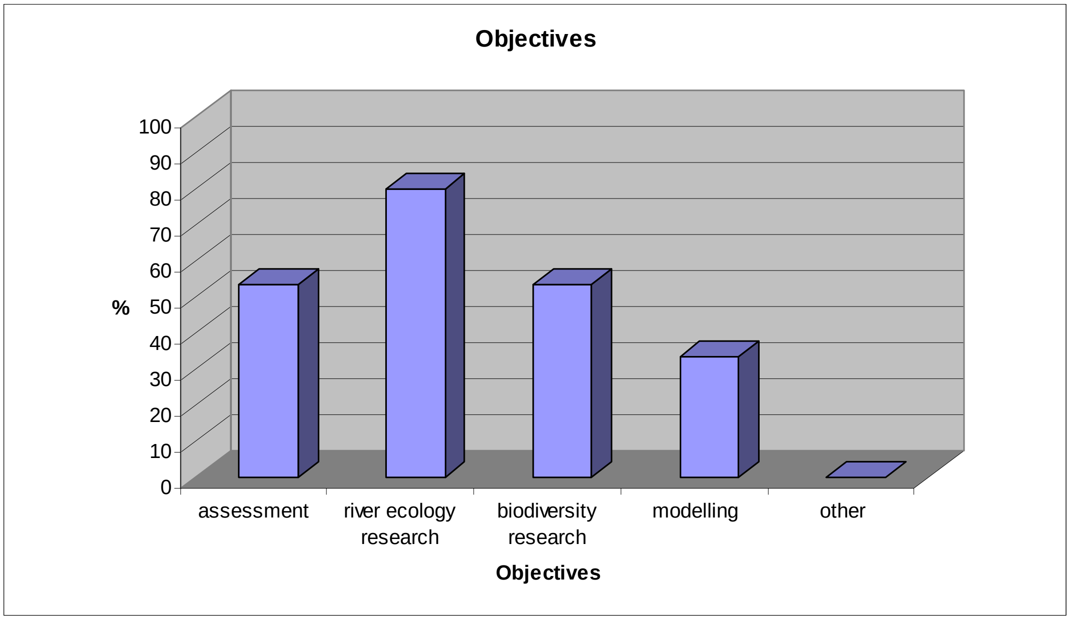
The aim of this questionnaire was to collect information on the de sampling techniques used by the different member states involved in the COST 626 network. Also questions on the use of the data, the data management and research needs were part of this study.

8.4.2 Feedback classified per member state

In total 15 reactions:
3 : Spain and France
2 : Finland and Belgium
1 : Norway, Germany, Slovenia, Denmark, Luxembourg

8.4.3 Question 1: Objectives of the macrobenthos data collection

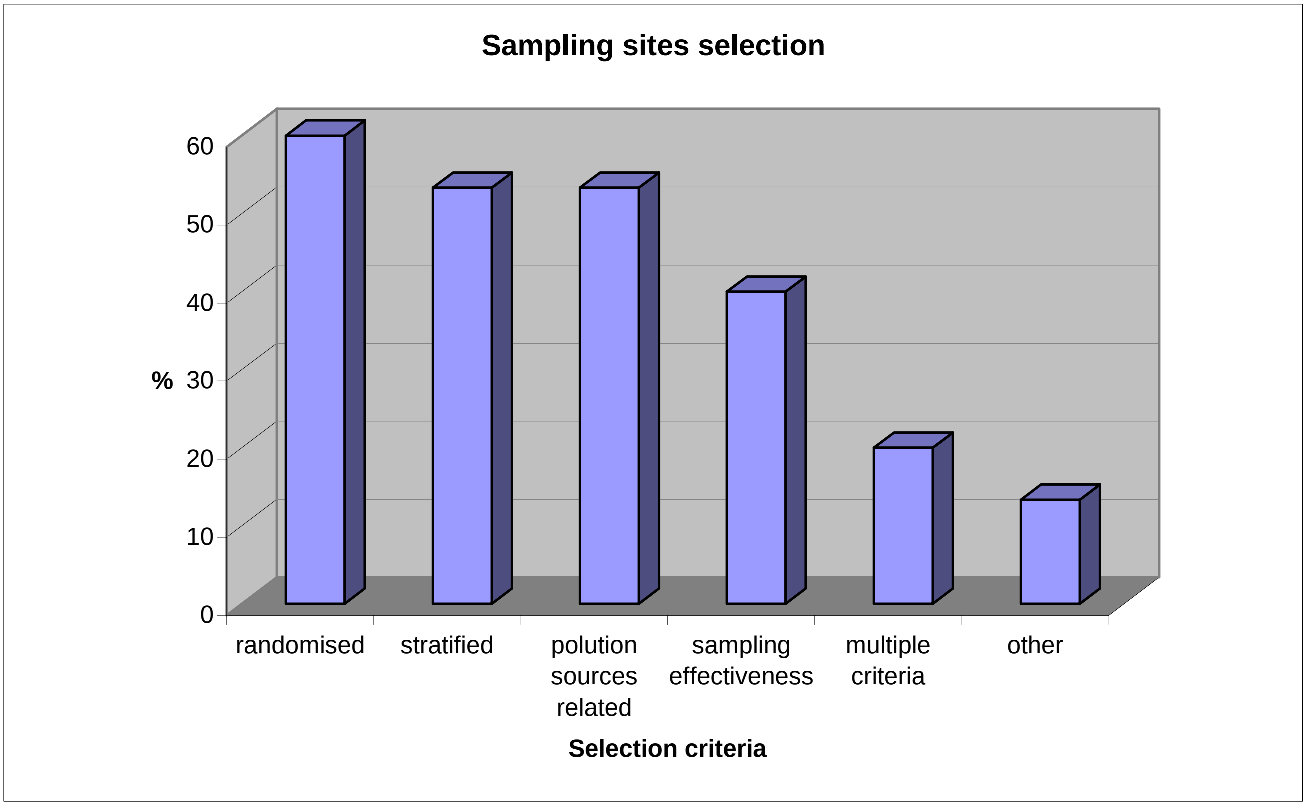
Are you using macroinvertebrate for?
a) Just assessing water quality
b) Study river ecology (including habitat assessment)
c) Study biodiversity
d) Modelling (specify what are your modelling aims)
e) other (specify)



8.4.4 Question 2: Criteria for sampling sites selection

What are the selection criteria for the monitoring sites?

a) randomised
b) stratified
c) pollution source located
d) both
e) sampling effectiveness
f) other



8.4.5 Question 3: Sampling protocols

Are you taking samples
a) Qualitative (specify methodology…)
b) Quantitative (specify methodology…)
c) Semi quantitative
d) All microhabitats
e) Selected habitats (i.e.: riffles….)
f) Other

8.4.6 Question 4: Training of the operators

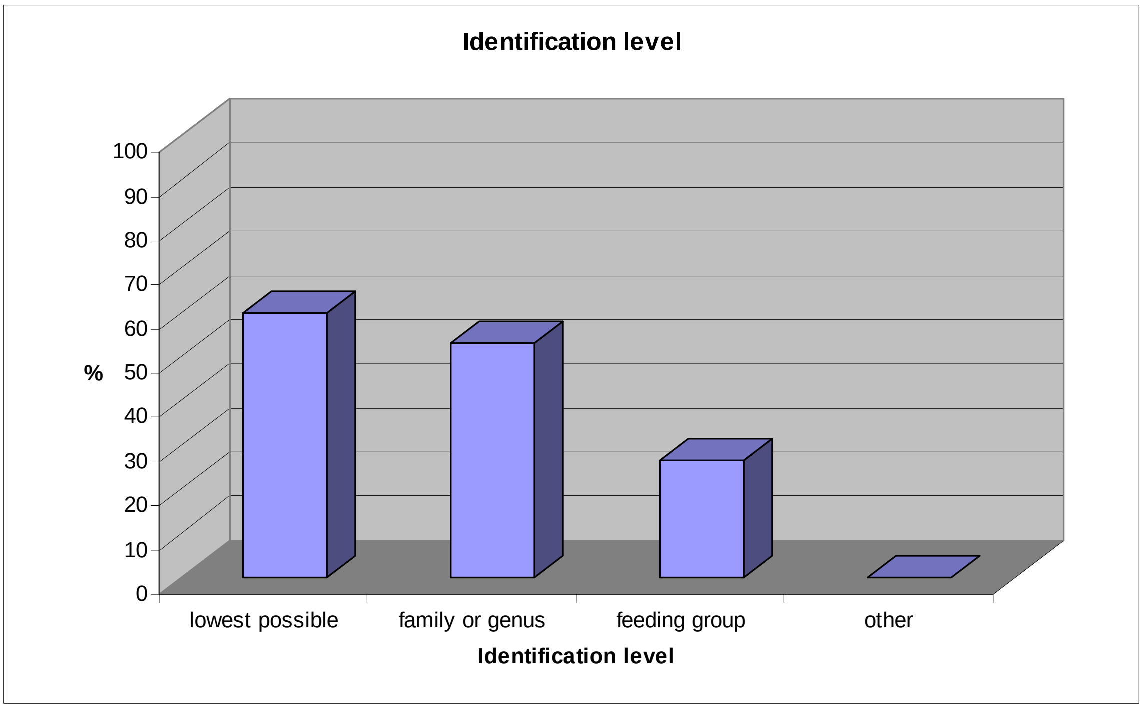
Are operators specifically trained for sampling and/or identify macroinvertebrates?:
a) Yes (please specify how..)
b) No

8.4.7 Question 5: Identification level

What is the identification level?

a) The lowest as possible (mostly species…)
b) Family or genus level
c) Feeding group
d) Other

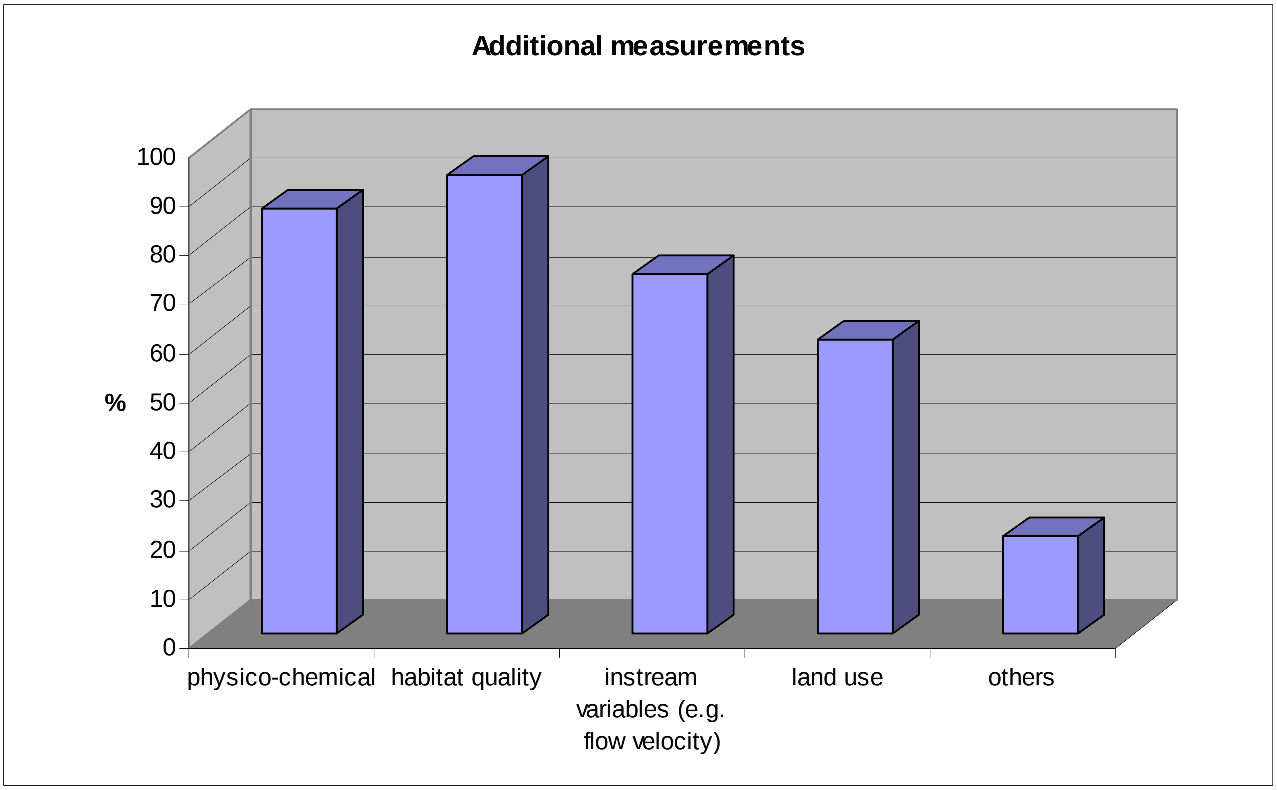
What bibliographic source do you use to identify?



8.4.8 Question 6: Additional measurements

Specify other physical, or habitats measurements:

a) Physico-chemical (DBO, pH, etc… specify…)
b) Habitat quality
c) Instream variables (depth, velocity, shear stress , etc… specify)
d) Land use
e) Other



8.4.9 Question 7: Data base set-up and use

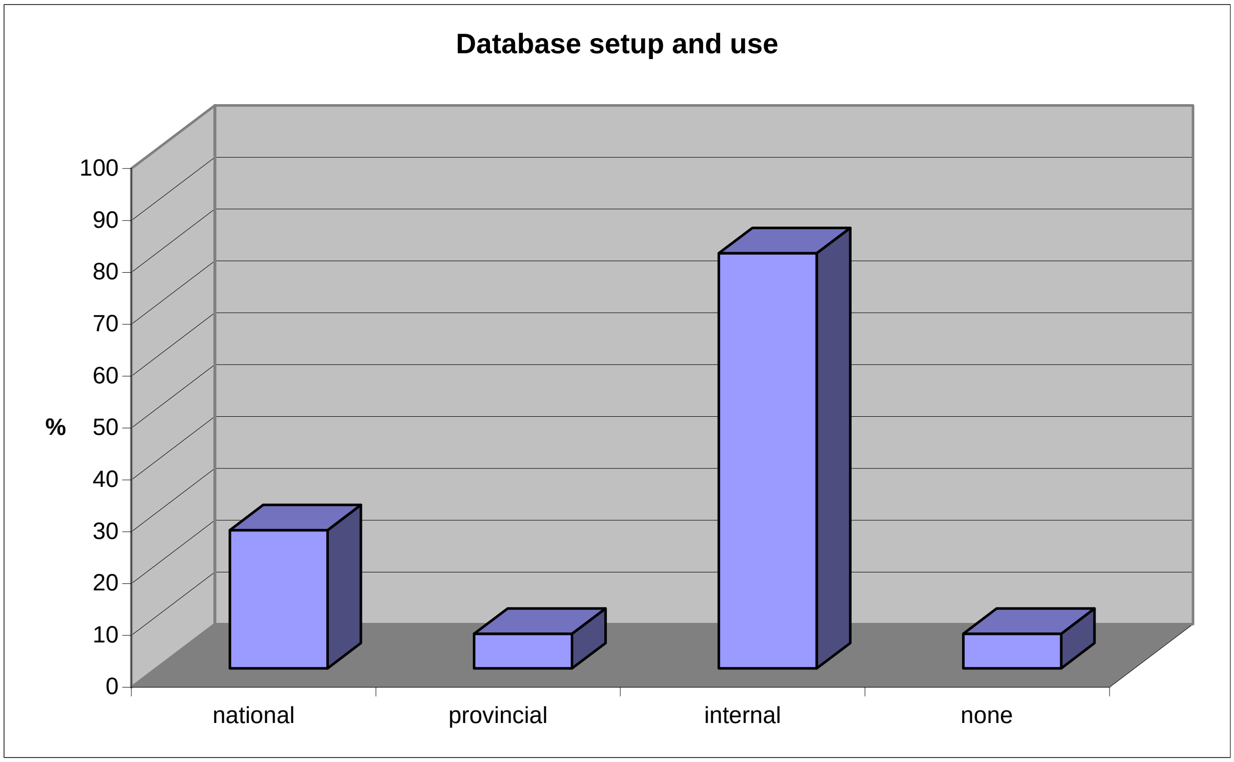
Do you maintain a data-base? (if affirmative, specify format and say if it is):

a) National
b) Provincial
c) For your own research group purposes

What the ecoregions the data-base covers?.

Are the data of public domain?

a) Yes (how?, i.e.: web site, river basin authority, etc.. ……specify)
b) No


Database setup and use
%
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80
70
60
50
40
30
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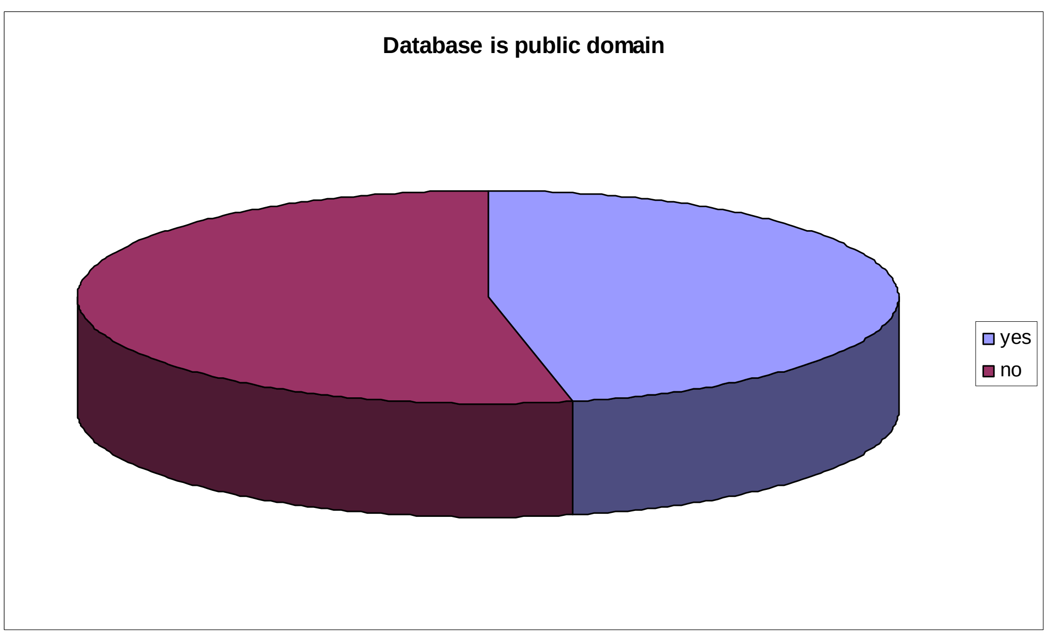
national
provincial
internal
none


Database is public domain
yes
no

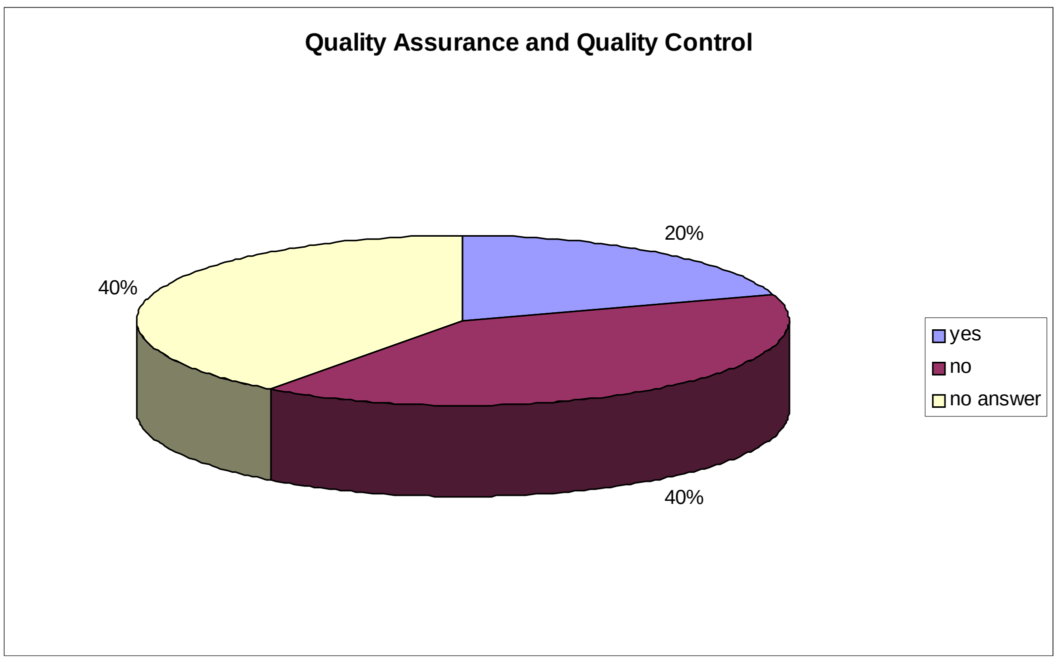
8.4.10 Question 8: Water quality assessment method

How do you assess the water quality ("Ecological Status") according with the new EU Directive?

Please specify method(s) and reference(s).

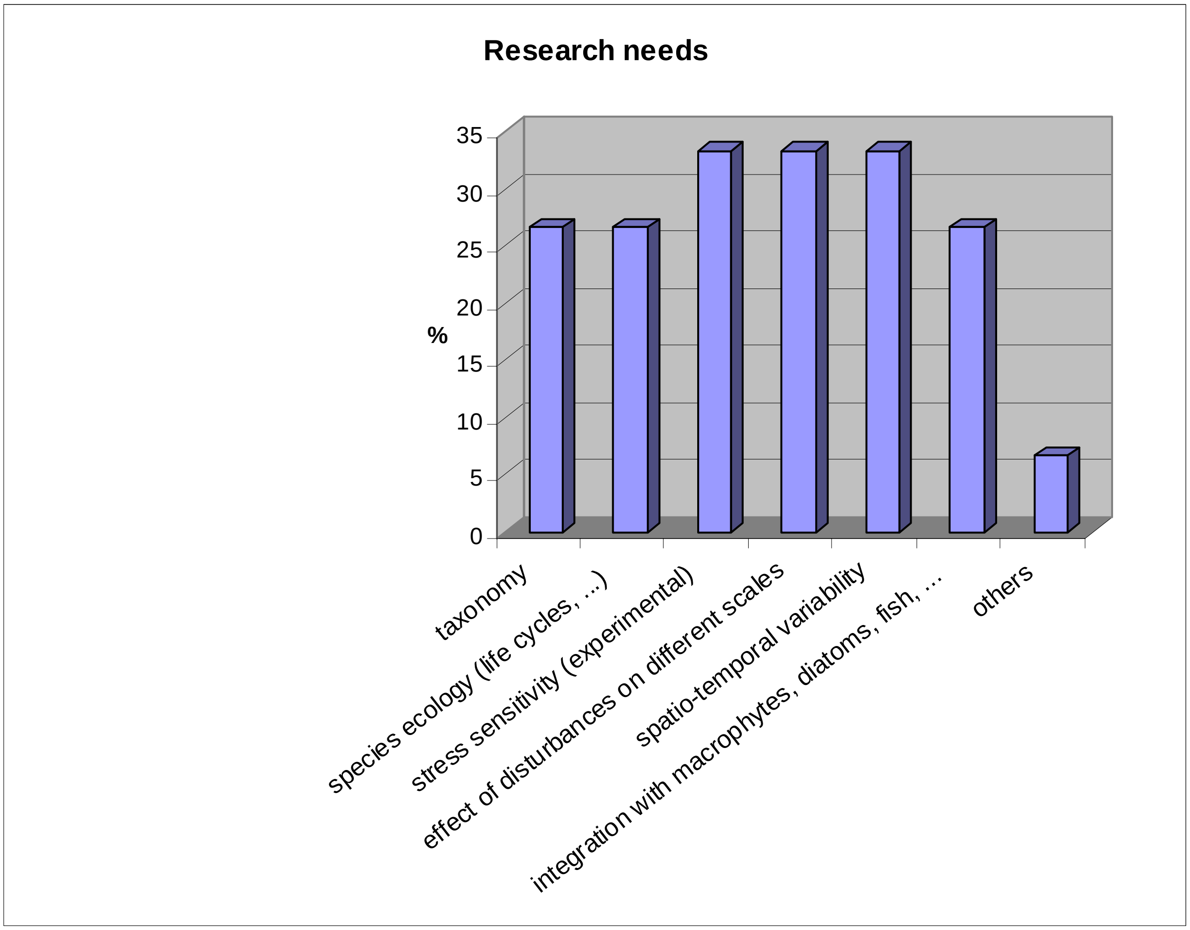
8.4.11 Question 9: Quality Analysis/Quality Control (QA/QC)

To verify the quality and reliability of your data did you follow any QA/QC protocols?. If affirmative, please specify how do you perform it?



8.4.12 Question 9: Research needs

a) Knowledge on taxonomy
b) Knowledge on species ecology (habitats requirements, factors affecting life cycles
c) Stress sensitivity based on experimental experiments.
d) The role of macroinvertebrates in upscaling levels when different disturbances occur.
e) Spatial and/or temporal variability and its implications on monitoring
f) Integration of diatoms-algae, with macroinvertebrate, macrophytes, fish, amphibians riparian-vegetations, birds, etc…to perform a good method ("index") to assess the "Ecological Status"



8.4.13 Conclusions of the questionnaire

From the questionnaire, the following conclusions could be made:

- Most data collections are done within the framework of ecological research
- The design of the monitoring network is mainly based on randomly chosen sites
- Macroinvertebrate identification is mainly done at the lowest possible level or at family/genus level
- Quite some physical-chemical and habitat measurements are combined with the macroinvertebrate samplings
- Most databases are merely used internal
- Quality control is rather limited