

Antti Kanninen

Aquatic Macrophytes in Status
Assessment and Monitoring of
Boreal Lakes



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Aquatic Macrophytes in Status Assessment and Monitoring of Boreal Lakes

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Antti Kanninen

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ABSTRACT

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Diss.

Water management legislation requires the status assessment and monitoring of freshwaters to be based on the changes in biota by anthropogenic disturbances. Despite long research traditions, the methods and metrics for ecological assessment and monitoring of boreal lacustrine macrophytes have not been well established and hence require evaluation. The first aim of this study was to evaluate two commonly used field survey approaches and remote sensing as macrophyte monitoring methods. Secondly, the best-suited metrics for status assessment were appraised. In addition, the responses of macrophytes to anthropogenic land-use in the catchment and to rehabilitation measures were studied. Macrophyte species were more cost-effectively detected by a phytolittoral species inventory than by using a transect method, making the former better suited for surveys of biodiversity and rare taxa. However, the ecologically important zonation of vegetation can be quantified with transects, favouring the latter for ecological assessment. Aerial photograph interpretation produced life-form-level, not taxonomically exact, information on macrophytes, making it useful for evaluating long-term changes of vegetation in response to management. A metric based on remote sensing data was found applicable for comparing macrophyte abundance among lakes and hence in bioassessment. The general taxonomic metrics were able to unify assessments across pressure gradients. Therefore, despite being more sensitive to methodological variation, conceptually sound general measures, rather than conventional stressor-specific indicator metrics, should be used as primary tools in the bioassessment of freshwaters. Status grading of macrophytes was in accordance with other present biota and biotic sediment records in showing significant changes in the ecological condition of a managed, eutrophic lake. Macrophyte metrics, especially general taxonomic metrics, showed a stronger response to land-use in the direct vicinity of the lake shoreline than in the whole catchment. This may have implications for optimal targeting of mitigation measures.

Keywords: remote sensing, survey methods, Water Framework Directive

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LIST OF ORIGINAL PUBLICATIONS

The thesis is based on the following original papers, which will be referred to in the text by their Roman numerals I-VI.

- I Kanninen A., Vallinkoski V.-M., Leka J., Marjomäki T.J., Hellsten S. & Hämäläinen H. 2012. A comparison of two methods for surveying aquatic macrophyte communities in boreal lakes: implications for bioassessment. *Aquatic Botany*, In press.
- II Valta-Hulkkonen K., Kanninen A., Ilvonen R. & Leka J. 2005. Assessment of aerial photography as a method for monitoring aquatic vegetation in lakes of varying trophic status. *Boreal Environment Research* 10: 57–66.
- III Valta-Hulkkonen K., Kanninen A. & Pellikka P. 2004. Remote sensing and GIS for detecting changes in the aquatic vegetation of a rehabilitated lake. *International Journal of Remote Sensing* 25: 5745–5758.
- IV Kanninen A., Hellsten S. & Hämäläinen H. 2012. Comparing measures of taxonomic composition and stress-specific indices in status assessment of lacustrine macrophytes. Submitted manuscript.
- V Alahuhta J., Kanninen A. & Vuori K.-M. 2012. Response of macrophyte communities and status metrics to natural gradients and land use in boreal lakes. *Aquatic Botany* 103: 106–114.
- VI Kauppila T., Kanninen A., Viitasalo M., Räsänen J., Meissner K. & Mattila J. 2012. Comparing long-term sediment records to current biological quality element data – Implications for bioassessment and management of a eutrophic lake. *Limnologica* 42: 19–30.

Author contributions:

- I The original idea was by Antti Kanninen (AK), with contributions from Veli-Matti Vallinkoski (VMV), Jarkko Leka (JL) and Heikki Hämäläinen (HH). AK, VMV and JL collected the field data. AK performed all analyses except the bootstrapping, which was conducted by Timo J. Marjomäki. AK wrote most of the paper, with comments from all other authors.
- II The original idea was by Kirsi Valta-Hulkkonen (KVH) and AK. All authors performed the field surveys jointly. Riitta Ilvonen (RI) and KVH were responsible for the digital image processing and interpretation. AK defined the measures of vegetation abundance. All authors interpreted the results. KVH and AK wrote most of the paper with comments from RI and JL.
- III The original idea was by KVH. KVH and AK planned the study. AK performed the field survey. KVH was responsible for the digital image processing and interpretation and created the biomass model. AK created the DEM. KVH and AK wrote the paper and Petri Pellikka made revisions to it.
- IV The original idea was by HH, with contributions from AK. AK and HH planned the study design, AK collated and processed data, Seppo Hellsten (SH) contributed the regulated lake data, and AK performed all analysis. AK wrote most of the paper with revisions from HH and comments from SH.
- V Janne Alahuhta (JA) was responsible for the study idea. All authors planned the study design. AK and JA jointly collated the macrophyte data. JA processed vegetation and catchment data and AK was responsible for water quality and lake position data. AK performed NMS-analysis and JA conducted all other statistical analyses. The results were interpreted together with JA and AK. JA and AK wrote the manuscript jointly, with comments from Kari-Matti Vuori.
- VI The original idea was by Tommi Kauppila (TK) and AK. TK, AK and Matias Viitasalo (MV) planned the study. The subfossil diatom and akinete data was analysed by MV and Johanna Räsänen, respectively. Kristian Meissner (KM), AK and TK performed numerical analysis of the subfossil chironomid data and TK of the diatom data. Jukka Mattila and TK performed sediment dating. TK was responsible for the sediment chemistry data. AK was responsible for collating the current biological and water quality data – including performing the macrophyte field survey – and their analysis. TK, AK and KM wrote most of the paper, with comments and methodological sections from other authors.

ABBREVIATIONS

BQE	Biological quality element
CEN	European Committee for Standardization
CIR	Colour Infrared (aerial photograph)
DEM	Digital elevation model
EI	Ellenberg Index
EQR	Ecological quality ratio
GAM	Generalized additive models
GIS	Geographical information system
MCD	Maximum colonization depth
MMI	Multimetric index
MTC	Metric of taxonomic composition
NMDS	Non-metric multidimensional scaling
O/E	Observed/Expected (-ratio)
OTS	Oligotrophy score
PI	Phytolittoral inventory
PMA	Percent Model Affinity
PVL	Percentage of vegetated littoral
SSI	Stressor-specific index
TI	Trophic Index
TS	Transect survey
TT50	Proportion of type-specific taxa
WFD	Water Framework Directive
WI	Water level fluctuation index

1 INTRODUCTION

1.1 Macrophyte communities of boreal lakes

Macrophytes, i.e. macroscopic aquatic plants, are a heterogeneous group of taxa: flowering plants, mosses, and macroscopic algae such as charophytes. They share the common feature of inhabiting an aquatic environment; growing either permanently submerged or rooted in at least temporarily inundated areas. Macrophytes can be functionally classified to life-forms, based on the occurrence of emergent, floating and submerged leaves. The main life-forms are helophytes, nymphaeids, isoetids, elodeids, ceratophyllids, lemnids, bryids and charids (Mäkirinta 1978). In addition, shore species like sedges (*Carex* spp.) – taxa occurring in the eulittoral – are often regarded as part of the aquatic flora (Heino & Toivonen 2008).

In boreal lakes, the macrophyte flora is rather diverse, including e.g. 72 hydrophyte, 29 helophyte and 25 bryophyte species, in addition to ca. 54 shore species in Finland. There is a strong latitudinal gradient in macrophyte diversity with northern waters harbouring less species (Heino & Toivonen 2008). Humic lakes – common in the boreal region with abundant peatlands – usually contain less species than their clear-water counterparts with the same trophic status (Rørslett 1991, Mäkelä et al. 2004).

In lakes, macrophytes inhabit the most heterogeneous part of the ecosystem, the shallow littoral zone. The heterogeneity results from high variation in shore morphology and surrounding geology, as well as the morphometry of the lake basin. Consequently, lacustrine aquatic flora is spatially strongly variable with various abiotic, as well as biotic, environmental factors (Lacoul & Freedman 2006). The main local determinants of the composition of aquatic flora are water level fluctuation (Rørslett 1989), exposure (Keddy 1983, Weisner 1991, Riis & Hawes 2003), substrate composition and organic matter content (Barko & Smart 1986), the amount of light (Spence 1982), and water chemistry (Toivonen & Huttunen 1995, Jeppesen et al. 2000).

Macrophytes have several important functions in lake ecosystems. Together with microscopic algae, they are the most important primary producers in lakes (Krause-Jensen & Sand-Jensen 1998, Nöges et al. 2010). With their physical structure, they provide habitat and shelter for fish, zooplankton and benthic invertebrates as well as a substrate for the growth of phytobenthos (Gasith & Hoyer 1998). Many fish species use macrophyte beds for spawning and some macrophytes are food for fish and avifauna (Schriver et al. 1995, Hansson et al. 2010). Aquatic vegetation alters the composition of its physical environment by shading and suppressing water turbulence, stabilizing sediments and transporting nutrients and oxygen between the sediment and the water (Smith & Adams 1986, Gasith & Hoyer 1998, Møller & Sand-Jensen 2012). Emergent macrophytes form an interface between the surrounding land and the water, and can thereby act as buffers against direct nutrient run-offs and be prone to reflect land use changes in the direct vicinity of the lake shoreline (Wetzel 1990).

In addition to being important components of the ecosystem, macrophytes are often the most conspicuous element of aquatic biota from the human perspective. As macrophytes inhabit the near-shore area, their abundance can limit the recreational value of the lake, and actions to reduce vegetation are commonly undertaken as part of lake management projects (Liddle & Scorgie 1980, Russell & Kraaij 2008). A part of a healthy ecosystem of especially clear-water lakes is abundant submerged vegetation (Scheffer 1998), which can become suppressed by anthropogenic pressures like nutrient enrichment and resulting phytoplankton dominance. On the other hand, in humic lakes, where submerged vegetation is naturally sparse, overgrowth – the expansion and thickening of e.g. reed beds – is a common response to eutrophication (Andersson 2001, Mäemets & Freiberg 2004, Partanen & Luoto 2006) and altered water-level fluctuation (Partanen & Hellsten 2005).

1.2 Ecological status assessment of freshwaters

The effects of anthropogenic activities on aquatic ecosystems are increasingly evaluated by biotic responses. This trend is reflected in and driven by legislation, like the Clean Water Act in the United States (Anon. 2002), the National River Health Program in Australia (Norris & Norris 1995) and the European Water Framework Directive (WFD; Anon. 2000). The WFD places ‘good ecological quality’ as the target of freshwater protection and restoration. In lacustrine ecosystems, the ecological quality is evaluated based on four ‘biological quality elements’ (BQEs): phytoplankton, aquatic flora (macrophytes and phytobenthos), macroinvertebrates and fish. Water physical-chemical quality and hydromorphological conditions are used as supportive information. To assess the condition of biota, certain defined community characters (most often ‘composition and abundance’; Annex V in the WFD) need to be interpreted and quantified. Finally, for the purpose of quality classification, the resulting metrics need to be integrated and expressed in the form of ecological quality ratios

(EQR), which represent the relative difference between the value of a biological parameter observed and the value expected in the reference conditions and range from 0 ('bad') to 1 ('high status'; Hering et al. 2010).

The concept of reference conditions (Stoddard et al. 2006) lies in the core of all bioassessment procedures. Since the primary focus of biological assessments is to evaluate the magnitude of the effect of anthropogenic activity, ecosystem status in the absence of human disturbance is the benchmark against which the status of biota in impacted sites needs to be compared (Hawkins et al. 2010a). Several options to derive reference conditions exist: if sites with minimal human disturbance still exist, these can be used to derive (spatially based) reference conditions (Anon. 2000, Stoddard et al. 2006). In the context of the WFD, other options are to use modelling or temporal changes i.e. historical data or palaeo-reconstruction (Anon. 2003).

When establishing reference conditions, the natural variation in the composition of biota is usually controlled for either spatially (ecoregions; Stoddard 2005), through continuous modelling (Moss et al. 1987), or through categorical typology (Aroviita et al. 2008). Typology i.e. the grouping of naturally similar ecosystems together is the default option in the WFD-approach (European Commission 2000). To be useful, a typology should adequately reduce the within-type natural variation in biological parameters used for the assessment of ecological status (e.g. Dodkins et al. 2005a). In practice, the resulting EQRs among reference sites should be close to unity, varying little and thus making it possible to detect any significant change in response to anthropogenic disturbances and to grade the status (Aroviita et al. 2008). However, simple typology systems do not necessarily cover all relevant abiotic variation affecting biota, as the BQEs may respond differently to environmental conditions (Mykrä et al. 2009, Nöges et al. 2009). Therefore, it is important to know the residual effect of natural variation on ecological quality metrics. In the typology context, equally important is to be able to assign each waterbody to a correct type to avoid status classification errors which may lead to unnecessary management efforts or failure to detect deterioration that has actually occurred. One way to validate status classification based on spatial reference conditions is to use site-specific reference conditions, derived via paleolimnology (Alahuhta et al. 2009a, Jyväsjärvi et al. 2010, Sayer et al. 2010) or modelling (e.g. Willby et al. 2009).

1.3 Macrophytes in lake status assessment

As macrophytes are vulnerable to a variety of changes in the aquatic environment (e.g. Arts 2002) they have for long attracted the attention of academic researchers (Linkola 1933, Ulvinen 1937, Kurimo 1970). Several features of macrophytes, like their relatively easy identification and immobility, make them suitable for bioassessment (Toivonen 2000). However, the use of macrophytes in continuous biomonitoring has been limited; e.g. in Finland,

operational macrophyte monitoring programs date back only a few years (Niemi 2006). However, the recent development of macrophyte-based monitoring and assessment schemes driven by the legislative demands, has been rapid in Europe (Schaumburg et al. 2004, Sondergaard et al. 2005, Leka et al. 2008, Penning et al. 2008a, Pall & Moser 2009, Kolada et al. 2011, Cellamare et al. 2011). Likewise, elsewhere in the world, macrophyte-based assessment of lacustrine ecological condition is increasingly common (e.g. Beck et al. 2010, Nichols et al. 2000, de Winton et al. 2012).

The WFD verbally indicates that 'taxonomic composition' and 'abundance' of macrophytes should be used in ecological status grading (Anon. 2000; Annex 5). In addition, according to the definition of 'good' status, 'excessive growth' of macrophytes and phytobenthos is not acceptable (Anon. 2000). However, the WFD does not specify which indices or metrics should be used to measure these features of the flora (Hering et al. 2010) and a multitude of different assessment metrics have been created (Birk et al. 2012).

Most commonly macrophytes are used as indicators of the trophic status of their habitat (Linkola 1933, Melzer 1999, Penning et al. 2008b). Most WFD-related macrophyte assessment systems, especially those of river flora (Schneider & Melzer 2003, Dodkins et al. 2005b, Haury et al. 2006, Holmes 2009), still rely heavily on the trophic indication value of different taxa, often formulated into stressor-specific indices. Multimetric indices (MMIs), often incorporating trophic indication in the collection of metrics, are also common (Willby et al. 2009, Beck et al. 2010).

However, there is a fundamentally differing alternative to stressor-specific indices: the general measures of community change. These general measures directly compare, via differing mathematical formulations, the observed taxonomic composition to that expected under reference conditions. An advantage of general measures of taxonomic composition is that they provide conceptually unified assessments across different stressor gradients, habitats and organisms (Jyväsjärvi et al. 2011). The most common application is the evaluation of 'taxonomic completeness' or the ratio of the observed taxa (O) at a site to those expected to be present (E) in the absence of human disturbance (Hawkins et al. 2010a); an approach originally developed for river benthic macroinvertebrates (Wright et al. 1998). In addition to lotic macroinvertebrates, O/E-ratios have been applied to many other groups of aquatic organisms (Joy & Death 2002, Kennard 2006, Cao et al. 2007, Mykrä et al. 2009, Hawkins et al. 2010b, Aguiar et al. 2011, Jyväsjärvi et al. 2011), whilst direct applications to lacustrine flora are still rare (Keto et al. 2006, Sutela et al. 2012).

The species pool used in macrophyte-based status assessment and monitoring is almost equally important as the selection of metrics. Helophytes, inhabiting the lake eu- and geolittoral, are most often omitted from assessment systems (Poikane et al. 2011), mainly due to the presumably high variation in taxonomic composition caused by e.g. soil characteristics and shore morphometry (Penning et al. 2008b). However, the effects of omission or inclusion of emergent taxa has previously only rarely been specifically tested, and with variable outcomes (Dudley et al. 2011, Kolada et al. 2011).

1.4 Methods for monitoring macrophytes

1.4.1 Field surveys

Biological assessment schemes necessitate consistent survey and sampling protocols. The WFD requires monitoring of biological quality elements to be conducted using standardized methods (Anon. 2000). Even though the recently established CEN standard for monitoring aquatic macrophytes in lakes (Anon. 2006) recommends a transect-based method, considerable variation still exists in the monitoring protocols implemented under the WFD (Gunn et al. 2006, Kolada et al. 2009). According to a query covering 14 European countries (Gunn et al. 2006), some type of transect survey method is in use in at least 8 countries as the primary monitoring method. In Finland, the monitoring methodology based on transects was established in late 1930s (Vaarama 1938) and has been further developed and applied by e.g. Toivonen and Lappalainen (1980) and Hellsten (2001). At present, the so called 'main belt transect method' (Leka & Kanninen 2003; described also in chapter 2.2 in p. 17), is used in the WFD compliant surveillance and operational monitoring programs.

Albeit a common preference for transect-based surveys, a so-called phytolittoral mapping method (a species inventory supplemented with the mapping of the main stands of macrophytes) is used in several countries as a WFD-compliant monitoring method, either supplementary to transect surveys or as the sole method (Kolada et al. 2009). Although not meeting the recommendations of the CEN standard, floristic inventories and/or mappings, have earlier been successfully applied to investigating patterns in biodiversity, community composition and environmental relationships of macrophytes (Toivonen & Huttunen 1995, Heegaard et al. 2001, Mäkelä et al. 2004, Capers et al. 2010), as well as to reveal long-term changes in macrophyte communities (Toivonen 1985, Rintanen 1996, Ranta & Toivonen 2008). In Finland, most of the early studies (e.g. Linkola 1933) of lake macrophytes were conducted using floristic inventories without any specific methodological standardization.

1.4.2 Remote sensing

One of the advantages of using macrophytes in biomonitoring is the possibility to employ remote sensing, a technique that is not applicable to other BQEs, to derive information (Silva et al. 2008). Aerial photographs have been used frequently to map aquatic flora during the last decades (e.g. Meriläinen & Toivonen 1979, Toivonen & Nybom 1989, Marshall & Lee 1994, Partanen et al. 2009). Satellite based imagery has also been used for long (e.g. Ackleson & Klemas 1987) and its use is becoming increasingly common as the sensor technology continues to develop (Xie et al. 2008). Historically, the aerial images have been interpreted visually – aided by field observations – and by manually delineating vegetation stands on maps (Silva et al. 2008). Since the rapid development of information technology, digital image processing and use of

Geographic Information Systems (GIS) has become the norm (Xie et al. 2008). The production of thematic maps of different vegetation categories in a GIS can be based on visual interpretation (Partanen & Hellsten 2005, Valta-Hulkkonen et al. 2003a) or automated image analysis (Marshall & Lee 1994, Valta-Hulkkonen et al. 2003b, Silva et al. 2008).

In automated image analysis - or spectral sorting - digital images are used and image analysis software is trained to recognize a set of spectral patterns or 'signatures' which are unique for particular macrophyte species or groups (Marshall & Lee 1994). All pixels that comprise the image are then classified on the basis of their conformance with these signature values; this results in a map or GIS overlay of aquatic plant distribution. A number of studies have shown that spectrally distinct signals are governed by the density of the vegetation, the openness of the canopy, and the amounts, forms and orientations of the leaves (Ackleson & Klemas 1987, Marshall & Lee 1994, Malthus & George 1997, Silva et al. 2008), and thus at least life-form discrimination of macrophyte taxa is attainable (Valta-Hulkkonen et al. 2003a).

One advantage of remote sensing data is that they provide spatial and quantitative information on the aquatic vegetation. They also allow geometrically rectified images to be superimposed on and compared with other geographical data in order to study temporal changes or interactions between parameters (Jensen et al. 1992, Lehmann 1998, Partanen et al. 2009).

1.5 Aims of the study

Despite long research traditions, aquatic macrophytes have only recently become an established constituent of boreal lake bioassessment and monitoring. Therefore, various questions relating to the practical implementation of macrophyte monitoring and status assessment schemes have needed to be addressed throughout the last decade. The six studies in this thesis explore a range of these issues. Reliable assessment and monitoring schemes necessitate consistent and well-established methods to study macrophyte vegetation. Hence, the first aim of this thesis was to evaluate alternative macrophyte survey methods. First, two commonly applied field survey approaches, transect-based survey (TS) and a phytolittoral inventory (PI) were contrasted directly by pairwise, within-lake comparisons, and indirectly through reference lake data collected with either TS or PI (I). The methods were compared with respect to species detection rate and cost-efficiency, observer-related variation and variability of derived ecological status metrics. Secondly, utility of remote sensing as a macrophyte mapping method was tested (II, III). Lakes of varying type and trophic status were studied with the aim to evaluate the accuracy with which information on different types of aquatic vegetation could be derived from contemporary colour infrared aerial photographs under various lake conditions. In addition, historical black-and-white images were used to analyze long-term changes of the vegetation (II, III).

The second major aim of the thesis was to compare different approaches in using taxonomic information to define metrics for status assessment of lacustrine macrophytes (IV, V). In particular, two fundamentally different approaches, stress-specific metrics and general measures of taxonomic composition, were contrasted by assessing their performance with regard to the two most common anthropogenic stressors, nutrient enrichment and water level regulation (IV). However, the performance of metrics included in the current operational assessment scheme was evaluated as well (I, V, VI). In a comprehensive case study, the results of macrophyte status assessment along with those of other BQEs were compared with the paleolimnological record of biotic change in a currently eutrophic lake (VI). In addition, the possibilities to derive quantitative macrophyte abundance metrics from remotely sensed data were explored (II).

Ultimately, management and restoration of waterbodies significantly impacted by anthropogenic activities are needed to reach the ambitious goal of 'good ecological condition' set in water resource legislation. The third major theme of this thesis - the role of macrophytes in lake management - was addressed in two studies (III, IV). First, remote sensing data was used to study the long-term effects of rehabilitation efforts of a shallow, humic, hypertrophic lake on aquatic vegetation (III). Secondly, the existence of scale-dependent patterns that might have important implications for the optimal targeting of management efforts between anthropogenic land-use and the status of macrophytes was studied (IV). This was achieved by contrasting the relationship between land-use of lake marginal zones of various widths (up to the whole catchment) and macrophyte status metrics.

2 MATERIAL AND METHODS

2.1 Study lakes

The study area consisted of whole Finland excluding the northern-most part; the majority of lakes were situated in central-eastern Finland (Fig. 1). Macrophyte data of altogether 182 lakes were used (I-VI); they were either directly obtained for the purpose of the studies (I, II, III, VI) or derived from various database sources (IV, V). The studied lakes represented a variety of environmental conditions, from near-pristine lakes (I, II, IV, V) to variably eutrophic (I-VI) and regulated lakes (II, IV). The nutrient loading originated mainly from diffuse sources such as agriculture, forestry and peat production, while a few lakes also received nutrient loading from point sources such as wastewater treatment plants. The gradient of water level regulation ranged from mildly regulated to the most heavily regulated lake in Finland (IV). The vast majority of the lakes were small ($< 5 \text{ km}^2$), but some larger lakes were also included (IV) with the largest ones exceeding 100 km^2 in area.

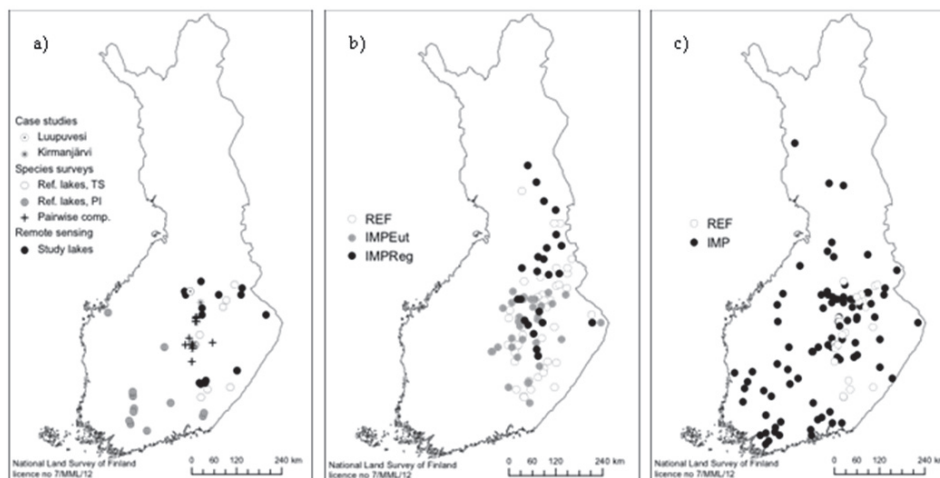


FIGURE 1 Location of the study lakes: a) I (Species surveys), II (Remote sensing), and the case study lakes III (Luupuvesi) and VI (Kirmanjärvi) b) IV and c) V. REF=reference lakes, IMP=impacted lakes (Eut=eutrophied, Reg=Regulated). TS=Transect survey, PI=Phytolittoral inventory.

The establishment of reference conditions for lake ecological status classification in Finland is currently based on the 'system B' typology of the WFD (Anon. 2000). The categorical typology separates a total of 14 lake types and is primarily based on lake area, mean depth, dissolved organic carbon concentration (measured as colour) and altitude (Vuori et al. 2006). In addition, turbidity and alkalinity are used to separate naturally nutrient rich and calcareous lakes, respectively. Lakes with a short retention time (< 10 d) are also separated to a specific type. In the Finnish provisional macrophyte classification system, lakes are further divided to northern and southern sub-types due to strong latitudinal gradient in species richness and composition (Heino & Toivonen 2008, Leka et al. 2008)). The operational typology, along with the nationally derived reference values for the biological quality elements, was used for establishing the type-specific reference conditions in II, III and VI. In I, to ensure an adequate number of reference sites, the operational typology was slightly modified by combining the largest ($A > 5 \text{ km}^2$) lake types of each humic (colour) category. Altogether, the study lakes (I-VI) represented a wide array of lake types present in Finland from small, oligohumic lakes to large humic lakes, and naturally nutrient rich lakes.

2.2 Macrophyte species data (I-VI)

The macrophyte species surveys (I-VI) were mainly conducted using a 'main belt transect' method (Leka & Kanninen 2003, Leka et al. 2003). The method conforms to the CEN standard (Anon. 2006), suggesting a transect-based method. The

method has been in use in the Finnish national macrophyte monitoring programs since their official start in 2006 (Niemi 2006) and quality assurance protocols have been developed for its application (Kuoppala et al. 2008).

In the main belt transect method observations of macrophyte species are made along a 5 m wide transect extending from the upper eulittoral to the outer depth limit of macrophyte vegetation (Fig. 2A). Observations are made in shallow water by wading, using a hand-held rake and a bathyscope. In deeper water, observations are made from a boat, using a bathyscope, a long-shafted rake, Luther-type rake and, as a supplement, an underwater drop-camera to determine the abundance and depth limit of isoetid vegetation. The transect is divided to zones (units of observation) according to the dominating life-form or species (hence the term 'main belts'). In the original application of the method (Leka & Kanninen 2003) the macrophyte species are recorded and their frequency and abundance estimated on a continuous percentage scale separately in each zone. In 2007, the methodology was revised and a slight simplification has been applied since: the species are only given one frequency and abundance estimates once for the whole transect, i.e. only one unit of observation is used. Further details are given in I and Kuoppala et al. (2008). In IV, the species data of 23 lakes were collected using a slightly differing transect method: the coverage estimation was at 1-7 Norrlin scale instead of the usual percentage scale, following earlier field standard used in regulated lakes (Partanen & Hellsten 2005).

The main belt transect method was compared to another commonly applied survey method, the phytolittoral inventory (PI), in five lakes (I). In this application of PI, the entire lake shoreline was divided to five stretches equal in length (20 % of the total shoreline length; Fig. 2B). The survey team travelled each stretch mainly by a rowing boat, supplemented by regular wading in shallow water, and occasional examination of the shore vegetation by walking along the shoreline. Observations of submerged vegetation were made with the help of rakes and a bathyscope, occasionally viewing with an underwater drop-camera. Floating and emergent vegetation was also recorded. At the end of each stretch of shoreline, the detected taxa were given estimates of frequency and abundance at percent scale.

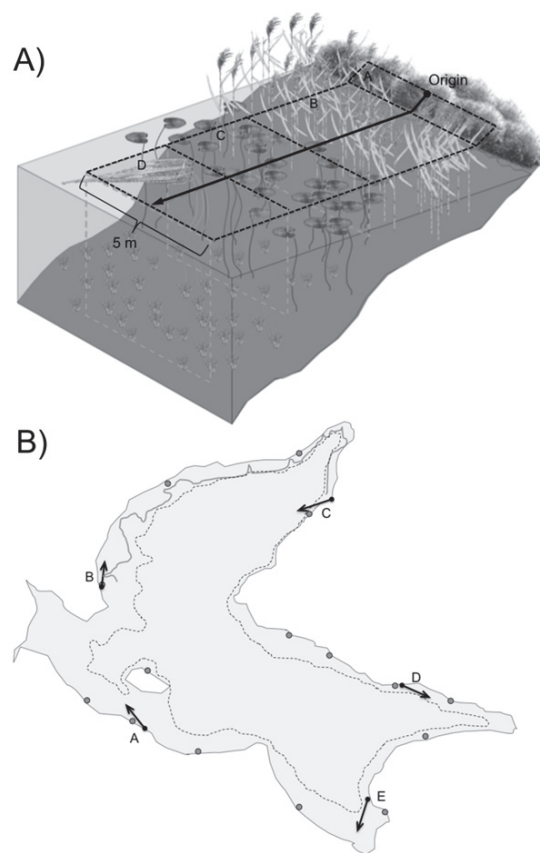


FIGURE 2 Survey methods used in the study. A) A schematic illustration of the transect method ('main belt transect'). The 5 m wide transect is divided into zones (A-D) according to the dominant life-form or species. B) An illustration of the pairwise comparison of methods (I) from Lake Suurijärvi. Origins of 15 transects are depicted by points and origins of the five shoreline stretches surveyed by the phytolittoral inventory method by arrows (A-E). In stretch B the actual route (based on GPS-measurement) travelled by the survey team is illustrated with a solid grey line and the approximate outline of the vegetated littoral (based on aerial image interpretation) with a dashed line.

2.3 Remote sensing data (II, III)

The remote sensing method, applied here (II, III), was originally developed by Valta-Hulkkonen (2003a, 2004). The method utilizes digital colour infrared (CIR) aerial photographs, which are acquired under specifically defined conditions, to minimize disturbance by various factors like light fall-off, shading by near-shore trees and wave effects. In addition to newly acquired data, the image interpretation methods can be applied to historical aerial images obtained from

archives for analyzing long-term changes in vegetation abundance (II, III, Vallinkoski et al. 2004).

The contemporary remote sensing data (II, III) were based on digital CIR aerial photographs to a scale of 1:20 000. The photographs were scanned to a ground resolution of ca. 0.5 m. The images consisted of three wavelengths: green (500–575 nm), red (575–675 nm) and near-infrared (675–900 nm), which were separated into different bands. The acquisition of contemporary data is described in more detail by Valta-Hulkkonen et al. (2004).

The historical data were derived from black and white aerial photographs to a scale of 1:20 000 from the 1950s (II, III), consisting of wavelengths of visible light (400–700 nm) in a single band. The photographs were scanned to a ground resolution of ca. 0.5 m. In addition, archived CIR data from 1996 (scale 1:30000 and ground resolution of 0.8 m) was used (III).

The ground data, against which the image interpretation and classification accuracy assessment is performed, were based on main belt transects (see chapter 2.2) and additional reference plots, which were representative areas of different densities of each dominant life form and/or species of aquatic macrophytes present in the lake (for details, see II).

2.4 Palaeobiology and other biological data (VI)

Palaeolimnological methods were applied to sediment core samples of lake Kirmanjärvi (VI). The sediment cores were analyzed for diatom valves and chironomid head capsule remains along with cyanobacterial akinetes and remains of green algae. In addition to palaeobiological analyses, the cores were analyzed for sediment chemistry, and dated. The sediment-record derived changes in composition of aquatic biota were compared with contemporary biological data of macrophytes, phytoplankton, profundal zoobenthos and fish. All sampling of the current biological data – including macrophytes – were performed according to the relevant standard procedures.

2.5 Explanatory data

Explanatory environmental data (I–VI) were obtained from various databases. Lake water quality parameters were derived from the national water quality database (Hertta, maintained by the Finnish Environment Institute) and epilimnetic concentrations of the productive period (1.6. –30.9.) were used (I–VI). Hydrological data (III, IV) were derived from the hydrological database (Hertta). The percentage of arable land (IV) was extracted from the Watershed Simulation and Forecasting System of the Finnish Environment Institute.

GIS were applied to obtain explanatory or supportive data (II, III, V). GIS were used to analyze land-use at different spatial scales around the marginal

zones and the whole topographic catchment of the study lakes (V). Various spatial land-use databases were used to derive the percentage of arable and urban land-use, as well as the drainage ditch density of the different catchment areas (V).

In the remote sensing studies (II, III), bathymetric data and GIS were used to construct digital elevation models (DEM) of the study lakes. The DEMs were applied to delineate the zone of suitable depth for the growth of nymphaeid and helophytic vegetation by outlining the depth zone above the maximum observed depth of nymphaeid vegetation (II). DEMs were also utilized to correlate the degree of change in vegetation cover to lake depth (V).

2.6 Numerical methods

2.6.1 Multivariate methods for community composition

The macrophyte community data (I, IV, V) were analyzed using non-metric multidimensional scaling (NMDS). NMDS is considered a robust ordination method and has the advantage over other ordination methods that it has no strict assumptions on the data, such as the normality or linearity (McCune & Grace 2002). Thus, it can be used also with presence-absence, as well as with abundance data. Formation of distinct groups in the n-dimensional ordination space, according to the grouping factor of interest, indicates a difference in species composition. In addition, environmental variables can be *a posteriori* correlated with the ordination axes, to explore relationships of community variation with the environment.

NMDS was used to explore if macrophyte species composition differed between reference and impacted sites, and to study the most relevant environmental variables relating to community variation (IV, V). It was additionally used to explore the grouping of reference lakes by a priori lake types (IV), and to analyze whether the detected community composition of macrophytes differed between two survey methods (I).

Cluster analysis was applied to diatom data to form distinct zones of diatom community composition (VI). Ordination based on principal components analysis (PCA) was used to visualize the results. Non-parametric analysis of similarity (ANOSIM; Clarke 1993), which tests for differences between groups in multivariate data, was used on the chironomid data to test for differences between the diatom-based zones. A related method, multi-response permutation procedure (MRPP), was used to test for differences in macrophyte composition between status groups and lake types (IV).

2.6.2 Macrophyte metrics for species data

For the purpose of macrophyte status assessment, several metrics or indices were utilized (Table 1). The provisional ecological classification system in Finland

(Vuori et al. 2009, Rask et al. 2011) relies on three macrophyte metrics, which were used as such (I, V and VI). Two of these metrics (TT50 and PMA; Table 1) are based on the comparison of the general taxonomic composition of impacted and reference sites, while one (TI; Table 1) is a stressor-specific index (SSI). In addition to TI, two other SSIs (EI and OTS; Table 1) relating to nutrient enrichment and one (WI; Table 1) to water level fluctuation were calculated. SSI were compared to general measures of taxonomic composition (MTC) as tools in macrophyte status assessment (IV), for which two MTCs (O/E and BC index; Table 1), in addition to PMA, were utilized.

TABLE 1 Status metrics based on macrophyte species data used in this study, with original references. The specific metric formulas are defined in the corresponding studies (I-VI).

Metric	Original reference	Studies
Trophic Index (TI)	Penning et al. 2008b	I, IV, V, VI
Ellenberg Index (EI)	Kolada et al. 2011	IV
Oligotrophy score (OTS)	Kanninen et al. 2009	IV
Water-level fluctuation index (WI)	Hellsten & Mjelde 2009	IV
Proportion of type-specific taxa (TT50)	Vallinkoski et al. 2004	I, V, VI
O/E -ratio	Moss et al. 1987	IV
BC index	van Sickle 2008	IV
Percent Model Affinity (PMA)	Novak & Bode 1992	I, IV, V, VI

2.6.3 Ecological quality ratios (EQRs)

For the purpose of presenting the results of ecological classifications, the WFD (Anon. 2000) necessitates the use of Ecological Quality Ratios (EQRs). The standard procedure (e.g. Hering et al. 2006) for defining an EQR is to calculate it as the relation between observed (O) and expected (E) metric values. O is the metric value calculated from the biological data of the water body under evaluation and E is derived as the average (or median in some cases; Vuori et al. 2009) of metric values across reference sites of the corresponding site category (e.g. lake type).

To enable direct comparison among metrics and lake types, the EQR values need to be re-scaled. A linear re-scaling (Mykrä et al. 2012) was applied (IV, V), which results in each quality band corresponding 0.2 units within the 'classification scale' from 0 to 1. The metrics can also be re-scaled via a simple scoring system, as utilized in the provisional ecological classification system in Finland (Vuori et al. 2009) and in I and VI. In this procedure, each metric is first classified to a quality band based on its individual expected values and class limits. Each quality band corresponds to a score (bad = 0.1, poor = 0.3, moderate

= 0.5, good = 0.7 and high = 0.9). The BQE status class is given as the median of the metric scores.

2.6.4 Methods for relating macrophyte metrics to explanatory variables and sample size

Generalised additive models (GAM) were used to analyze the relationships between explanatory environmental variables and aquatic macrophyte metrics (V). GAMs are non-parametric extensions of generalised linear models that allow linear and complex additive response shapes or a combination of the two within the same model (Hastie & Tibshirani 1990). GAMs estimate response curves with a non-parametric smoothing function instead of parametric terms, thus allowing exploration of shapes of response curves to environmental gradients (Lehmann et al. 2003). Therefore, GAMs are more data driven than generalised linear models, and enable the fitting of statistical models in better agreement with ecological theory (Lehmann et al. 2003).

Simple parametric scatter-plotting and or non-parametric correlation analysis were used to explore the associations of macrophyte metrics and environmental variables in other studies (II, IV).

For the purpose of evaluating the effect of survey-effort on the precision of the metrics, as well as on the detected number of species, resampling with (transects) and without (shoreline stretches of the PI) replacement was performed (I).

2.6.5 Classification methods of remotely sensed data

A numerical classification procedure was applied to the digital CIR and black-and-white aerial photographs (II, III). Firstly, only photographs or parts of photographs without strong geometric or radiometric effects, specular reflectance, relief displacement or light falloff (Lillesand & Kiefer 1994, Pellikka 1998), were chosen for processing. In the case of the 1950s images, the effect of light falloff was normalised using a method by Pellikka (1998). The photographs were geo-referenced, mosaicked (several adjoining images combined) and terrestrial areas masked out (II, III). The photographs were classified, using a maximum likelihood classifier, to several (up to 6) categories of aquatic vegetation for the CIR aerial photographs and two categories, vegetation and water, in the case of the black and white images. The field data (see 2.3), when available, were divided for use as training areas for the automated classification and as an independent validation set for accuracy assessment. For archived images the training and validating areas were chosen by visual interpretation.

The accuracy of the resulting classification to vegetation categories was evaluated by using confusion matrices, which compare, on a category-by-category basis, the relationship between known reference ('ground truth') data and the corresponding results of an automated classification (Lillesand & Kiefer 1994, Xie et al. 2008)). The accuracy of each vegetation category was defined by

examining whether the majority of the areas of the reference plots in the validation set were classified into the right category or not.

A spatial biomass model of Common Club-Rush (*Schoenoplectus lacustris*) in lake Luupuvesi by correlating image digital number (DN) values of the near-infrared band to biomass measurements (III).

3 RESULTS AND DISCUSSION

3.1 Optimal species survey protocols (I)

In the pairwise comparison of the transect survey and phytolittoral inventory, more species in less time were detected with PI than with transects (I). The pattern is partly attributable to general positive species-area relationship (Connor & McCoy 1979, Rørslett 1991, Mäkelä et al. 2004) and greater area covered by PI than TS, but also to differing species-observation probabilities of emergent and submerged species among methods (Croft & Chow-Fraser 2009). The cost-effectiveness of the PI makes it a more suitable method for biodiversity surveys focusing on overall species richness and rare species; this was exemplified by the detection of a spatially limited and scarce, previously unknown population of the red-listed species *Najas tenuissima* from Lake Suurijärvi only during the PI survey (I).

Survey-effort (number of transects or percentage of shoreline surveyed) method had an effect on both the detected number of species and ecological quality metrics relating to taxon composition (I, V). The number of transects used in the TS (I) was premised on a previous study in which rarefaction results based on 30–50 transects per lake suggested that 15 transects were enough to capture $82 \pm \text{s.d. } 4.9\%$ of all taxa (Leka & Kanninen 2003, Leka et al. 2003). In line with this, the current findings (I) showed that the 15 transects yielded approximately 80 % of the total number of species found in an inventory of the entire shoreline of a small lake. Hence, for capturing the general patterns of macrophyte community composition and diversity by TS, 15 transects can be regarded sufficient, although some rare species will inevitably remain undetected. For ecological quality metrics, a smaller survey effort may be sufficient: the results showed that 8–10 transects or 40 % of shoreline should be surveyed to reach relatively stable estimates of the metrics. However, both of these findings are only valid for the types of lakes ($A < 500$ ha) included in these studies, as in larger lakes higher within-lake variation might require more transects to obtain reliable metric

values by TS. The figures concerning classification metrics are also conditional to the reference data and the metrics used.

The TS and PI methods were found not to differ with respect to operator variability (II). There were no significant differences in macrophyte metrics among observers in either method, which contrasts with some studies in riverine environments (Pentecost et al. 2009, Hurford 2010), but is similar to other studies with a limited number of trained observers (Staniszewski et al. 2006, Croft & Chow-Fraser 2009). The observers in the current study (II) had good prior expertise in macrophyte monitoring, had undertaken training and a common intercalibration session before commencing fieldwork. Hence, the results highlight the importance of quality assurance protocols (e.g. Kuoppala et al. 2008) in implementing monitoring programs and producing consistent monitoring results.

3.2 Remote sensing as a monitoring method (II, III)

The contemporary digital CIR aerial photographs were suitable for mapping helophytes and nymphaeids, but less accurate for mapping submerged vegetation (II, III). Generally, helophytes could be divided into two main categories according to their phenotype: species with well-developed leaves, such as *Phragmites australis* and *Carex* spp., and species without (or with reduced) leaves, such as *Schoenoplectus lacustris* and *Equisetum fluviatile*. In some cases it was possible to divide these two categories further at the species or genus level, or in lakes with abundant vegetation, to further classify them to two density categories (II, III). The classification of submerged vegetation succeeded only in one clear-water lake, where the isoetid *Lobelia dortmanna* formed a category of its own. The classification accuracy ranged from 69–100 %, the average accuracy being 82 % and the most obvious confusions occurring between density categories and with submerged vegetation and water (II).

The classification of vegetation from CIR photographs did not produce taxonomically exact categories, but the life-form-based classification is applicable in e.g. management purposes (III) and is similar to those used in previous remote sensing studies, e.g. that of Marshall and Lee (1994) and Valta-Hulkkonen et al. (2003a). Since features such as water depth and color, and bottom type, have a strong effect on the discrimination of submerged vegetation, digital CIR aerial photographs cannot be regarded as reliable for mapping elodeids and isoetids. In the case of helophytes, visual interpretation may produce taxonomically more accurate categories (Valta-Hulkkonen et al. 2003b). The development of remote sensing methods has recently enabled more detailed taxonomic classifications, especially of helophyte and floating-leaved vegetation (Husson et al. 2012), but even for submerged taxa, especially when using hyperspectral imagery (Hunter et al. 2010).

The most obvious application of remotely sensed data is analysis of long-term changes in the abundance of aquatic macrophytes (Toivonen 2000). For

example, water level regulation leads to changes in the zonation of aquatic vegetation: while increased winter draw-down hampers the success of sensitive isoetid species, decreased spring flood usually reduces upper eulittoral vegetation, e.g. *Carex* spp., and increases overgrowth by helophytes such as *Phragmites australis* (Partanen et al. 2006). The long-term changes in helophyte extent of regulated lakes have been successfully quantified by using historical and contemporary aerial images (Partanen & Hellsten 2005, Partanen et al. 2009). In the current study, the vegetation of six slightly humic, originally oligotrophic lakes had expanded according to a comparison between historical and current images (II). The relative increase was not related to nutrient concentrations, however, although the increase was greater (> 350 %) in slightly impacted (Keihäsjärvi and Ukonvesi currently in 'good' ecological status; Hertta database) or moderately impacted lakes (Alimmainen, 'moderate' status) than in reference quality sites (expansion of 74 - 233%; Keskimmäinen, Syysjärvi, Suuri-Vahvanen; 'high' status). The use of historical images, while enabling the quantification of long-term changes of aquatic vegetation, does have the drawback that usually no data for field validation of the results are available. The black-and-white images also lack the spectral heterogeneity and spatial resolution of contemporary CIR images (Silva *et al.* 2008). Therefore, underestimation of vegetation cover in historical images is possible (III).

One of the greatest advantages of remotely sensed data is that they can be superimposed and compared with other spatial data in a GIS. In this study, the thematic vegetation maps of different years were superimposed on a lake DEM to analyze the effect of water depth on the change in vegetation cover: greater changes in the extent of emergent vegetation had occurred between 1996 and 2001 in the deeper than in the shallow water areas of lake Luupuvesi (III). 'Colonization degree' or percentage of vegetated littoral (PVL) of emergent and nymphaeid vegetation - derived by superimposing the vegetation maps on the map of potential colonization area - was found to positively correlate both with the field-estimated abundance of helophytes and nymphaeids and the nutrient (Total P and N) concentrations of the lake (II).

3.3 Methods and metrics for evaluating macrophyte status (II, IV, V, VI)

NMDS ordinations indicated that the main environmental factors relating to the differences between lakes in macrophyte taxonomic composition were nutrient concentrations and anthropogenic land use, but lake morphometry and geographical position were also important (IV, V). The grouping of reference lakes according to type supported the utility of the current typology in macrophyte status assessment (IV), although some of the residual variation in status metrics was still explained by typology-related factors (V). Most importantly, the ordination analyses (IV, V) confirmed differences in the

taxonomic composition between *a priori* reference and impaired sites – a basic prerequisite for formulating metrics which measure the difference in species composition between natural and deteriorated ecosystems (Stoddard et al. 2006, Beck & Hatch 2009).

The performance of the metrics used in the current Finnish macrophyte classification scheme (Vuori et al. 2009) varied. The sensitivity shown by TT50 and PMA – measures of taxonomic composition – was higher than that of the stressor-specific TI (V, VI). Similarly, the general comparison of SSIs and MTCs (IV) showed that MTCs measure the community change caused by both eutrophication and water-level regulation, while SSIs mainly react to the specific stressors. The best performing MTC – the BC index – was capable of classifying impacted sites as impaired almost as often as the minimum of several SSIs (IV). The BC index incorporates not only taxa loss, but also taxa gained under impairment (van Sickle 2008), which makes it especially suitable for assessment of lacustrine macrophytes; the flora of naturally oligotrophic or dystrophic boreal lakes tends to (at least initially) diversify with increased disturbance, especially with nutrient enrichment (Penning et al. 2008a, Rørslett 1991). The eutrophication-induced shift in species composition is due to sensitive species, like isoetids, decreasing and/or disappearing and several more demanding species, e.g. lemnids, appearing, which is reflected by the BC index. These features are also, at least partly, shared by the TT50 index, the best performing macrophyte metric of the current Finnish operational classification scheme (V; Kanninen et al. 2009). In addition, a recent comparison among Nordic countries demonstrated a lower affinity of the Finnish, MTC-based macrophyte classification scheme – compared to SSI-based systems – for penalizing lakes harboring red-listed species by assigning them to impaired status (Ecke et al. 2010). Therefore, using MTCs in status assessment may also relieve conflicts between potentially contrasting legislation, like the WFD and the Habitats Directive (Anon. 1992, Ecke et al. 2010)

Inclusion of emergent taxa did not significantly affect the performance of the SSI or MTC metrics according to the numerical criteria used (IV). On the other hand, assessment metrics including helophytes yielded a stronger response to anthropogenic land-use pressure than a metric (TI) omitting them (V). While the numerical evidence supporting the use of helophytes is limited, ultimately, the metric used to evaluate ecosystem status should be selected using ecological criteria, and ideally, be related to ecosystem function or other values (Smyth et al. 2007, Aroviita et al. 2010, Cao & Epifanio 2010). Helophytes are important components of boreal lacustrine ecosystems, providing feeding and spawning habitat and shelter to aquatic life especially in humic lakes where submerged vegetation is sparse (Gasith & Hoyer 1998, Lacoul & Freedman 2006). Hence, their inclusion in bioassessment of ecosystem condition might be justified for this reason alone, despite the drawback that inclusion of helophytes in macrophyte monitoring programs comes at the expense of more time needed for conducting the survey (I).

Including macrophyte abundance in assessment metrics is necessitated by the normative definitions of the legislation (European Commission 2000). In this

study, including abundance did not enhance the performance of SSIs (IV). Similarly, the PMA index – an MTC incorporating abundance – underperformed its presence-absence counterparts (IV, V). In riverine macrophyte status assessment, compositional metrics incorporating abundance have been reported to perform better than presence/absence metrics (Aguiar et al. 2011) and several studies have pointed to abundance being more sensitive to hydrological change than species occurrence (Nilsson & Keddy 1988, Coops & van der Velde 1996, Hellsten 2001). In the present study, the weaker performance of abundance-related metrics may be explained by the confounding effect of interannual variation, which is greater for abundance than taxon presence (Diekmann 2003) or methodological noise, like among-surveyor variation, which may be higher in metrics considering abundance (I).

Changes in the vertical and horizontal distribution of macrophyte stands are one of the most easily detected changes in macrophyte vegetation (Toivonen 2000, Partanen & Hellsten 2005, Partanen et al. 2006). Most current European macrophyte assessment systems utilize quantitative measures of abundance by e.g. using maximum colonization depth (Poikane et al. 2011), which is widely considered a robust metric for lake ecosystem quality, corresponding to abundance of submerged vegetation (Spears et al. 2009, Kolada et al. 2011). Other abundance metrics, e.g. based on relative taxon abundances, are rarely utilized (Poikane et al. 2010). Remote sensing is an alternative for providing quantitative measures of vegetation abundance – especially that of helophytes and floating-leaved plants – (II, III), in addition to field measurements of depth distribution.

The utility of the PVL (II) as a metric of macrophyte abundance was – based on the initial finding (II) – studied by Vallinkoski et al. (2004), using available archived CIR images on a larger lake population ($n = 19$). In this study, the same methods for aerial image interpretation and PVL-derivation were utilized as in (II), but a binary classification (vegetation-water) was used. As the study lakes were humic (color $> 40 - 230$ mg Pt l^{-1}) and the submerged vegetation was sparse according to the field surveys (with the exception of deep-growing isoetids in some reference sites; Vallinkoski et al. 2004), no separation between submerged vegetation and water was expected (II). Thus, the ‘vegetation’ category was taken to represent the helophyte and nymphaeid vegetation, as in (II). In the combined data, the PVL is positively associated with total phosphorus concentration, a commonly applied proxy for eutrophication pressure (Fig. 3a). It also negatively correlates with an independent measure of the status of the macrophyte vegetation, the BC index (IV; Fig. 3b).

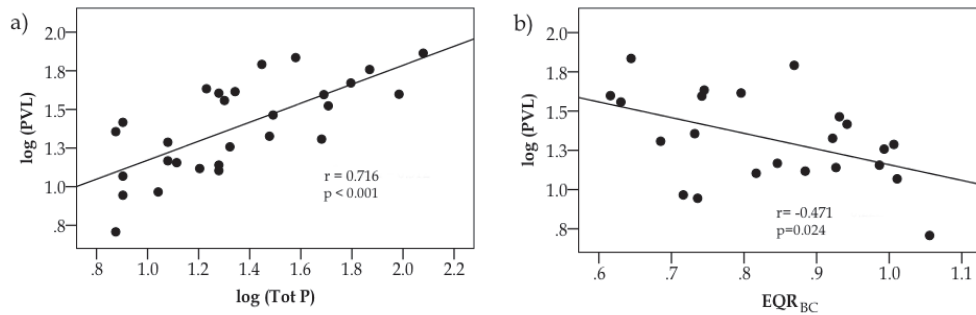


FIGURE 3 Percentage of vegetated littoral (PVL) plotted against a) logarithm of lake total phosphorus concentration ($\mu\text{g/l}$) and b) macrophyte EQR based on the BC-index. A linear fit line, Pearson correlation coefficient and significance of the correlation are shown. The PVL data are combined from II and Vallinkoski et al. (2004), the EQR_{BC} data are taken from IV.

The PVL can be regarded as a measure of macrophyte ‘abundance’ or ‘excessive growth’, as required by the normative definition of the WFD (Annex V: Anon. 2000), and the use of total vegetated area as a status metric has been advocated before (Anon. 1998, European Commission 2003). Likewise, catchment land-use (Cheruvilil & Soranno 2008) and lake water quality (Smith & Wallsten 1986) have been shown to predict the cover of emergent and floating-leaved macrophytes. Given its promising performance, PVL could be used – instead of or in addition to field-survey-based abundance metrics – as a bioassessment metric, cost-effectively derived from aerial images or other sources of suitable remotely sensed data. An obvious substantial advantage of remote sensing data and PVL derived from those is the potential of obtaining and using historical lake specific information. The use of PVL does, however, include potential sources of error, which need to be addressed before the metric is used in operational status classification. The DEMs – on which the delineation of the potential colonization relies – were based on bathymetric maps of varying origin and quality (II, Vallinkoski et al. 2004). Likewise, the maximum growing depth was determined as the maximum of all transects studied (12 – 15 per lake). Although 15 transects is adequate for detecting the overall species composition (I), it may not be enough to reliably estimate the maximum colonization depth (MCD); Spears et al. (2009) found that 15 – 20 replicate transects were needed to get a reliable estimate of MCD. Therefore, additional analysis will be needed on the sufficient survey effort for determining the maximum potential growing depth of helophytes and nymphaeids in humic lakes (Vallinkoski et al. 2004). In addition, other factors such as bottom quality and wave exposure affect macrophyte abundance (Riis & Hawes 2003, Feldmann & Nöges 2007) and should be considered in order to achieve a more reliable assessment of the potential colonization area of helophytes and nymphaeids. This could be achieved through more detailed spatial modeling (Jensen et al. 1992, Lehmann 1998), which would take into account e.g. fetch, in addition to depth, to predict the probability of occurrence of vegetation.

The ecological classification based on current monitoring data of macrophytes was largely in accordance with the assessment based on other biological elements, phytoplankton, macrozoobenthos and fish, as well as with sediment records of diatom, phytoplankton and chironomid assemblages in the eutrophic Lake Kirmanjärvi. All indicated significant changes in the composition of lake biota (VI). The 'moderate' contemporary status of macrophytes - with no palaeobiological proxy to compare - agreed with the overall classification. There were, however, marked differences between metrics, MTCs (TT50, PMA) yielding stricter classification results than TI. This is in line with the general finding that measures of taxonomic composition are more sensitive than stress-specific metrics (IV). The sediment record also revealed Lake Kirmanjärvi's naturally high productivity; naturally eutrophic lakes have been identified in other paleolimnological studies (e.g. Räsänen et al. 2006) as well, and they form a unique type in the Finnish typology system (Vuori et al. 2006). The ecological classification of naturally eutrophic lakes is generally problematic, as few pristine sites remain in catchments with nutrient rich soils ideally suitable also for agriculture and settlement. For macrophytes, some historical data are available, which has made possible the reconstruction of reference conditions (Madgwick et al. 2011) for naturally eutrophic, turbid lakes (Alahuhta et al. 2009b). However, the shift from 'small humic' to 'naturally eutrophic' lake type did not affect the overall status classification in this case, as differences in the type-specific reference conditions between these particular lake types are minor (VI).

3.4 Implications for lake management (III, V, VI)

In the shallow, humic, hypertrophic Lake Luupuvesi - undergoing vegetation management efforts at time of the study - the long-term (from 1953 to 1996) increase of helophyte vegetation had been significant, based on comparison of historical and current aerial images (III). The vegetation had since (from 1996-2001) reduced, presumably mostly due to harvesting (III). The biomass model for *Schoenoplectus lacustris* in Lake Luupuvesi allowed the lake scale change in the plant biomass between 1996 and 2001 to be placed at 511 tonnes (III). The phosphorus content of the biomass removed from the system was estimated to be ca. 3 % of the influx and likely not to have any significant effect on the nutrient balance of the lake (III). The remote sensing methods, however, provided an applicable means for quantifying and illustrating the long-term changes caused by lake vegetation management.

Compared to the whole catchment, the land use directly adjacent to the lake shoreline had a stronger effect on the status of macrophytes and water quality, judged by the GAMs and correlation analyses (V). Land use and water quality are inevitably closely related (Johnson et al. 1997, Carpenter et al. 1998, Ecke 2009) and water quality is widely known to be the primary determinant of macrophyte composition (Toivonen & Huttunen 1995, Lacoul & Freedman 2006). The detected scale dependency may relate to direct anthropogenic impacts such

as direct nutrient leaching and morphological alterations of the littoral zone mediated via sediment loading and siltation. On the other hand, especially in interlinked lakes, nutrients bound to particles are sedimented along the system and only a portion of land use-derived diffuse pollution reaches the downstream lakes (Kratz et al. 1997), a fact which indirectly amplifies the importance of the nearest catchment as a determinant of lake status. Some previous studies have also shown stronger effects of the nearest buffer zones than the whole catchment on macrophyte status (Pedersen et al. 2006, Akasaka et al. 2010), while lack of a difference between scales has also been reported (Sass et al. 2010). The effect may also be metric specific and related to e.g. the species pool used (III). While there is obviously correlation between the land-use of different catchment scales, the results point to the importance of the near-shore area in mitigating the deteriorative effects of anthropogenic land-use.

Monitoring data of Lake Kirmanjärvi showed changes in the concentration of epilimnetic phosphorus and chlorophyll-*a* (proxy for phytoplankton biomass), coinciding temporally with intensive fishing and other management efforts (VI). Biomanipulation has proved an effective lake management tool (Mehner et al. 2002) and the positive changes in lake status detected suggested the intensive fishing effort to be effective in this case as well. Hypolimnetic oxygenation may have enhanced the effects of biomanipulation by decreasing the release of nutrients during hypoxic conditions (Kauppinen 2006), a conclusion supported by the detected increase in the phosphorus concentrations of recent sediments. The study demonstrated that reaching the goal of "good" ecological status may be attainable in this case, but requires further effective in-lake and catchment management efforts along with frequent monitoring to validate the efficacy of management measures.

4 CONCLUSIONS

A variety of practical implications considering the use of macrophytes in lake status assessment can be derived from the current study, the primary ones being methodological. The main advantage of the transect survey method over a phytolittoral inventory is that information on the zonation of the vegetation can be obtained. However, when comprehensive taxonomic data are needed – e.g. for biodiversity surveillance purposes – the information is most readily and cost-effectively obtained through a phytolittoral inventory, at least in moderate sized lakes. The reliability of both survey methods necessitates surveyor training and quality assurance. Most importantly, for ecological classification purposes, both within and among-method variation needs to be taken into account, preferably by basing the establishment of reference conditions and assessments on homogenous data.

Stressor specific macrophyte metrics, while commonly utilized and less prone to method-related variation, are largely redundant compared to general measures of taxonomic composition. Moreover, a clear distinction should be made between metrics of system state and indicators of stressors or human pressures. General measures of taxonomic composition should be the primary tools in classifying ecological status based on taxon occurrence. SSIs in turn, should be used as secondary variables, in assisting to identify the likely causes of the changes in community composition, and to assist in making correct management decisions. The conclusion extends to the current Finnish macrophyte classification system, in which the phosphorus-specific TI is used along with two MTCs in a multimetric index. This may bias the assessment system into more sensitively detecting trophy-related than other – e.g. hydromorphological – impacts.

The good performance, as measured by statistical criteria, and ecological relevancy of the MTCs lays ground for the development of continuous modeling, instead of categorical typology, in establishing reference conditions for macrophytes. In addition to metrics based on taxonomic composition, quantitative abundance metrics call for further studies to increase their accuracy. Remote sensing is increasingly used in studying aquatic vegetation allowing

more comprehensive and cost-effective assessment and monitoring compared to field surveys. As demonstrated here, remotely sensed data provide possibilities not only for long-term monitoring of individual water-bodies, but even for reference condition based status assessment of macrophytes. However, the rapid advances in sensor and image analysis technology call for further study before operational applications are at hand.

The study demonstrated that choosing the most influential catchment scale for the evaluation and mitigation of anthropogenic impacts on lacustrine biota may be significant in conserving and restoring ecological quality, as required by the legislation. Likewise, methods allowing long-term changes in lake biota to be analyzed can be helpful in setting realistic goals for restoration and management. Palaeobiological data can provide a viable – and in some cases even more informative – alternative to the typology-based method of assessing lakes status. The palaeolimnological approach is especially useful in validating the deterioration of ecological status, and when there are too few pristine sites left to allow spatially based reference conditions to be set such as in the case of naturally eutrophic lakes. Similarly, aerial photograph interpretation can provide important insights into long-term changes in vegetation abundance through the use of historical data.

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YHTEENVETO (RÉSUMÉ IN FINNISH)

Vesimakrofyytit boreaalisten järvien tilan arvioinnissa ja seurannassa

Viimeaikainen vesien hoitoa ja suojelua koskeva lainsäädäntö on nostanut vesiekosysteemin rakenteen tärkeimmäksi pintavesien tilan arvioinnin kriteeriksi; vesien hyvä ekologinen tila on lainsäädännössä asetettu tavoite. Järvissä ekologinen tila määritellään keskeisten eliöryhmien – kasviplanktonin, vesikasvien, päällislevästön, pohjaeläinten ja kalojen – yhteisökoostumuksen ja runsaussuhteiden perusteella. Järvien vesikasveja eli makrofyyttejä on tutkittu Suomessa jo pitkään, ja eliöryhmä onkin hyvin tunnettu. Vesimakrofyyttien seurantamenetelmät eivät kuitenkaan ole vakiintuneita eikä vesikasveja ole systemaattisesti käytetty järvien tilan arvioinnissa. Tämän vuoksi soveliaimpien seurantamenetelmien ja tilamittareiden arviointi ja käyttöönotto on vaatinut laajaa tutkimus- ja kehitystyötä viimeisen vuosikymmenen kuluessa.

Tämän väitöstutkimuksen ensimmäisenä tavoitteena oli vertailla kahta yleisesti sovellettua järvikasvillisuuden maastotutkimusmenetelmää: linjamenetelmää sekä koko litoraali-alueen kattavaa lajistokartoitusta. Menetelmien välisiä eroja arvioitiin sekä suoran maastovertailun avulla että epäsuorasti käyttäen molemmilla menetelmillä kerättyjä vertailujärvien aineistoja. Koko litoraalin kartoituksella havaittiin enemmän lajeja lyhyemmässä ajassa kuin linjamenetelmällä. Viidentoista linjan tutkimisella havaittiin n. 80 % kaikista järvellä tavatuista lajeista, mutta ekologisessa luokituksessa käytettävien muuttujien luotettavuuden määrittämiseen riitti 8–10 linjan tai 40–60 %:n rantaviivaosuuden tutkiminen. Eri menetelmät tuottivat poikkeavia yhteisökoostumusta kuvaavien muuttujien arvoja. Tämän vuoksi seuranta-aineistojen menetelmällinen yhdenmukaisuus on tärkeää ekologisen tilan mittareita sovellettaessa. Koska linjamenetelmä tuottaa lajikoostumuksen lisäksi tietoa kasvillisuuden vyöhykkesyydestä, sitä voi suositella ekologisen tilan arviointiin, kun taas koko litoraalin kartoitus soveltuu kustannustehokkaampana lajihavaintomenetelmänä esim. biodiversiteettiseurantoihin.

Tutkimuksessa arvioitiin myös kaukokartoituksen soveltamismahdollisuuksia vesikasvillisuuden seurantamenetelmänä. Tutkimuksessa käytettiin digitaalisia vääräväri-ilmakuvia, joista saatavaa informaatiota vesikasvillisuuden koostumuksesta luontaiselta tyypiltään ja nykyiseltä rehevyystasoltaan erilaissa järvissä analysoitiin. Lisäksi käytettiin mustavalkoilmakuvia 1950-luvulta kasvillisuuden pitkäaikaismuutosten arviointiin. Työssä sovellettiin automatisoituja kuva-aineiston luokittelumenetelmiä ja havaittiin, että tietoa saatiin pääosin ilmaversoisista ja kelluslehtisistä lajeista elomuototasolla, vain harvoin yksittäistä lajeista tai uposkasvillisuudesta. Tämä luokitustaso soveltuu kuitenkin hyvin mm. kunnostustoimenpiteistä aiheutuvien muutosten analysointiin, kuten todettiin matalan, hyvin rehevän ja runsashumuksisen järven kasvillisuuden pitkäaikaismuutosten seurannassa: tutkimuksessa havaittiin paitsi kasvillisuuden runsastuminen pitkällä aikavälillä, myös vasteet kunnostustoimenpiteille eli kasvillisuuden niitolle. Paikkaan sidottu ilmakuva-aineisto mahdollisti

myös tiedon yhdistämisen muuhun kasvillisuuden esiintymistä selittävään ympäristötietoon, kuten järven syvyyttä kuvaaviin korkeusmalleihin. Ilmakuvatulkittu kasvillisuus ja syvyystieto yhdistämällä määritettiin ilmaversoisten ja kelluslehtisten vesikasvien runsausmuuttuja, kasvillisuuden peittämän litoraalilin osuus, jota voitaisiin soveltaa kaukokartoitusaineistosta johdettuna vesikasvillisuuden tilamuuttujana.

Ekologisessa tila-arvioinnissa käytettävien mittareiden tulisi kuvata eliöiden yhteisökoostumusta ja runsaussuhteita. Mittareiden avulla verrataan ihmistoiminnan muuttamien vesien eliöyhteisöjä vertailuvesistöjen eli lähellä luonnontilaa säilyneiden vesien eliöyhteisöihin. Tämän tutkimuksen yhteisöanalyysien perusteella vertailutilaa edustavien ja ihmistoiminnan kuormittamien järvien vesikasvillisuuden lajikoostumus erosi toisistaan. Yhteisöjen eroja mittavien muuttujien määrittelemiseksi on yleisesti sovellettu kahta toisistaan periaatteeltaan poikkeavaa lähestymistapaa: voidaan käyttää joko eliöiden tunnettuja vasteita erilaisiin paineisiin tai yleisiä, lajikoostumuksen muutoksiin perustuvia mittareita. Väitöstyössä verrattiin painespesifien, lajien indikaatioarvoihin perustuvien mittareiden sekä yleisten, taksonikoostumuksen eroihin perustuvien mittareiden soveltuvuutta kahden yleisimmän ihmistoiminnan paineen, rehevöitymisen ja vedenpinnan säännöstelyn, aiheuttaman muutoksen arvioinnissa. Oletusten mukaisesti yleiset kasvillisuuden koostumuksen mittarit reagoivat sekä rehevöitymiseen että säännöstelyyn, mutta painespesifit pääsääntöisesti vain siihen paineeseen, joita vasten ne oli kalibroitu. Herkin yleisistä muuttujista ilmensi muutosta järvien kasvillisuudessa lähes yhtä usein kuin heikointa tilaa ilmaiseva painespesifi mittari. Yleisiä taksonikoostumuksen mittareita tulisikin lähtökohtaisesti käyttää ekologisessa tilanarvioinnissa, mutta painespesifit muuttajat voivat olla avuksi muutosten syiden analysoinnissa.

Väitöstutkimuksessa vertailtiin myös vesikasvillisuuden ja muiden eliöryhmien ilmentämää nykytilaa rehevän järven pohjasedimenttiin kerrostuneiden eliöjäänteiden (piilevät, surviaissääsket ja sinilevien keustosolut) avulla rekonstruoitua eliöstön muutoshistoriaan. Tutkimuksessa todettiin kyseisen järven olleen jo luonnontilassa rehevä mutta rehevöityneen edelleen ja olevan ekologiselta tilaltaan heikentynyt. Nykytilaa kuvaavat muuttajat ilmensivät pääosin samansuuntaista muutosta. Vesikasvillisuuden osalta tässäkin tapauksessa taksonikoostumukseen perustuvat mittarit olivat herkempiä kuin ravinteisuutta ilmentävä mittari.

Ihmisperäisen maankäytön – erityisen maanviljelyn – sijainnin merkitystä vesikasvillisuuden tilan määrääjänä tutkittiin tässä työssä suhteuttamalla vesikasvillisuuden tilaa kuvaavia mittareita eri etäisyydelle järven rantaviivasta ulottuvan alueen (100, 300, 500 m sekä koko valuma-alue) maankäyttöön. Useita ympäristömuuttujia huomioineen mallinnuksen avulla havaittiin rantaviivaa lähempänä olevien vyöhykkeiden maankäytön selittävän vesikasvillisuuden tilamuuttujia paremmin kuin koko valuma-alueen maankäytön. Vaikutukset olivat selkeimmät tilamuuttujissa, jotka huomioivat koko kasvuyhteisön koostumuksen, mutta heikommat vain vedenlaatuun reagoivalla spesifillä muuttujalla. Kokonaisuutena järven vedenlaatu oli tärkein vesikasvillisuuden tilan se-

littäjä ja maankäyttö heijastuu kasvillisuuteen pääosin epäsuorasti vedenlaadun kautta. Kuitenkin myös suorat lähellä rantaa tapahtuneet maankäytön muutokset ja ravinnevalumat voivat muuttaa vesikasvillisuutta. Lähimmän valuma-alueen erityisellä merkityksellä tilan määräytymisessä voi olla vaikutusta järvi-en hoito- ja kunnostustoimenpiteiden kohdentamiseen. Pitkäaikaisten muutosten analysoinnin mahdollistavat menetelmät, kuten paleolimnologia ja historiallisten ilmakehä-aineistojen tulkinta, voivat tuoda merkittävää lisätietoa ihmis-peräisten muutoksien laajuuden arviointiin ja realististen kunnostustavoitteiden asettamiseen.

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ORIGINAL PAPERS

I

A COMPARISON OF TWO METHODS FOR SURVEYING AQUATIC MACROPHYTE COMMUNITIES IN BOREAL LAKES: IMPLICATIONS FOR BIOASSESSMENT

by

Antti Kanninen, Veli-Matti Vallinkoski, Jarkko Leka, Timo J. Marjomäki, Seppo
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A comparison of two methods for surveying aquatic macrophyte communities in boreal lakes: Implications for bioassessment

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ABSTRACT

To compare the performance of two common methods for surveying boreal lake aquatic macrophyte communities (the transect survey and the phytolittoral inventory) in lake bioassessment, we studied five small humic lakes with both methods. In addition to this pairwise comparison, we contrasted available independent sets of reference lake data of either methodological origin. Specifically, we compared the observed species richness and a number of ecological quality (community) metrics and their variability both between the methods and within them in relation to sampling effort and operator. With the phytolittoral inventory, more taxa (mean $38.8 \pm \text{s.d. } 12.1$) were observed than with transect survey (29.8 ± 6.5) in a shorter time. Ecological quality metrics relying on taxonomic composition were sensitive to variation both between (reference lake data: *t*-test, $p < 0.001$) and within methods (metric values varied with sample size), whereas a trophy-specific metric based on indicator species was more robust in this respect (ref. lakes: *t*-test, $p = 0.788$; minor variation with sample size). The two methods did not differ in among-observer variation (CV of community metrics 5.0–8.7), which was, as indicated by an NMDS ordination, relatively low compared to among-lake variation. Overall, the results highlight the importance of establishing reference conditions using unified methods, which are also consistent with those used in the macrophyte-based ecological assessments. Even though the more cost efficient phytolittoral inventory method is more suitable for biodiversity surveys focusing on the overall species richness and rare species, we suggest that the transect survey better meets the requirements of ecological status assessment, as it produces information also on the zonation of the littoral vegetation.

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1. Introduction

Recent legislative stipulations have made direct measurements of the biotic condition of ecosystems the basis of water policy and management (European Commission, 2000). Underlying prerequisites for valid biological assessment schemes are consistent biological survey and sampling protocols. For instance, the European Water Framework Directive (WFD; European Commission, 2000) requires ecological classification to be conducted using standardized methods. Even though the recently established CEN standard for monitoring aquatic macrophytes in lakes (CEN, 2006) recommends a transect-based method, considerable variation exists in the macrophyte monitoring protocols (Gunn et al., 2006;

Kolada et al., 2009). Despite some type of transect survey (TS) being used by at least 8 European countries as the primary monitoring method (Gunn et al., 2006), a so-called phytolittoral mapping is used in several countries as a WFD-compliant monitoring method, either supplementary to transect surveys or as the sole method (Kolada et al., 2009). Moreover, floristic inventories (hereafter referred to as phytolittoral inventories, PI) and/or mappings, have been successfully applied to investigating patterns in biodiversity, community composition and environmental relationships of macrophytes (Capers et al., 2010; Heegaard et al., 2001; Mäkelä et al., 2004; Toivonen and Huttunen, 1995), as well as to reveal long-term changes in macrophyte communities (Ranta and Toivonen, 2008; Rintanen, 1996; Toivonen, 1985).

Development of commonly applicable assessment tools and intercalibration of assessment results across Europe (Hering et al., 2010; Poikane et al., 2011) require large datasets with a wide geographical extent (G.-Tóth et al., 2008; Moe et al., 2008). Currently, these data are necessarily from varying sources and of contrasting methodological origins. These differences in the datasets are often

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acknowledged as a source of variation (Moe et al., 2008), but the effects of the differing methods on the results are rarely quantified. As the WFD-compliant assessment relies on the reference condition approach (e.g. Hawkins et al., 2010), the quality and quantity of the reference data are of utmost importance and, therefore, possible effects of varying survey methodology on the reference conditions should be evaluated in particular.

There are several sources of variability in biological monitoring results and, consequently, uncertainty in the quality classifications (Clarke and Hering, 2006). Variability can be categorized to spatial (e.g. within-lake) and temporal (e.g. inter-annual or seasonal) variation and measurement error (e.g. sampling variation). Within-lake variability in macrophyte surveys arises from the high heterogeneity of the littoral habitat in exposure, slope and bottom substrate, and subsequent variation in macrophyte species distribution (Hellsten, 2001; Lacoul and Freedman, 2006). Therefore, within-lake variation is likely to be an important source of imprecision also in assessment metrics based on the composition of macrophyte communities, and is dependent on sampling effort (e.g. number of transects).

As macrophyte surveys are conducted in the field by observers, measurement error or sampling variability equals to observer-related (operator) variation, arising from among-observer differences in species detection, identification and abundance estimation. As the operator variability may be a significant source of variation and differ between methods, it should be considered in a methodological comparison. The reported observer-related variability in river macrophyte surveys ranges from relatively minor (Staniszewski et al., 2006) to significant (Hurford and Lansdown, 2010; Pentecost et al., 2009). However, in lake macrophyte surveys the observer-related variation has seldom been reported (for a wetland case study see Croft and Chow-Fraser, 2009) and, generally, published methodological research is relatively rare (but see e.g. Capers, 2000; Gunn et al., 2010; Rodusky et al., 2005; Spears et al., 2009).

The purpose of this study was to compare two widely applied aquatic macrophyte survey methods: a transect survey and a phytolittoral inventory with respect to their accuracy and precision in describing the species composition and status of macrophytes in small boreal lakes. The specific questions addressed are:

- (1) Do the methods produce comparable information on species composition?
- (2) Are the metrics used in status assessment of lake macrophytes comparable across the methods?
- (3) Do the methods differ in cost-efficiency and effort needed to reach reliable estimates of macrophyte metrics?
- (4) Do the methods differ with respect to among-observer variation?

2. Materials and methods

2.1. The transect method

In the transect survey (TS) method ('main belt transect method' as first described by Leka and Kanninen, 2003), observations of macrophyte species are made along a 5 m wide transect, perpendicular to the shoreline and extending from the upper eulittoral to the outer limit of macrophyte vegetation. Observations are made in shallow water by wading, using a hand-held rake and a bathyscope. In deeper water, observations are made from a boat, using a bathyscope, a long-shafted rake, Luther-type rake and, as a supplement, an underwater drop-camera to determine the abundance and depth limit of isoetid vegetation. The transect is divided to zones (units of observation) according to the dominating

life-form or species and in each zone the species are recorded and their frequency and abundance estimated on a continuous percentage scale. Frequency is defined by 'virtually' dividing an observation unit into 100 subunits, and estimating the percentage of subunits occupied by the species. Abundance is the mean estimated coverage of the stands of the species in these subunits. Each zone is positioned by measuring the water depth and the distance to the transect origin at both edges of the zone (for further details see Kuoppala et al., 2008).

The potential transect sites were initially determined according to Jensen (1977). The method produces a stratified random sample of shoreline points and the number of transects per lake is dependent on the complexity of the shoreline (Jensen, 1977). We selected 15 of these sites per lake to be investigated by the TS method as previous studies (Leka and Kanninen, 2003; Leka et al., 2003) had suggested that 15 transects were enough to capture more than 80% of all taxa. The average combined length of the survey transects was 751 m (\pm s.d. 184 m) per lake. Sites were selected to represent the whole lake spatially and all different habitat types present (from shallow, gently sloping shores to open and steeper shores).

2.2. The phytolittoral inventory method

For the phytolittoral inventory (PI), the lake shoreline (mean length $6840 \pm$ s.d.1375 m) was divided into five stretches equal in length (20% of the total shoreline length) using GIS with 1:20,000 scale shoreline data. Hard-copy maps were used in the field to localize the stretches. The survey team traveled each stretch mainly by a rowing boat, supplemented by regular wading in shallow water, and occasional examination of the shore vegetation by walking along the shoreline. Submerged vegetation was surveyed with the help of rakes and a bathyscope, occasionally viewing with an underwater drop-camera. Floating and emergent vegetation was also recorded. For each stretch of shoreline and detected taxon an estimate of frequency was determined as the estimated percentage of shoreline at which the species was present and abundance as the average coverage of the stands.

2.3. Pairwise comparison of the methods

For direct comparison of the two survey methods, five lakes in eastern Finland (Table 1) were surveyed using both methods. In two lakes, both surveys were carried out in the same year (2005), whereas in the three other lakes, the TS and PI surveys were conducted in 2003 and 2005, respectively. The lakes represented both high (reference quality) and lower ecological status classes (Table 1).

For both methods, the time taken to survey each transect and stretch of shoreline was recorded to allow comparisons of 'cost-efficiency'. The surveys are almost exclusively conducted in the field, as only identification of some bryophyte species may require microscopic examination in the laboratory. The technical equipment needed is similar for both methods. Therefore, the working time spent conducting the survey by the field team is an adequate measure of the costs.

To allow for comparison between methods, abundance values were obtained by calculating for each taxon a vegetation index (Ilmavirta and Toivonen, 1986), which combines the frequency and abundance values to a single metric. Prior to calculation the frequency and coverage values were transformed to a 1–7 Norlin scale (Ilmavirta and Toivonen, 1986). The frequency value was the share of transects with the species present, and the average frequency across five shoreline stretches in the TS and PI methods, respectively. Abundance was the mean coverage of stands in both methods. As the frequency and coverage estimates are not quantitative, relative abundances (RA) based on the vegetation index were

Table 1

Typology-related morphometric and water quality properties, together with ecological status of the lakes used for pairwise comparison of the survey methods. A = Comparison of the TS and PI methods and analysis of observer-related variability in PI, B = Analysis of observer-related variability in TS. The ecological status is the result of the provisional Finnish classification scheme, based on available data on phytoplankton, macrophytes, macrozoobenthos and fish from 2000 to 2007.

Lake	N (WGS84)	E (WGS84)	Area (ha)	Mean depth (m)	Altitude (m)	Color (mg Pt/L)	Alkalinity (mmol/L)	Ecological status
A	Oinasjärvi	62.5031	26.5001	238	4.6	123	0.182	High
	Varpanen	63.3584	27.7746	81	3.1	102	0.187	Moderate
	Suurijärvi	62.5475	27.9089	117	2.4	82	0.208	Good
	Syväjärvi	62.6478	26.7031	65	2.9	97	0.353	Moderate
	Valkeinen	63.1167	27.0395	144	3.3	141	0.106	High
B	Mataroinen	62.4282	26.8534	152	4.4	141	0.09	High
	Ahveninen	62.4804	26.9002	633	6.1	111	0.105	Good
	Liesjärvi	63.0253	27.0322	315	1.7	107	0.175	Good
	Härkäjärvi	62.1096	26.8327	571	7.1	103	0.13	Good
	Pieni-Myhi	62.5096	26.8378	238	5.8	97	0.13	Good
	Haukijärvi	63.0428	27.0794	154	3.2	131	0.128	High

used in all further analyses utilizing taxon abundance. For the Lake Syväjärvi PI, some missing coverage values disabled RA calculations with the whole lake PI data.

To compare the two methods in variation related to observer, three different field teams repeated surveys in the same localities. For the TS method, independent repetitive surveys of 4–5 transects per lake in 6 lakes (altogether 29 transects) were performed within five weeks in July–August 2004, during the optimal season for monitoring macrophytes (Table 1). For the PI method, three stretches of shoreline per lake in 5 lakes were repeated in July–August 2005. Each transect and stretch of shoreline was marked in the field by the first team to ensure repeatability of the location. Before the field campaign of each year, a one-day ‘intercalibration’ session was organized to review the methodology by all the field teams. Each field team consisted of two persons: a team leader with good expertise in lake macrophyte surveys was mainly responsible for species identification and abundance estimation; and an assistant made the field notes and supplementary observations of the vegetation. Altogether there were four team leaders involved in the field campaign.

2.4. Reference lake data

Data for an indirect comparison of the effects of survey method on the establishment of reference conditions were extracted from a national database, containing both TS and PI data. The TS data of the database have been primarily collected during 2002–2009 using the main belt transect method. The PI data originate from a study published earlier (Rintanen, 1996). The database – in which the lakes represent least-disturbed conditions (Stoddard et al., 2006) – has been used to establish type-specific reference conditions for aquatic macrophytes in the provisional ecological classification of Finnish lakes (Vuori et al., 2009). The categorical typology of Finnish lakes is primarily based on lake area, mean depth, DOC concentration (measured as color) and altitude. In addition, alkalinity and turbidity are used to separate naturally eutrophic and calcareous lakes (see e.g. Jyväsjärvi et al., 2011). In the macrophyte classification system, lakes are further sub-divided to northern and southern sub-types due to a latitudinal gradient in species richness (Heino and Toivonen, 2008).

Only those three lake types (shallow humic lakes = ShH, small humic lakes = Smh and small and medium sized oligohumic lakes = SmMeOH) represented by at least four lakes for both methods were included. Of the selected 29 lakes, 16 and 13 lakes had been surveyed with the TS and PI method, respectively (Table 2). The TS and PI reference lakes did not differ in morphometry, humus content or alkalinity but their geographic location was somewhat different; TS lakes were more north-eastern and spatially restricted than PI lakes (Table 2).

2.5. Numerical analysis

In the pairwise data, univariate macrophyte metrics were compared between the two methods. The metrics included total species richness, as well as species richness and relative abundance of functionally similar life-forms: emergent (incl. helophytes and upper littoral species), floating (floating-leaved and lemniids) and submerged (elodeids, isoetids, charophytes, ceratophyllids and bryophytes) species.

To evaluate the effect of sample size (number of transects in TS and percentage of shoreline examined in PI) on the detected species richness and classification metrics (see below), resampling techniques were used. The resampling was performed using customized procedures, as composite samples are used in classification metric calculations, which is complex. For the transect data, assuming that the population is well represented by the total sample of 15 transects, we formed 1000 random combinations (each with replacement) for each sample size of 1–15 transects using an Excel macro. For the PI method, the individual studied shoreline stretches are unique and thus cannot be resampled with replacement; therefore we manually formed the maximum number of unique combinations of 1–5 shoreline stretches (20–100% of total shoreline length). For each of these sample sizes we then calculated the median and the 90% probability interval of EQR to be plotted against the sample size.

We included in the analyses the three ecological classification metrics (TI, PMA, TT50) used in the initial Finnish macrophyte classification system (Vuori et al., 2009, for details of metrics calculations). The Trophic Index (TI) formulated by Penning et al. (2008) is based on division of hydrophytes to tolerant, indifferent and sensitive species based on their occurrence probabilities along a phosphorus gradient, and it can be calculated independently of reference communities. The other two metrics, Percent Model Affinity (PMA) and proportion of ‘type specific’ taxa (TT50), rely on the average relative abundance (RA) and probability of occurrence of taxa in

Table 2

Typology-related morphometric and water quality properties of the reference lake groups surveyed with the different methods. The variables (excluding easting) are those used in the Finnish lake typology system as categorical descriptors. *p*-Values are significances of among-group differences (based on nonparametric Mann–Whitney *U*-tests).

Variable	TS lakes (<i>n</i> = 16)	PI lakes (<i>n</i> = 13)	<i>p</i>
Area (ha)	127 ± 13.8	184 ± 30.6	0.400
Mean depth (m)	3.7 ± 0.41	3.1 ± 0.31	0.314
Altitude (m)	123 ± 8.9	104 ± 9.0	0.131
Easting (FNG)	3,534,371 ± 10,946	3,374,630 ± 21,614	<0.001
Northing (FNG)	6,932,430 ± 21,458	6,797,673 ± 23,519	<0.001
Color (mg Pt/L)	48 ± 7.3	37 ± 5.8	0.074
Alkalinity (mmol/L)	0.109 ± 0.013	0.125 ± 0.020	0.499

reference conditions, respectively. In the case of PMA, the average RA of taxa in reference lakes was used as taxon-specific expected abundance values to which observed RAs are compared (Novak and Bode, 1992). In the case of TT50, species with the estimated probability of occurrence >0.5 in the reference lakes are considered 'type specific' and their proportion of all species observed is used as the metric value. Ecological Quality Ratios (EQRs) are obtained by dividing the observed (O) metric value by the expected (E), represented by the average value of the metric in reference conditions.

To evaluate the possible effect of the survey method on classification outcome, the multimetric classification scheme (Vuori et al., 2009) was applied to the pairwise data. The reference values and class boundaries for each metric (TT50, PMA, TI) were obtained from the national reference database and values for the 'small humic lake' – type were used. The three metrics are first assigned to corresponding classes and the lake-specific status class is determined as the median class (see Alahuhta et al., 2009; Rask et al., 2011).

We measured the operator variability in both methods by the coefficient of variation (CV) of the lake-specific univariate metrics (number of species and classification metrics) among the surveys repeated by different field teams. Differences in the variation between the methods were tested using *t*-tests. When not normally distributed, the data were log-transformed prior to tests. A Non-metric Multidimensional Scaling (NMDS) ordination was conducted for the combined dataset of TS and PI repetitive surveys to evaluate if the distances between observers were different between the methods. Furthermore, a Bray–Curtis distance measure was calculated for each pair of observations and the difference of average distances between observers was compared between methods using a *t*-test.

In the reference data, we analyzed differences between the methods in two ways. Firstly, we evaluated the significance of the method as a grouping variable by an NMDS analysis on RA data of species. If differences between methods are significant (i.e. different methods produce systematically different observed community compositions), reference lakes of different methodological origin should occupy distinct areas in the ordination space; otherwise, only grouping by natural environmental gradients (e.g. lake types or geographical position) would be expected (cf. Herlihy et al., 2008).

Secondly, we evaluated the effect of survey method on the expected values of the classification metrics within each lake type by cross-validation. First, for each lake, we compared the observed metric values based on one survey method with the expected values based on lakes surveyed with the other method. Second, for within-method leave-one-out cross-validation, we compared the observed community of each reference lake to the other reference lakes of the same lake type surveyed with the same method. If method-related differences existed, the values of the differently cross-validated EQRs would differ. This was tested statistically using a pair-wise *t*-test.

All univariate statistics were performed in SPSS 18.0, the NMDS ordinations in PC-Ord 4.35 and calculations of Bray–Curtis dissimilarities in R, using the package 'vegan'.

3. Results

3.1. Effects of survey method in the pairwise comparison

In the pairwise comparison of methods, more species per lake were detected with the whole lake PI method than with a 15 transect TS (Table 3 and Fig. 1). The species richness (S) detected with 15 transects corresponded on average to 79% (\pm s.d. 9.1%) of the S detected with PI. The species accumulation curves showed that, on average, examination of 40% of the total shoreline length with the

Table 3

Univariate macrophyte metric values (mean \pm s.d.) in the pairwise comparison lakes ($n = 5$) for TS and PI methods. *p*-Values are significances of between-method differences based on pair-wise *t*-tests (significant differences highlighted).

Metric	TS	PI	<i>p</i>
<i>Number of taxa</i>			
Total	29.8 \pm 6.5	38.8 \pm 12.1	0.037
Emergent	12 \pm 3.4	15.6 \pm 4.5	0.018
Submerged	11.6 \pm 2.6	16.8 \pm 7.2	0.077
Floating	6.2 \pm 1.8	6.8 \pm 7.2	0.305
<i>Relative abundances</i>			
Emergent	50.3 \pm 6.9	44.3 \pm 6.7	0.199
Submerged	17.3 \pm 10.5	17.2 \pm 10.4	0.900
Floating	32.4 \pm 2.1	38.5 \pm 5.1	0.177
<i>Classification metric EQRs</i>			
TT50	0.78 \pm 0.18	0.72 \pm 0.18	0.012
PMA	0.88 \pm 0.09	0.82 \pm 0.15	0.220
TI+100	0.83 \pm 0.12	0.85 \pm 0.07	0.495

PI method was required to produce a similar S as with 15 transects (Fig. 1). The differences in S were mostly due to higher number of emergent species obtained by PI, as the numbers of submerged or floating species were not different (Table 3). Eighty percent of the total number of taxa detected with TS was reached after examining 5–8 (median 6) transects. In PI, 80% species saturation was reached after examining 40–60% (median 40%) of the shoreline.

The dissimilarities in detected species composition were mainly attributed to rare and scarce species: the average relative abundance (RA) of species detected by only one method ($n = 78$, mean RA 0.84 \pm s.d. 1.48%) was remarkably lower than the average RA of species detected with both methods ($n = 126$, 3.7 \pm 5.28%; Mann–Whitney *U*-test, $p < 0.001$; Appendix A). Submerged species (55% of cases) remained most frequently undetected by one or the other method; followed by emergent (34%) and floating (11%) species. The TS and PI methods produced comparable RA estimates of emergent, floating-leaved and submerged plants (Table 3).

EQR values of the classification metric TT50 were systematically lower when calculated from the PI data compared to TS data, but the EQRs of the other two metrics, PMA and TI, displayed no difference between the methods (Table 3). The EQR values of the three metrics calculated from resampled transect data also displayed contrasting responses to increasing survey effort: values of TT50 decreased and those of PMA increased, while the values of TI showed no systematic pattern (Fig. 2). The values of the metrics sensitive to survey effort (TT50 and PMA) remained relatively constant after approximately 8 transects in TS and 60% of shoreline in PI were surveyed (Fig. 2). In TS, the center most 90% of the resampled metric values of TT50, PMA and TI fell within one status class width after an average of 9, 7 and 6 transects were surveyed, respectively (Fig. 2 and Appendix B).

Application of the operative classification scheme to the pairwise data (Table 4) in most cases yielded similar results for both inventory methods for all metrics when the entire data were used, although in 27% ($n = 4$) of all individual cases ($n = 15$) the metric-specific classification differed by one status class between the methods. Differences were most prevalent in PMA, where the EQR values fell near class boundaries. For two lakes (Oinasjärvi and Suurijärvi), the two methods would have resulted in different overall classification results according to the multimetric classification scheme (Table 4).

3.2. Cost-efficiency of methods

The average effective time spent surveying one stretch of the shoreline (20% of the total shoreline length) in the PI method and one macrophyte transect using the TS method were 86 min

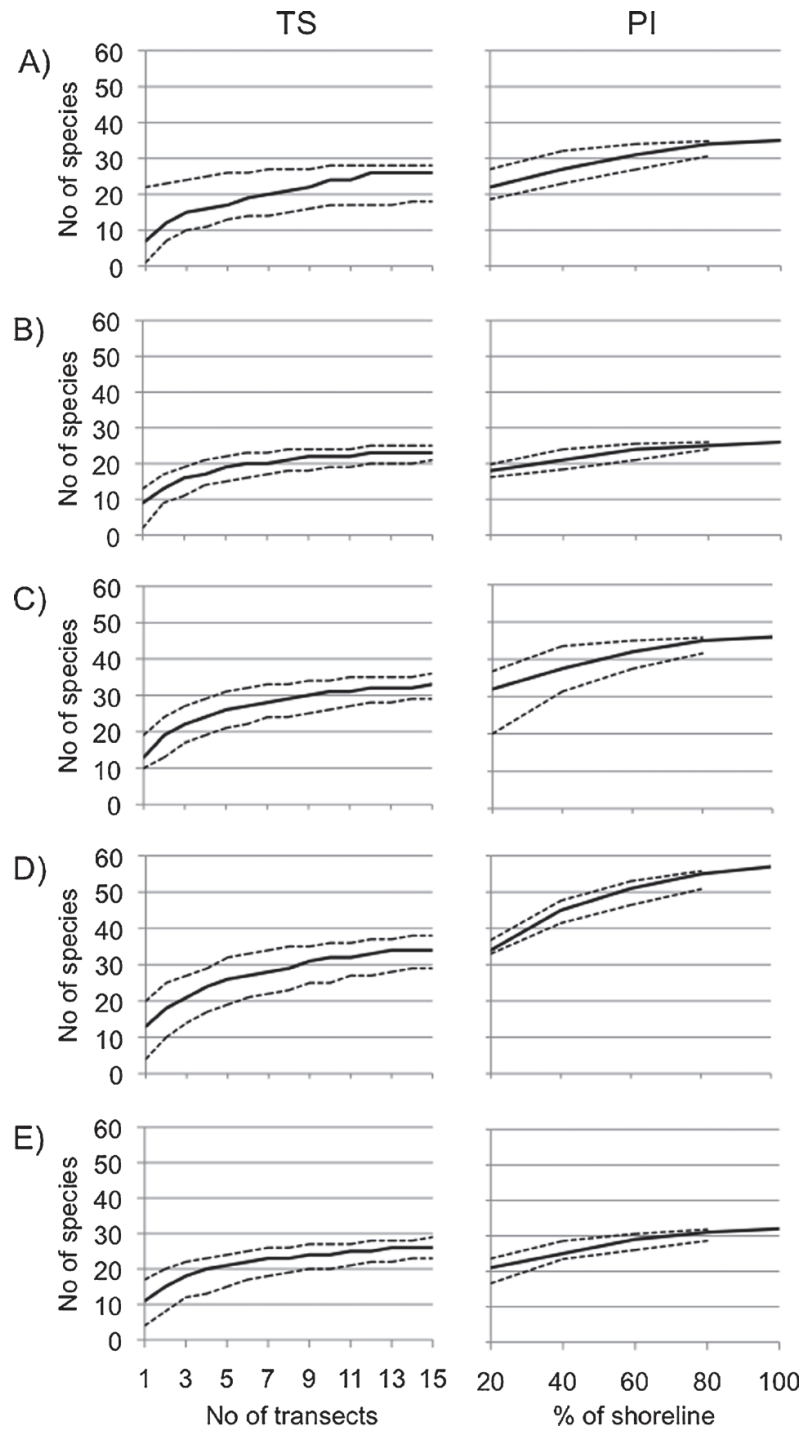


Fig. 1. Number of species detected with increasing survey effort based on resampling the TS and PI data of the study lakes; (A) Oinasjärvi, (B) Valkeinen, (C) Varpanen, (D) Suurijärvi and (E) Syväjärvi. Solid line represents the median and dashed lines the 5th and 95th percentiles of the resamples.

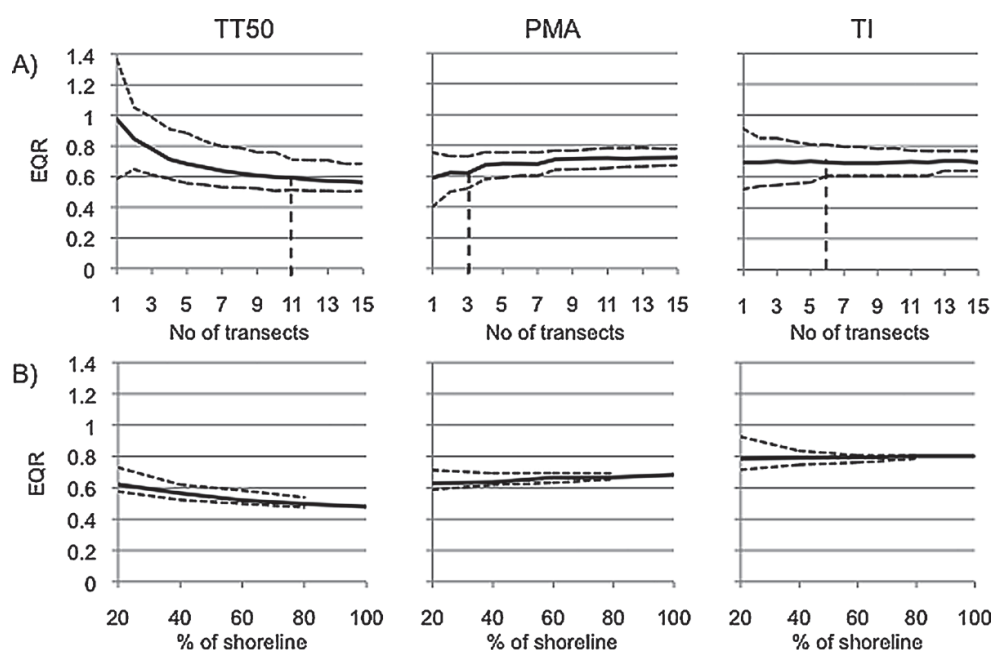


Fig. 2. Resampled ecological quality metrics in lake Suurijärvi. Response of the ecological quality metrics (TT50, PMA, TI) to increased survey effort based on resampling of the (A) TS and (B) PI data. Solid line represents the median and dashed lines the 5th and 95th percentiles of the resamples. In (A) the vertical dashed line represents the number of transects where 90% of the resampled EQRs fall within one ecological class. For other lakes, the curves are presented in Appendix B.

Table 4

The status classification results (Ecological Quality Ratio/Class) of macrophytes derived from the TS and PI surveys of five experimental lakes. The metric-specific EQR class boundaries for High/Good, Good/Moderate, Moderate/Poor and Poor/Bad status are 0.86, 0.65, 0.43 and 0.22 for TT50, 0.91, 0.68, 0.46 and 0.23 for TI and 0.91, 0.69, 0.46 and 0.23 for PMA, respectively.

Lake	TT50		PMA		TI		Lake median	
	TS	PI	TS	PI	TS	PI	TS	PI
Oinasjärvi	0.93/H	0.82/G	0.93/H	0.77/G	0.95/H	0.93/H	H	G
Valkeinen	0.97/H	0.95/H	0.97/H	1.03/H	0.96/H	0.91/H	H	H
Suurijärvi	0.53/M	0.48/M	0.77/G	0.68/M	0.69/G	0.80/G	G	M
Varpanen	0.72/G	0.65/G	0.91/H	0.81/G	0.78/G	0.83/G	G	G
Syväjärvi	0.75/G	0.68/G	0.80/G	n/a	0.75/G	0.77/G	G	G

(s.d. ± 28 min), and 36 min (s.d. ± 17 min), respectively. The effective total operating time per lake used to conduct the surveys did not differ significantly between methods (TS: 424 ± 53 min; PI: 538 ± 162 min; pair-wise *t*-test $p = 0.215$). The average species detection efficiency (effective working time to detect one species), calculated from individual transects ($n = 75$) and shoreline stretches ($n = 38$), was similar (*t*-test: $p = 0.735$) for TS (3.7 ± 2.2 min/species) and PI (3.4 ± 1.2 min/species).

3.3. Operator variability

The among-observer variability of all univariate metrics was similar between methods (Table 5). In general, the observer-related variability was lower in the species richness and classification metrics than in the relative abundance metrics (Table 5). There were no systematic differences between observers in any of the univariate metrics in either method (Table 5).

A two-dimensional NMDS ordination of the species data indicated that the grouping was stronger by lake than by the observer (Fig. 3). The distances among observers did not differ

Table 5

Coefficients of variation (CV) of macrophyte univariate metrics among three different observers in the transect survey (TS, $n = 6$) and phytolittoral inventory (PI, $n = 5$) methods and significance of differences between methods (p_{CV} , based on *t*-test). Significance of among observer difference in univariate metrics within each method (p_{TS} and p_{PI} values based on Friedman's test).

Metric	TS	PI	p_{CV}	p_{TS}	p_{PI}
<i>Number of taxa</i>					
Total	7.6 ± 2.2	7.6 ± 1.4	0.998	0.227	0.620
Emergent	7.6 ± 2.0	4.1 ± 1.3	0.192	0.210	0.424
Submerged	18.1 ± 3.4	15.9 ± 3.9	0.685	0.212	0.513
Floating	12.1 ± 4.1	14.3 ± 2.0	0.661	0.368	0.247
<i>Relative abundances</i>					
Emergent	16.1 ± 2.2	18.9 ± 2.9	0.458	0.311	0.819
Submerged	19.6 ± 2.8	25.3 ± 6.2	0.394	0.513	0.091
Floating	19.1 ± 2.7	22.3 ± 6.9	0.657	0.513	0.449
<i>Classification metric EQRs</i>					
TT50	6.0 ± 1.3	5.1 ± 0.9	0.558	0.846	0.449
PMA	5.0 ± 1.0	8.7 ± 2.7	0.270	0.513	0.549
TI + 100	5.8 ± 2.3	4.2 ± 0.7	0.551	0.676	0.074

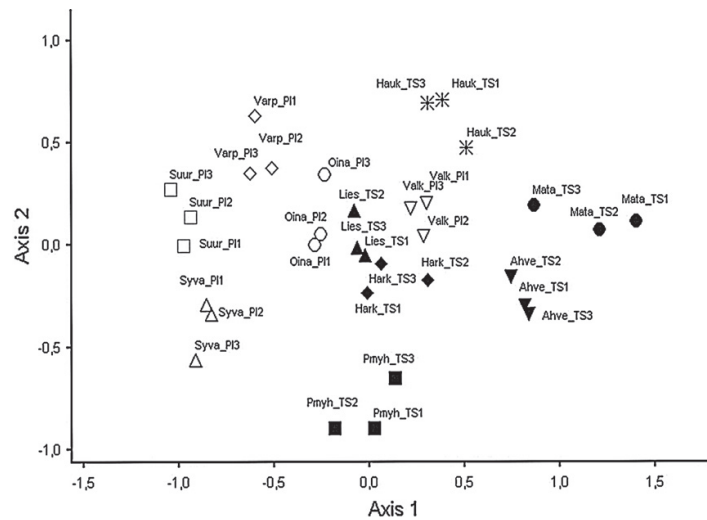


Fig. 3. An NMDS ordination bi-plot of the among-observer repeatability data. Closed and star symbols denote transect survey (TS) and open symbols phytolittoral inventory lakes (PI). Different symbols denote different lakes with repetitive surveys by three different field teams. Labels are formed from the lake name (abbreviation), method (TS or PI) and the number of repetition (1–3).

statistically between the methods (Bray–Curtis dissimilarity between observers on average 0.24 ± 0.17 and 0.28 ± 0.04 in TS and PI, respectively; t -test: $p = 0.095$). Lakes appeared to form distinct groups by the survey method, although this grouping cannot be attributed solely to method, as the TS and PI lakes also differed in status (Table 2).

3.4. Effect of survey method on definition of reference conditions of macrophyte communities

The reference lake data ($n = 29$) comprised 94 aquatic macrophyte taxa. A two dimensional NMDS-ordination (Euclidean distances) explained 81% of the original community variability (Bray–Curtis distances) among lakes. There was significant overlap among the three lake types (Fig. 4), although the typology factors, color ($r = 0.576$ with the axis 2), lake area ($r = -0.533$), mean depth ($r = -0.401$) and northing ($r = -0.360$ and $r = 0.339$ with axis 1 and 2, respectively) showed considerable correlations with the ordination axes (Fig. 4). Within each type, lakes surveyed using different methods formed distinct sub-groups and the TS lakes as a whole formed a more uniform group than the PI lakes (Fig. 4). Grouping of the PI lakes, better than grouping of the TS lakes, appeared to reflect the a priori typology.

The within-method and between-methods cross-validated EQRs differed for the species composition metrics, TT50 and PMA (Fig. 5), i.e. different methods did not produce comparable reference values of these classification metrics within the same lake type. However, the EQR values of the metric TI were independent of the methodological origin of the reference data (Fig. 5).

4. Discussion

4.1. Species richness

Higher species richness was detected with the PI than with the TS method. Partly, this certainly results from the larger area of vegetated littoral covered by the PI survey. The fundamental positive species richness–area relationship (Connor and McCoy,

1979) also exists for macrophytes among lakes (Rørslett, 1991; Mäkelä et al., 2004), and – as evidenced, e.g. by our species area curves – also within lakes. The difference in the number of detected species was most pronounced for emergent plants, which inhabit the upper littoral zone. These habitats are more representatively covered by the PI survey, whereby the surveyor travels parallel to the shoreline rather than studies a limited number of narrow, a priori randomly fixed transects perpendicular to the shoreline. Furthermore, in the PI method, the surveyor can use his/her expertise in localizing exceptional patches possibly rich

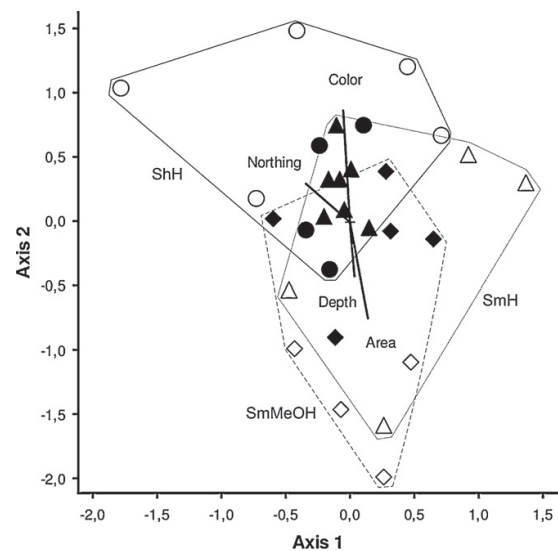


Fig. 4. An NMDS ordination bi-plot of the reference lake data ($n = 29$). Closed symbols denote transect survey and open symbols phytolittoral inventory data. Different lake types are outlined and represented by different symbols (ShH = shallow humic lakes, SmH = small humic lakes, SmMeOH = small and medium sized oligohumic lakes).

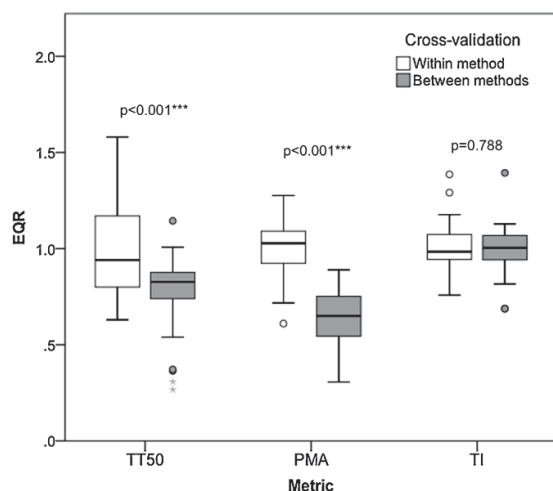


Fig. 5. Boxplots of 'within method' and 'between methods' cross-validated reference lake Ecological Quality Ratios of three aquatic macrophyte status assessment metrics. *p*-Values are based on *t*-tests and significant differences are flagged with asterisks.

in species or with locally rare species. This pattern was documented by Croft and Chow-Fraser (2009) in wetlands, where a non-random stratified sampling detected better than two random sampling procedures the wholesale species richness and rare species. Moreover, the authors found, similarly to our results, that the differences between methods could be attributed to emergent species, stands of which can be detected from a distance (Croft and Chow-Fraser, 2009). Spotting submerged species, on the other hand, is more dependent on their occurrence along the path taken by the surveyor. Generally, however, the differences between the survey methods in our study were limited to species with low abundance.

The survey effort also had a strong effect on the number of species detected. The number of transects we used in TS is premised on a previous study in which the rarefaction results based on 30–50 transects per lake suggested that 15 transects were enough to capture $82 \pm 4.9\%$ of all taxa (Leka and Kanninen, 2003; Leka et al., 2003). In line with this, the present study showing that 15 transects yielded approximately 80% of the total number of species found in an inventory of the entire shoreline. Hence, for capturing the general patterns of macrophyte community composition and diversity by a transect survey, 15 transects can be regarded sufficient for the type of lakes (area < 500 ha) included in these studies.

Despite no difference in the total operative survey time per lake or species detection efficiency at site level, more species per lake (Table 3) in less overall working time were found using the PI compared to TS, as conducting a TS includes time consuming transitions between transect sites. Therefore, in this exercise, the time-frame of one regular working day (including travel to and from the lake, launching a boat, breaks, etc.) enabled the survey of on average 5.7 transects (38% of total transects/lake) using TS and three shoreline stretches (60% of total shoreline/lake) using PI. On average, one additional working day was needed to perform a TS compared to a PI, which also resulted in more travel costs associated with TS. Hence, in reality, the PI was more efficient than TS in floral inventory, as the method produced a comprehensive species list in less overall time and with smaller costs.

4.2. Among-observer variability

The magnitude of observer-related variation was similar between the methods, but differed among the metrics. The non-significant differences among observers were somewhat unexpected, as our previous, smaller scale exercises (Leka, unpublished data) had indicated high operator variability for PI. The finding also contradicts some previous observations of high operator variability in river macrophyte surveys (Hurford, 2010). On the other hand, small variation has also been reported, especially among a more limited number of trained operators (Croft and Chow-Fraser, 2009; Staniszewski et al., 2006). Moreover, some metrics are likely to be more prone to variation than others (Staniszewski et al., 2006). In our study, we only had four different field teams, which hinders drawing any universal implications from the exercise. All observers had also undertaken a common intercalibration exercise prior to each survey season and were experienced in macrophyte surveys; a condition not always met in practical monitoring. However, the current results do highlight that even a relatively coarse survey method like the PI can be applied with reasonable repeatability by experienced and trained staff under clear guidelines. These issues can be tackled via quality assurance protocols that should be integrated into any operative monitoring program.

4.3. Ecological assessments

The metrics of ecological quality displayed differing methodological variation in both the pairwise comparison and the reference lake data; TT50 and PMA, derived from the entire community dataset, differed between methods and varied with survey effort. The dependence of the observed community composition on the survey method was also evident from the ordination of the reference lake data. On the other hand, the metric TI, based on indicator values rather than direct comparison of taxa lists, was robust against methodological variation.

According to the definition of the metric TT50, which was the most sensitive to methodological variation, the 'type-specific' taxa are those with detection probability of 50% or more in the reference sites of the corresponding lake type (Vuori et al., 2009, see also Aroviita et al., 2008). The negative curvilinear response of the EQR based on TT50 to increasing survey effort is due to accumulation of rare and scarce 'non-typical' species (with $p < 0.5$) lowering the overall proportion of type-specific taxa. Therefore, the metric TT50 is closely associated (inversely correlated) with macrophyte species richness and shares its sensitivity to methodological variability.

In the pairwise comparison, there was no evidence of differences in the abundance metrics (PMA and abundance of functional groups) between the methods, despite the slightly different ways of estimating abundance in TS and PI. It is plausible that in the reference data the differences between methods were greater, as in the version of PI used to collect those data the abundance estimation differed more from the TS method. However, the apparent differences between methods can at least partially be attributed to spatial and temporal variation as the geographical range and period of sampling differ between the PI and TS lakes. The clearly lower EQR-values derived from a small number of transects (<5) are likely to be partly a mathematical artifact: species frequency values of the vegetation index cannot be reliably calculated from a very small set of transects and, therefore, resulting relative abundances are imprecise, non-continuous and hence necessarily mismatching with the expected values based on more representative reference data.

TI was least sensitive to methodological variability, both between and within methods. Unlike TT50 and PMA, TI does not directly compare identities of observed species or their abundances

expected under reference conditions, but uses the indicator values, which can be similar among different taxa. The lesser variability may also be due to the fact that the TI uses only the submerged species, and only those which are relatively common, whereas methodological differences are mostly related to rare and scarce species. In our pairwise comparison data, altogether 32 index species were present, comprising 42% of all species (76) detected.

Nevertheless, sampling error or other error variation is only one aspect of bioassessment metric selection. More importantly, the metrics should conform to definitions of the 'ecological status' and reflect ecologically relevant anthropogenic impact on biota. The metric most prone to methodological influences, TT50, meets these criteria and has in previous studies also been reported to be most sensitive at separating impacted lakes from reference lakes and responding to pressure in boreal lakes (Kanninen et al., 2009; Alahuhta et al., 2012). On the other hand, the methodologically robust TI has been least sensitive at differentiating among status classes, especially in humic lakes, where the use of total phosphorus as a proxy of eutrophication is hindered by the naturally high amount bound in humic substances (Rask et al., 2011). While our data were too limited to make reliable inferences on metric performance, it was clear that TT50 was most and TI least variable among lakes (Table 4), lending support to contrasting patterns of sensitivity and methodological variability of the metrics. Further development of macrophyte assessment schemes should involve consideration of the sensitivity of the metrics to sampling variability.

4.4. Conclusions on optimal survey methodology

In a bioassessment system, a certain degree of variation in sampling, sample processing, identification and application of metrics is always present and leads to uncertainty in the classification results (Clarke and Hering, 2006). Although we did not seek to quantify the classification uncertainty related to methods, our results (Table 4) highlight that different methods may result in different classification outcomes. The variation in survey effort may have the same effect, as some metrics show not only decreasing variation in response to increasing effort, but a systematic increase or decrease in EQR values (Fig. 2). The results suggest that at least 8–10 transects or 40% of shoreline should be surveyed to reach relatively stable estimates of the ecological quality metrics in relatively small lakes covered by this study. One should, however, recognize that these figures are conditional to the reference data used. Moreover, they are not valid for larger lakes in which the within-lake variation is likely to be higher and which require more transects to obtain reliable metric values by TS, and where a PI covering even 40% of shoreline – which may be adequate – is probably unfeasible.

Our pairwise comparison showed higher efficacy of the PI compared to a TS in detecting the species richness of aquatic

macrophytes, especially emergent and locally rare species. Therefore, in lake surveys aimed at comprehensive biodiversity analysis, the PI method is more applicable. While a transect-based method is recommended by the current standards for monitoring according to the WFD (CEN, 2006), our results show that the less time-consuming PI might be a valid option at least in small lakes. For the utility of assessment metrics the survey effort should be standardized, however, and in PI, the suggested option is to use shoreline stretches of uniform length (McElarney and Rippey, 2009), rather than a fixed percentage of shoreline.

The drawbacks of using the PI include that it produces information only on taxonomic composition, but not any quantitative data on, e.g. the maximum colonization depth (MCD) or general zonation patterns (Hellsten, 2001; Partanen et al., 2006) of macrophytes. The MCD is widely considered a robust metric for lake ecosystem quality, corresponding to abundance of submerged vegetation, and in the transect method it is always measured. If the PI were used in routine monitoring, it should involve complementary records of MCD with the aid of underwater cameras or via other methods (see Spears et al., 2009). Considering these limitations, using TS as the basic method with the minimum number of transects producing reliable results is probably the most valid option; the species richness information produced can be supplemented with a PI when needed for biodiversity monitoring purposes (Janauer, 2002). As the survey method and effort within a method have clear effects on the ecological assessment metrics – especially when the whole community data are used, instead of only indicator species – assessments of lake ecological condition should only be made using methodologically homogenized data.

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Appendix A.

List of detected taxa in the direct pairwise comparison of the survey methods in five lakes. The life form (B=Bryophytes, Ce=Ceratophyllids, Ch=Charophytes, F=Floating-leaved, E=Elodeids, I=Isoetids, H=Helophytes, L=Lemnids, S=Shore species), Trophic Index value (S=sensitive, T=tolerant and I=indifferent), mean relative abundances (RA) in the transect surveys (TS) and phytolittoral inventories (PI) are given.

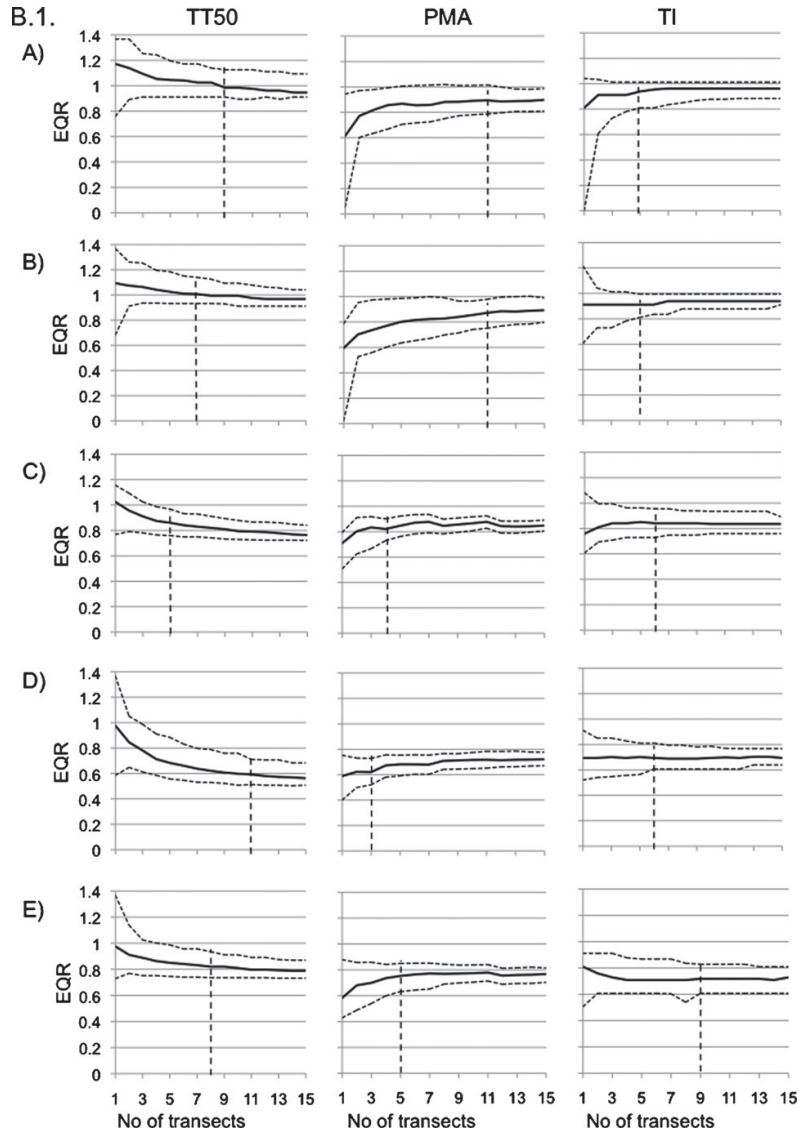
Taxon	Life-form	Trophic Index value	RA (TS)	RA (PI)
<i>Nuphar lutea</i> (L.) Sibth. & Sm.	F	I	15.17	23.88
<i>Phragmites australis</i> (Cav.) Trin. ex Steud.	H		7.23	11.17
<i>Carex rostrata</i> Stokes	S		7.26	9.15
<i>Comarum palustre</i> L.	S		11.27	4.40
<i>Carex lasiocarpa</i> Ehrh.	S		4.69	5.72
<i>Nymphaea alba</i> ssp. <i>candida</i> (C. Presl & J. Persl) Korsh	F	I	4.10	5.98
<i>Sparganium</i> sp. (<i>gra/ang/eme</i>)	F		6.02	3.25
<i>Calla palustris</i> L.	S		6.20	2.09
<i>Potamogeton perfoliatus</i> L.	E	S	3.66	2.48
<i>Equisetum fluviatile</i> L.	H		3.11	2.96
<i>Potamogeton natans</i> L.	F	I	3.06	2.81
<i>Lysimachia thyrsoiflora</i> L.	H		3.88	1.88

Appendix A (Continued)

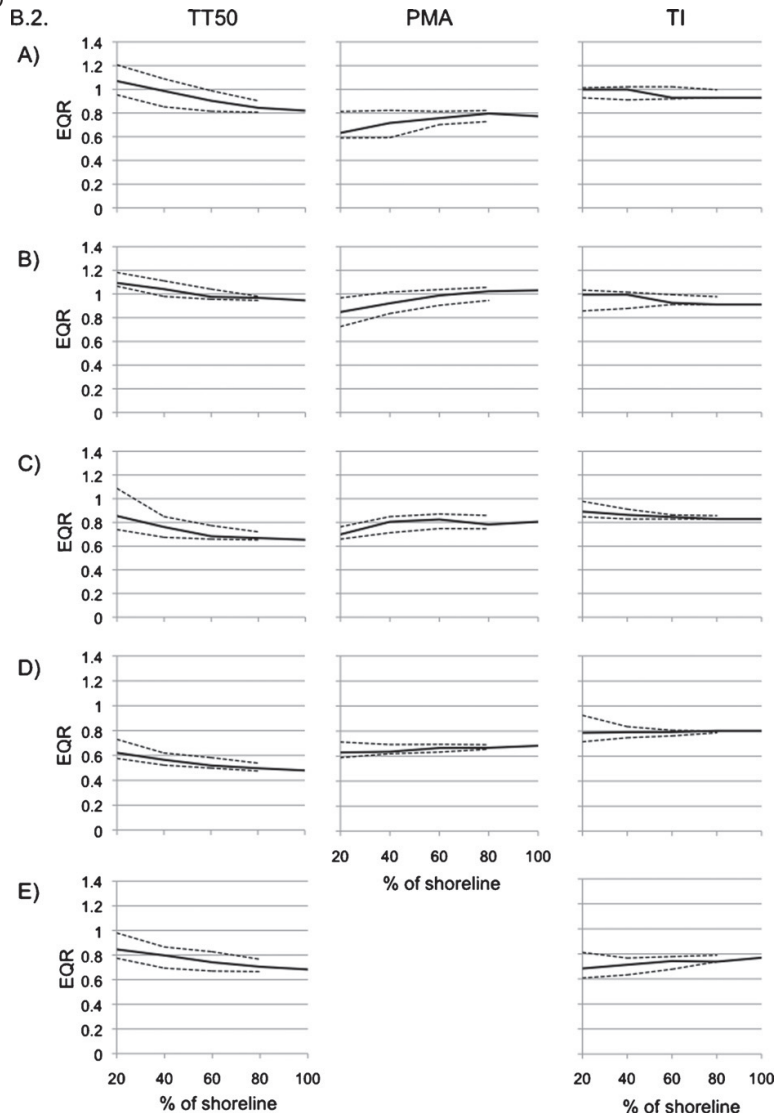
Taxon	Life-form	Trophic Index value	RA (TS)	RA (PI)
<i>Isoetes lacustris</i> L.	I	S	1.61	2.24
<i>Lobelia dortmanna</i> L.	I	S	1.20	2.24
<i>Wamstorfia trichophylla</i> (Warnst.) Tuom. & T. J. Kop.	B		2.17	0.87
<i>Nymphaea tetragona</i> Georgi	F	I	1.46	0.98
<i>Eleocharis acicularis</i> (L.) Roem. et Schult.	I	S	1.78	0.60
<i>Cicuta virosa</i> L.	S		1.43	0.82
<i>Caltha palustris</i> L.	S		1.44	0.81
<i>Persicaria amphibia</i> (L.) Delarbre	F	I	0.98	1.10
<i>Myriophyllum alterniflorum</i> DC.	E	S	0.37	1.59
<i>Ranunculus reptans</i> L.	I	S	0.58	1.34
<i>Schoenoplectus lacustris</i> (L.) Palla	H		0.69	1.19
<i>Isoetes echinospora</i> Durieu	I	S	1.24	0.54
<i>Elodea canadensis</i> Michx.	E	I	0.79	0.86
<i>Utricularia australis</i> R. Br.	Ce	S	0.30	1.08
<i>Carex acuta</i> L.	S		0.46	0.81
<i>Utricularia vulgaris</i> L.	Ce	I	0.87	0.13
<i>Eleocharis palustris</i> (L.) Roem. et Schult.	H		0.50	0.48
<i>Drepanocladus aduncus</i> (Hedw.) Warnst.	B		0.00	0.79
<i>Sagittaria natans</i> Pall.	F	I	0.65	0.12
<i>Hydrocharis morsus-ranae</i> L.	L	T	0.49	0.28
<i>Lycopus europaeus</i> L.	S		0.54	0.21
<i>Alisma plantago-aquatica</i> L.	H		0.20	0.50
<i>Utricularia ochroleuca</i> R. W. Hartm.	Ce		0.54	0.01
<i>Iris pseudacorus</i> L.	H		0.00	0.55
<i>Phalaris arundinacea</i> L.	S		0.41	0.06
<i>Subularia aquatica</i> L.	I	S	0.27	0.19
<i>Menyanthes trifoliata</i> L.	S		0.00	0.44
<i>Potamogeton obtusifolius</i> Mert. & W. D. J. Koch	E	T	0.24	0.18
<i>Carex elata</i> All.	S		0.00	0.40
<i>Bryophyta</i> sp.	B		0.37	0.00
<i>Fontinalis antipyretica</i> Hedw.	B		0.22	0.15
<i>Fontinalis hypnoides</i> Hartm.	B		0.00	0.34
<i>Drepanocladus longifolius</i> (Mitt.) Broth. ex Paris	B		0.06	0.26
<i>Potamogeton alpinus</i> Balb.	E	I	0.00	0.30
<i>Myriophyllum verticillatum</i> L.	E	T	0.30	0.00
<i>Utricularia intermedia</i> Hayne	Ce	I	0.12	0.17
<i>Lemna minor</i> L.	L	T	0.24	0.05
<i>Chara</i> sp.	Ch		0.00	0.28
<i>Myriophyllum sibiricum</i> Kom.	E	S	0.27	0.00
<i>Typha angustifolia</i> L.	H		0.00	0.27
<i>Carex vesicaria</i> L.	S		0.14	0.12
<i>Carex aquatilis</i> Wahlenb.	S		0.25	0.00
<i>Hippuris vulgaris</i> L.	H		0.06	0.17
<i>Typha latifolia</i> L.	H		0.22	0.00
<i>Nymphaea alba</i> ssp. <i>candida</i> x <i>tetragona</i>	F		0.20	0.00
<i>Scorpidium scorpioides</i> (Hedw.) Limpr.	B		0.06	0.14
<i>Lythrum salicaria</i> L.	S		0.10	0.06
<i>Wamstorfia procera</i> (Renaud & Arnell) Tuom.	B		0.11	0.03
<i>Calliergon megalophyllum</i> Mikut.	B		0.06	0.07
<i>Potamogeton berchtoldii</i> Fieber	E	S	0.00	0.11
<i>Scirpus sylvaticus</i> L.	S		0.10	0.00
<i>Wamstorfia exannulata</i> (W. Gümbel) Loeske	B		0.07	0.00
<i>Elatine</i> sp.	I		0.00	0.06
<i>Utricularia minor</i> L.	Ce	I	0.06	0.00
<i>Bidens cernua</i> L.	S		0.06	0.00
<i>Bidens radiata</i> Thuill.	S		0.06	0.00
<i>Potamogeton praelongus</i> Wulfen	E	S	0.00	0.05
<i>Nuphar pumila</i> (Timm) DC.	F	I	0.00	0.04
<i>Scolochloa festucacea</i> (Willd.) Link.	H		0.00	0.04
<i>Callitriche palustris</i> L.	E		0.00	0.03
<i>Sagittaria</i> sp.	F/H		0.00	0.03
<i>Callitriche cophocarpa</i> Sendtn.	E	T	0.00	0.01
<i>Drepanocladus sordidus</i> (Müll. Hall.) Hedenäs	B		0.00	0.01
<i>Chara globularis</i> Thuillier	Ch	S	0.00	0.01
<i>Nitella opaca/flexilis</i>	Ch	S	0.00	0.01
<i>Najas tenuissima</i> (A. Braun) Magnus	E		0.00	0.01

Appendix B.

Response of the ecological quality metrics (TT50, PMA, TI) to increased survey effort based on resampling of the (B.1) TS and (B.2) PI data for all lakes. Solid line represents the median and dashed lines the 5th and 95th percentiles of the resamples. In (B.1) the vertical dashed line represents the number of transects where 90% of the resampled EQRs fall within one ecological class.



Appendix B (Continued)



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II

ASSESSMENT OF REMOTE SENSING AS A METHOD FOR MONITORING AQUATIC VEGETATION IN LAKES OF VARYING TROPHIC STATUS

by

Kirsi Valta-Hulkkonen, Antti Kanninen, Riitta Ilvonen & Jarkko Leka 2005

[Boreal Environment Research 10: 57-66.](#)

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III

REMOTE SENSING AND GIS FOR DETECTING CHANGES IN THE AQUATIC VEGETATION OF A REHABILITATED LAKE

by

Kirsi Valta-Hulkkonen, Antti Kanninen & Petri Pellikka 2004

International Journal of Remote Sensing 25: 5745–5758.

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IV

**COMPARING MEASURES OF TAXONOMIC COMPOSITION
AND STRESS-SPECIFIC INDICES IN STATUS ASSESSMENT
OF LACUSTRINE MACROPHYTES**

by

Antti Kanninen, Seppo Hellsten & Heikki Hämäläinen 2012

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Comparing stressor-specific indices and general measures of taxonomic composition for assessing the status of boreal lacustrine macrophyte communities

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Abstract

Biotic communities are increasingly used to assess and monitor aquatic ecosystems with two fundamentally contrasting approaches: i) responses sensitive to, and indicative of specific stressors; and ii) general measures of community change. For assessment of lacustrine macrophyte communities, we compared three trophy-related and one water level fluctuation-related stressor-specific indices (SSIs) with three general measures of taxonomic composition (MTC), using data from 48 reference, 33 eutrophicated and 24 water level regulated boreal lakes. Our hypothesis was that MTCs would yield robust ecological quality estimates across these differing stress-gradients, while the SSIs would only react to the specific pressures they are calibrated against. Judged by the criteria of accuracy, precision and sensitivity, most trophy-specific SSIs performed well with respect to eutrophication and even showed some sensitivity to regulation, whereas the water-level fluctuation index reacted only to heavy regulation amplitude. Metric performance was not significantly affected by inclusion or exclusion of emergent taxa or by the use of abundance instead of occurrence alone. As expected, MTCs responded to both eutrophication and regulation and the best performing MTC – incorporating both taxa loss and gain – indicated impairment almost as often as any one (minimum) of the SSIs. We argue that non-stressor-specific MTCs – with the demonstrated unifying response of biotic community to different stressors – should primarily be used in the assessment of changes in the status of aquatic biota, whereas using SSIs a posteriori might help in identifying the likely causes of these changes.

Keywords: bioassessment, boreal lakes, stress-specific indices, taxonomic composition

1. Introduction

The effects of anthropogenic activities on aquatic ecosystems are increasingly evaluated by biotic responses. This trend is also reflected in legislation, such as the European Water Framework Directive (WFD; European Commission, 2000), making 'good ecological quality' the target of freshwater protection and restoration. In lacustrine ecosystems, the status of littoral macroflora is one of the 'biological quality elements' (BQE) on which assessment of ecological condition should be based. While the WFD verbally indicates (European Commission, 2000; Annex 5) which characteristics of the BQEs should be assessed - 'abundance' and 'community composition' for aquatic macrophytes - it does not specify which indices or metrics should be used (Hering et al., 2010). However, two fundamentally contrasting approaches have been applied for WFD-compliant and other comparable bioassessment systems: i) responses sensitive to and indicative of specific stressors, and ii) general measures of community change.

Evaluation of 'taxonomic completeness' or the ratio of the observed taxa (O) at a site to those expected to be present (E) in the absence of human disturbance is an example of the latter type of approach and is the core of several bioassessment systems based on the reference condition approach (RCA, Hawkins et al., 2010a). O/E –ratios were originally developed for river benthic macroinvertebrates (Wright et al., 1998; Norris and Norris, 1995). Other, less frequently used general measures of taxonomic composition, like the BC index (van Sickle, 2008) and Percent Model Affinity (PMA; Novak and Bode, 1992), have also been formulated originally for stream invertebrates. Recently Aguiar et al. (2011) have used the O/E-ratio for riverine macrophytes, whilst direct applications to lacustrine flora are rare (Keto et al., 2006; Sutela et al., 2012). Some lake macrophyte assessment systems, however, use a modified O/E-ratio or other measures of taxonomic composition, like PMA (Alahuhta et al., 2009; Kanninen et al., 2009). Yet, the

majority of macrophyte-based assessment schemes rely on multimetric indices (MMIs; e.g. Beck et al., 2010; Willby et al., 2009) and different stressor-specific indices, which are created by deriving indicator values for taxa along stressor gradients (Hering et al., 2006), especially for river flora (Dodkins et al., 2005; Haury et al., 2006; Holmes, 2009; Schneider and Melzer, 2003). Similar indices for lacustrine macrophytes are less common (see review by Poikane et al. 2011), but include the phosphorus-related Trophic Index (TI; Penning et al., 2008a), the nitrogen-related Ellenberg Index (EI; Ellenberg et al., 1991; Kolada et al., 2011) and a water level regulation index, WI (Hellsten and Mjelde, 2009; Mjelde et al., 2012).

The conceptual thinking behind the O/E and related measures of taxonomic composition (MTC) versus stressor-specific indices (SSI) differs fundamentally. MTCs measure general taxonomic change at impacted sites relative to reference sites and are neither derived from nor calibrated against any specific stressor gradient (Hawkins, 2006), nor constrained by the data like many MMIs (Beck and Hatch, 2009; Hering et al., 2006). These features make them ecologically relevant and theoretically robust measures of anthropogenic impact on biota with the potential to unify across different types of stressors, habitats and organisms (Jyväsjärvi et al., 2011). In contrast, the SSIs are directly derived from, and tailored to indicate, stressor gradients; even though they can be argued to represent 'composition and abundance' of species assemblages (e.g. Penning et al. 2008a), as required by the WFD. A response in a SSI always requires either a loss of species or their altered relative abundances, or both, and hence should also be manifested by the MTCs. This relationship might, however, be asymmetrical: an index specific to a certain stressor does not necessarily respond to even considerable change in the taxonomic composition caused by other stressors. This is partly because species responses to different stressors do not correlate and partly because each index typically utilizes only a portion of the species pool.

Associated with the selection of metrics is the selection of taxa to be included in the ecological quality assessment. Several macrophyte assessment systems utilize submerged flora only and omit emergent plants, confined to

the land-water interface (Poikane et al., 2011). In some assessment schemes, however, emergent species are included and recent evidence suggests that their inclusion may enhance trophic index performance (Dudley et al., 2011). Emergent macrophytes may also - more readily than submerged species - react to changes in anthropogenic land-use in the adjacent catchment area of lakes (Alahuhta et al., 2012) and they have been shown to be sensitive to water-level regulation (Hellsten, 2001). The inclusion of emergent plants in the assessments could also be justified by their functional importance in lacustrine ecosystems, for example in providing shelter and spawning-sites for fish, protecting shores from erosion and providing habitat for other macrophyte species and epiphytic algae (Gasith and Hoyer, 1998; Lacoul and Freedman, 2006, Nöges et al., 2010).

In this study we compared an array of general measures of taxonomic composition (O/E, BC, PMA) with a range of commonly used stressor-specific indices in assessing the status of boreal lacustrine macrophyte communities. We used macrophyte data from reference sites and lakes impacted by the regionally two most important anthropogenic disturbances: eutrophication and water level regulation. Our hypothesis was that MTCs will yield robust status assessments of macrophyte vegetation across these differing stress-gradients, while the SSIs will only react to the specific pressures they are calibrated against. We expected the MTCs to indicate impairment in most cases when any one of the SSIs does, but also in cases where changes are caused by another stressor, or by their joint effect, and in some instances when none of the stressor-specific indices responds. In addition, we explored whether using hydrophytes alone results in more consistent and reliable status assessments than using both hydrophyte and emergent taxa (helophytes and shore plants).

2. Materials and methods

2.1. Data

Our data set consisted of 105 lakes, which had been surveyed for macrophytes once during 2001-2010. The macrophyte data were compiled

from monitoring databases and previous studies (Kanninen et al., 2009; Sutela et al., 2012). The study lakes are located in central-eastern and northern Finland (Fig. 1) and represent a wide range of morphometric and water quality variation (Table 1). Altogether 48 lakes (REF) represent minimally- or least-disturbed reference conditions (sensu Stoddard et al., 2006), whereas 33 lakes (IMP_{Eut}) are eutrophicated to a varying degree by nutrient loading originating mainly from diffuse sources. The remaining 24 lakes (IMP_{Reg}) are regulated, with varying degree of water level regulation amplitude and including the most heavily regulated lake in Finland.

The macrophyte vegetation was mainly surveyed by a 'main belt transect method' (see Kanninen et al., 2012 for details). However, in 23 lakes the survey method differed slightly as the coverage estimation was at 1-7 Norrlin scale instead of the usual percentage scale. A common abundance value was obtained by calculating for each taxon a vegetation index (Ilmavirta and Toivonen, 1986), which combines frequency (the percentage of transects with the taxon present) and abundance (mean coverage of stands). Prior to calculation, the frequency and coverage values were transformed to a 1-7 Norrlin scale, when applicable (Ilmavirta and Toivonen, 1986).

As survey effort (7 to 43 transects per lake in our initial data set) may significantly affect the MTCs (Kanninen et al., 2012), the data were balanced prior to metric calculation by randomly selecting 10 transects per lake when possible. For the remaining 13 lakes all the 7-9 transects initially sampled, were used in the analysis.

Altogether 123 species (45 emergent and 78 hydrophyte) were present in the data set used (Appendix A). The hydrophyte taxa included 2 lemniids, 7 ceratophyllids, 20 elodeids, 10 isoetids, 4 charophytes, 22 bryophytes and 13 floating-leaved species; the emergents included 19 helophyte and 26 shore species (Appendix A).

The environmental variables (Table 1) were derived from national databases of the Finnish Environment Institute (SYKE): water quality parameters (Hertta-

database) were median values from 0-2 m depth during the growing seasons of 2000-2007; lake morphometry parameters were extracted from the national lake registry; percentage cover of arable land in the catchment was extracted from the Watershed Simulation and Forecasting System (WSFS). Winter drawdown was calculated as the average difference between highest water level in October-December and lowest level during the following April-May. In cases of missing hydrological measurements, winter drawdown was estimated according to Keto et al. (2008).

2.2. Lake groupings

We used the Finnish operative lake typology for grouping the sites for the establishment of reference conditions. The categorical typology of Finnish lakes is primarily based on lake area, mean depth, DOC concentration (measured as colour) and altitude (see Jyväsjärvi et al., 2011). In addition, turbidity and alkalinity are used to separate naturally nutrient rich and calcareous lakes, respectively. In the Finnish provisional macrophyte classification system, lakes are further divided to northern and southern sub-types due to decrease in species richness with latitude (Heino and Toivonen, 2008). To ensure an adequate number of reference sites per type, we combined some of the most similar types to form six groups: 1) northern, moderately humic (colour 30-90 mg Pt/L), large (area >5 km²); 2) southern, moderately humic, large; 3) moderately humic, small (<5 km²), shallow (mean depth <3 m); 4) moderately humic, small, deep (>3 m); 5) humic (>90 mg Pt/L); and 6) oligohumic (<30 mg Pt/L) lakes. We evaluated the ability of this slightly modified typology to produce a meaningful biological grouping by using non-metric multidimensional scaling (NMDS; McCune and Grace, 2002) and multi-response permutation procedure (MRPP; Biondini et al., 1988) on macrophyte species relative abundance data from the REF lakes, using Bray-Curtis distance metric and PC-Ord 4.35 software.

To explore and illustrate patterns in the taxonomic composition variation among the REF, IMP_{Eut} and IMP_{Reg} lakes, we applied NMDS and MRPP also to the whole dataset, as above.

2.3. Stress-specific indices

We calculated three widely used stress-specific indices (EI, OTS and TI) relating to nutrient enrichment and one (WI) to water level fluctuation (Table 2). Ellenberg et al. (1991) defined for vascular plants of Europe a set of indicator values, which have since been extensively used and elaborated by other authors (Hill et al., 2000). We used the indicator values updated and reported by Kolada et al. (2011) for lake macrophytes to calculate a nitrogen-related **Ellenberg index** (EI; Table 2). Kanninen et al. (2009) formulated an index based on the general trophic requirements of lacustrine macrophytes (Linkola, 1933; Leka et al., 2008); in the present study we call this index the **Oligotrophy Score** (OTS; Table 2). The indicator values (sensitive, indifferent, tolerant) of the **Trophic Index** (TI, Penning et al., 2008b) are derived from the occurrence probabilities of taxa along a phosphorus gradient. The **Water level fluctuation Index** (WI; Hellsten and Mjelde, 2009; Mjelde et al., 2012) is conceptually similar, the tolerance grouping of taxa being based on occurrence along a gradient of winter drawdown, which is considered the most biologically relevant measure of regulation amplitude and is commonly used as a regulation-related stress-variable (Aroviita and Hämäläinen, 2008; Sutela et al., 2012). The EI and OTS were calculated both using all taxa (emergents and hydrophytes) and using hydrophytes alone; whereas TI and WI are based only on hydrophyte taxa. The influence of utilizing abundance, rather than presence/absence data only, on metric performance was explored by calculating the SSIs from both types of data.

2.4. Measures of taxonomic composition

We calculated three metrics of taxonomic composition (MTCs; O/E-ratio, BC index and PMA). The **O/E –ratio** measures the taxonomical completeness of a site in terms of the number of taxa lost relative to reference conditions (Hawkins, 2006; Moss et al., 1987). E is the number of taxa in the expected

(i.e. reference) community and equals the sum of predicted observation probabilities (which are conditional to the standard observation effort) of taxa reaching a designated probability threshold (P_t); O is the number of these taxa observed in the sample. In many applications of the O/E, low probability taxa ($P_t < 0.5$) are excluded from calculations, as this often decreases noise (variation of O/E for reference sites) and increases the sensitivity of O/E for detecting non-reference conditions (Aroviita et al., 2009; Clarke and Murphy, 2006; Ostermiller and Hawkins, 2004). In other instances, low probability taxa have been included (Jyväsjarvi et al., 2011). To examine the influence of P_t in our data, we calculated the O/E with four probability thresholds: 0+ (all taxa included), 0.25, 0.5 and 0.75.

Van Sickle (2008) proposed the **BC index**, an adaptation of the Bray-Curtis distance metric, to measure the compositional dissimilarity between an observed and expected assemblage. The BC index uses information on all taxa, not just taxa reaching a certain P_t . Furthermore BC, unlike O/E, takes species-specific probabilities directly into account giving contrasting weights to probable and improbable observed taxa (with high and small - or even zero - expected probabilities, respectively). We considered the latter feature a potential advantage over O/E for assessment of macrophyte flora, which tends to gain species rather than lose with disturbance, and with eutrophication in particular (Penning et al., 2008b). The index is calculated as: $BC = \sum |O_k - P_k| / \sum |O_k + P_k|$, where O_k is either 0 or 1 for each taxon (k) absent or present respectively, and P_k is the predicted observation probability. The index ranges from 0-1; a site with a BC = 0 has exactly the species composition expected in reference conditions. For the calculation of Ecological Quality Ratio, EQR (see 2.5.), the original BC was converted to $1 - BC$ to make the index value decrease with community change. For further details of calculus and a comparison of O/E and BC, see van Sickle (2008). For both O/E and BC, the type-specific probability of observation (P_k) for each taxon was estimated as the ratio of the number of REF-sites at which the taxon was recorded to the total number of studied REF-sites within the type (Aroviita et al., 2008).

The Percent Model Affinity (PMA; Novak and Bode, 1992), in contrast to O/E and BC, which operate on presence data only, utilizes information on taxon (relative) abundance. Otherwise, PMA is conceptually similar to BC, using all taxa and differs only in the distance metric used. First, the average proportion of each taxon among the reference lakes of each type is calculated to produce the ‘model’ assemblage. For each lake to be assessed the observed relative abundance of each species is compared with the expected relative abundance in the model assemblage and the PMA index is calculated as the percent similarity between the two assemblages (Alahuhta et al., 2012).

2.5. Ecological Quality Ratios

The WFD requires bioassessment results to be presented in the form of Ecological Quality Ratios (EQRs). We followed the standard procedure (e.g., Hering et al., 2006), where EQR is defined as the relation between observed (O_{Metric}) and expected (E_{Metric}) metric values. For the SSIs as well as for BC and PMA, E_{Metric} for each lake type was derived as the average of metric values across the reference sites. The O/E –ratio is directly analogous to EQR and was therefore used as such ($\text{EQR}_{\text{O/E}}$).

2.6. Evaluation of metric performance

Relative performance of the metrics was evaluated using four criteria common in such comparisons: accuracy, precision, sensitivity and strength of association with stressor variables (Hawkins et al., 2010b). As we did not have enough data to form a fully independent validation set for the reference conditions, we used internal leave-one-out cross-validation (Aroviita et al., 2009). For each REF-site we re-estimated E using the set of reference sites (of the respective lake type) from which that particular site was excluded; the respective O-value was then re-calculated to obtain a cross-validated O/E (EQR_{CV}) for that site. These values were used to evaluate the **accuracy** (mean EQR_{CV}) and **precision** (SD EQR_{CV}) of each metric.

To assess metric **sensitivity** (percentage of impacted sites deemed as non-reference) and to obtain metric grading to the five quality classes (high, good, moderate, poor and bad) required by the WFD, we used the 25th percentile of the type-specific reference site EQR_{CV} distributions as the boundary between high and good ecological class (H/G boundary). To enable direct comparison among metrics and lake types, with varying precision, the EQR values need to be re-scaled. We used a linear rescaling (Mykrä et al., 2012), whereby each quality band corresponds to 0.2 units, H/G boundary to the value 0.8 and lower class boundaries for each metric to the values 0.6, 0.4, and 0.2. This is achieved by multiplying the original EQRs by the ratio $0.8/\text{original H/G boundary value}$. Similarly, the values higher than the H/G boundary, corresponding to high status class, are rescaled by the factor $0.2/(1-\text{H/G value})$, to equalize the band width in the 'classification scale' (from H/G to 1 to correspond to 0.2 units) and to retain the original values of 1 (Mykrä et al., 2012). In addition, the separation of status groups with different metric EQRs was tested statistically using Mann-Whitney U-test for REF and IMP_{Eut} sites and REF and IMP_{Reg} -sites, respectively.

For the evaluation of the **association with stressor variables**, metric EQRs were correlated to an anthropogenic pressure variable (% of arable land) as well as to two uncorrelated stressor variables: 1) a synthetic PCA component derived from total phosphorus, total nitrogen and chlorophyll-a as a measure of eutrophication and 2) winter drawdown as a measure of water level fluctuation. As some of the metrics were non-normally distributed and associations non-linear even after transformations, non-parametric Spearman correlation was used.

2. 7. Comparison of metric classification results

After choosing the best performing candidates in each metric category (general, eutrophication and regulation metrics) the relationships among those metrics were evaluated. To address our primary research hypothesis - that MTC will classify a test site as impaired in most cases when any of the SSI does so - we used bi-variate scatter plotting of the re-scaled EQRs and

compared the classifications by MTC and SSIs. Additionally, to address whether the MTC also react in some cases where the non-reference status cannot be attributed to any one of the specific stressors (is not detected by SSI) we compared the minimum of SSIs to the MTCs.

3. Results

3. 1. Lake groupings

An NMDS of the reference sites produced a three dimensional solution (Fig. 2A) which explained 81.8 % of the original Bray-Curtis distances among lakes. Of the typology related environmental variables, latitude, water colour, lake area and mean depth were the strongest correlates with the ordination axes, but the trophic-related variables also correlated with the axes (Fig 2A; Appendix B). The lake types formed distinctive groups, although there was significant overlap, especially among closely related types (e.g. small, moderately humic shallow and deep lakes). The MRPP analysis – a non-parametric procedure for testing the hypothesis of no difference between groups – also suggested ($A=0.255$, $p<0.001$) that the modified typology produced a biologically significant lake grouping.

For the set of all lakes, a three-dimensional NMDS solution (Fig. 2B) was optimal, and explained 84.5 % of the original distances among lakes. The three *a priori* status groups formed distinctive aggregates in the ordination space, with some degree of overlap. The strongest environmental correlates were those related to trophic status (total P, total N, chlorophyll-a, arable land), lake geographical position (N, E), winter drawdown and water colour (Fig 2B; Appendix B). The MRPP analysis supported the status grouping being relevant with regard to macrophyte community composition ($A=0.115$, $p<0.001$).

3. 2. Metric performance

3.2.1 Accuracy and precision

The accuracy was high (mean EQR_{CV} close to 1) for all SSIs, whereas MTCs were slightly biased (mean $EQR_{CV} < 1$; Table 3). Precision was relatively high for all metrics (SD of $EQR_{CV} < 0.20$; Table 3). Incorporating abundance into the SSIs had negligible effect on accuracy and precision of EQR_{CV} ; as had using hydrophyte taxa only (OTS and EI) instead of using all taxa (i.e. hydrophytes and emergents). For the MTCs, however, omitting emergent taxa slightly reduced accuracy and precision (Table 3).

3.2.2. Sensitivity

Sensitivity varied more than accuracy and precision among metrics (Table 3). With regard to eutrophication, OTS was the most sensitive among the SSIs, classifying the highest proportion of IMP_{Eut} sites as non-reference (status class 'good'; $0.6 \leq EQR < 0.8$) or clearly impaired ('moderate' or lower; $EQR < 0.6$). As expected, the water-level fluctuation index (WI) was least sensitive to eutrophication. In contrast, WI classified a higher proportion of IMP_{Reg} sites as clearly impaired than did the other SSIs, albeit generally WI did not make a statistically significant distinction between the REF and IMP_{Reg} groups, unlike most other metrics (Table 3). Somewhat surprisingly, the TI and OTS also seemed to be sensitive to water level regulation (Table 3). Incorporating species abundance into index calculation did not show any consistent influence on metric sensitivity. On the other hand, omitting emergent taxa slightly increased the sensitivity of OTS and EI.

Of the MTCs, BC was the most sensitive, resulting in the highest number of IMP_{Eut} and IMP_{Reg} sites evaluated as impaired (Table 3). PMA was also responsive to both pressure types. The sensitivity of the O/E clearly increased when rare taxa ($P_t < 0.5$) were omitted (Table 3). Omitting emergents from the MTCs increased sensitivity to eutrophication, whereas for regulated lakes sensitivity increased for O/E but not consistently for PMA and BC (Table 3).

3.2.3. Correlation with stressor variables

Correlations of SSIs with trophy-related stressor variables were negative and strong, as expected (Table 3). The WI, on the other hand was not related to

winter water level drawdown, when calculated across all lake types (Table 3). The relationship was only evident ($r = -0.634$, $p = 0.001$) among regulated lakes; in hydrologically natural lakes (REF and IMP_{Eut}) variation of the index EQR was high despite limited variability in water-level draw down (Fig. 3). WI indicated significant impairment ($EQR < 0.6$) only in IMP_{Reg} lakes with high regulation amplitude (drawdown $> 3m$). On the other hand, the OTS – a trophy-related metric - also correlated with water level drawdown. Stressor variable associations with the MTCs were not as strong as with SSIs but, importantly, all (except O/E with $P_t < 0.5$) correlated negatively with both trophy- and hydrology-related stressor variables (Table 3, Fig. 3).

3.3. Among-metric relationships

As a general rule, including abundance in the SSIs did not enhance metric performance (Table 3) and indices based on abundance had strong, positive correlation (Spearman rho range 0.72 - 1.00, $p < 0.001$; except WI, with $r = 0.484$, $p < 0.001$) with their presence/absence counterpart. Similarly, indices including emergent taxa almost invariably exhibited slightly poorer performance than did metrics based on hydrophytes alone. Therefore, for the subsequent analysis of among-metric relationships, only metrics based on **presence/absence data of hydrophytes** are presented.

The three trophy-related EQRs of SSIs (TI, OTS, EI) were strongly, positively intercorrelated (Fig. 4), as were the three MTCs (PMA, BC, O/E). On the other hand, the WI was negatively associated with the trophy-related SSIs and showed also weak negative correlation with the MTCs (Fig 4). The associations between MTCs and the trophy-related SSIs were positive, with PMA showing relatively weak correlations (with TI and OTS) or no correlation at all (with EI; Fig. 4).

A comparison of the minimum SSI against the best performing MTC, the BC index, showed a strong correlation (Spearman rho = 0.657, $p < 0.001$; Fig. 5). In terms of significant impairment ($EQR < G/M$ boundary), the different approaches agreed in 87% of the cases. The methods disagreed in 9 cases

(1 REF, 3 IMP_{Eut} and 5 IMP_{Reg} sites) when the BC showed no significant impact (EQR < 0.6) while one or more SSI did; conversely, there were five cases (3 IMP_{Eut} and 2 IMP_{Reg} sites) where BC classified the status as impaired while none of the SSIs did. In classifying sites to non-reference category (0.6 < EQR < 0.8), the methods disagreed more often, in 34% (n = 36) of the cases. Only in 4 (3 REF, 1 IMP_{Reg}) cases did BC show non-reference status when none of the SSIs did, while in 32 (21 REF, 7 IMP_{Eut} and 4 IMP_{Reg}) cases the minimum of SSIs indicated non-reference status while the BC did not (Fig. 5).

4. Discussion

4. 1. Performance of stress-specific metrics

Generally, differences in performance among the trophy-related SSIs were minor; the greatest contrasts were in sensitivity, in which the OTS outperformed the other trophy-related metrics. One explanation for this is that the indicator values of OTS, first derived by Linkola (1933), and further elaborated by Toivonen and Huttunen (1995) and Leka et al. (2008), are specifically tailored to the biogeographical and environmental conditions of our study area. For instance, in the brown-coloured lakes, which dominate both in the boreal area and our data set (Table 1), a high concentration of phosphorus bound to humic substances has been linked to the suboptimal performance of common SSIs, like the phosphorus-specific TI (Rask et al., 2010). Another explanation for the higher sensitivity of OTS might be its inclusion of a larger proportion of the regional flora compared to the other trophy indices developed elsewhere: in our data, 92 taxa (74.8% of the 123 taxa in total) had an OTS-indicator value, while 87 (70.7%) and 45 (36.6%) had indicator values for EI and TI, respectively. Furthermore, TI contains only hydrophytes; but even for these the number of indicator taxa was higher for OTS (60) than for EI (49) or TI.

The WI, specific to water level fluctuation, performed poorly overall by the criteria used, and only indicated impairment in the most heavily regulated

lakes with winter drawdown more than 3 meters. Hence, we might simply conclude that there are no significant effects on species composition until regulation amplitude exceeds 3 meters (see also Sutela et al., 2012). Based on their analysis of macrophyte species occurrence in 73 lakes from Finland, Norway and Sweden, Mjelde et al., (2012), similarly proposed a 'good-moderate' status boundary for 3.5 m winter drawdown. However, the MTCs and even some of the eutrophication-specific indices were more sensitive to regulation, suggesting that WI did not consistently detect actual community composition changes, attributable to regulation. The WI was originally calibrated with mostly northern, low-alkalinity lakes with regular ice cover (Hellsten and Mjelde, 2009). Hence, the greater latitudinal gradient of our test data compared to the original calibration data may have contributed to low WI performance, as latitude is one of the main environmental determinants of macrophyte species distribution in the boreal region (Heino and Toivonen, 2008). In addition, our measure of water level fluctuation, winter drawdown, was modelled in most of the non-regulated lakes, which may have resulted in imprecision of the observed stress-relationships. Moreover, changes in spring flood and especially in average water level can be more significant drivers of change in mildly regulated lakes (Partanen and Hellsten, 2005). Furthermore, several of our regulated lakes were also eutrophicated, so our IMP_{Reg} lake population cannot strictly be used to draw inferences about the sole effects of regulation. In fact, the flora of some of the IMP_{Reg} lakes was similar to that of IMPEut –lakes, as indicated by the NMDS-analysis. This may explain why the trophy-specific indices, OTS in particular, also seemed to respond to regulation. In addition, several of the species unquestionably preferring oligotrophic conditions (e.g. large isoetids: *Isoetes lacustris*, *I. echinospora*, *Lobelia dortmanna*) are also well known to be sensitive to regulation, and to winter drawdown in particular (Rørslett and Brettum, 1989; Hellsten, 2002).

4.2. Performance of measures of taxonomic composition

The most commonly applied MTC, the O/E–ratio of taxa, displayed a similar pattern in relation to variable taxon observation probability as often reported for benthic fauna (Aroviita et al., 2009; Clarke and Murphy, 2006; Ostermiller

and Hawkins, 2004); omitting rare taxa increased metric performance. For our data, the O/E metric performed best with the highest probability threshold tested ($P_t = 0.75$) and inclusion of low probability taxa yielded no sensitivity. Low probability taxa ($P_t = 0+$ or $P_t = 0.25$) have commonly been included in assessments of species-poor communities like riverine fish (Joy and Death, 2002) and profundal invertebrates (Jyväsjarvi et al., 2011), while a higher P_t performs better in more diverse communities like lotic macroinvertebrate. Apparently, the macrophyte species pool of our study lakes was diverse enough for the O/E –type index to perform reasonably well with even higher P_t values. In practice, however, the number of species present in reference conditions with high probability ($P_k \geq 0.75$) was low (5-7 emergent and 6-8 hydrophyte taxa per lake type), and the ‘type-specific’ taxa often included isoetids (*Isoetes echinospora*, *I. lacustris*, *Lobelia dortmanna* and *Ranunculus reptans*), as well as helophytes (*Phragmites australis*, *Equisetum fluviatile*) and shore species (*Carex lasiocarpa*, *C. rostrata*, *Lysimachia thyrsiflora*, *Comarum palustre*), in addition to the floating-leaved *Nuphar lutea*, the most common macrophyte species in the region. It is evident that the best performing O/E metric ($P_t = 0.75$ of hydrophyte taxa) was heavily dependent on the occurrence of isoetid species, which are sensitive to both the disturbances represented in the present study. Hence, it is questionable how far this result of high P_t being optimal for lake macrophytes can be generalized. Moreover, the O/E metric of taxonomic completeness with a high P_t , while performing well by numerical criteria, provides limited information regarding overall changes in macrophyte taxonomic composition even when considering the presence and absence of the componential species.

The BC index, a dissimilarity metric which compares the predicted observation probabilities to actual observations for all taxa, was the best performing MTC for our data. The index was originally (Van Sickle, 2008) demonstrated to be more sensitive than O/E in assessment of lotic invertebrates, but has since received relatively little attention (but see Jyväsjarvi et al. 2011; Hallstan et al., 2012). The fact that the BC index incorporates not only taxa loss, but also taxa gained under impairment, makes it especially suitable for assessment of lacustrine macrophytes, as the flora of

naturally oligotrophic or dystrophic boreal lakes tends (at least initially) to diversify with increased disturbance, especially with nutrient enrichment (Penning et al., 2008b; Rørslett, 1991). The eutrophication-induced shift in species composition is due to sensitive species, like isoetids, decreasing and/or disappearing and several more demanding species, e.g. lemnids, appearing, which is reflected by the BC index. Importantly, in the case of water-level regulation the BC index, like other MTCs, was also capable of demonstrating community change, in unifying terms comparable with degree of change caused by eutrophication or in principle, any other type of disturbance.

PMA has been successfully used in status assessment studies of benthic fauna (Novak and Bode 1992; Tolonen and Hämäläinen 2010) and it has been incorporated in the provisional ecological classification system in Finland (Rask et al., 2010; Vuori et al., 2009). For our data PMA, the only MTC incorporating taxon abundance, performed reasonably, although not as well as its presence-absence counterparts. An obvious conceptual benefit of the PMA, in addition to those of BC, is that it can be interpreted to fulfill the requirement of 'abundance' that needs to be measured according to the normative definitions of the WFD (European Commission, 2000, Annex 5). Most current European macrophyte assessment systems (see review by Poikane et al., 2011), however, utilize quantitative measures of abundance (e.g. maximum colonization depth as a proxy for submerged vegetation abundance) instead of relative taxon abundance. Quantitative measures of vegetation abundance could, more effectively than field-based species surveys, be provided by alternative monitoring techniques, like remote sensing (Valta-Hulkkonen et al., 2005).

4.3. Role of emergent taxa and abundance

For our data, the inclusion of emergent taxa did not significantly affect the performance of the metrics, according to the numerical criteria used. Helophytes inhabiting the lake eu- and geolittoral are most often omitted from assessment systems (Poikane et al., 2011), mainly due to the presumed high

variation in taxa composition caused by soil characteristics and shore morphometry (Penning et al., 2008a). However, the effects of omission or inclusion of emergent taxa have previously only rarely been specifically tested and with variable outcomes. Recently, Kolada et al. (2011) reported improved performance of the EI with a large European-scale dataset when helophytes were included. On the other hand, with a smaller dataset helophytes provided little additional information (Dudley et al., 2011). Alahuhta et al. (2012) suggested, although with circumstantial evidence only, that assessment metrics including helophytes yield stronger responses to anthropogenic land-use pressure than a metric (TI) omitting them. Moreover, the vertical extension of helophyte stands (not evaluated in the current study) has been previously shown to be one of the most relevant responses to water level fluctuation (Partanen and Hellsten, 2005; Partanen et al., 2006). Nevertheless, helophytes are often considered important components of boreal ecosystems, providing feeding and spawning habitat and shelter to aquatic life, especially in humic lakes where submerged vegetation is sparse (Gasith and Hoyer, 1998; Lacoul and Freedman, 2006). Hence, their inclusion in bioassessment of ecosystem condition might be justified for this reason alone, notwithstanding the drawback that inclusion of helophytes in macrophyte monitoring programs comes at the expense of more time needed for conducting the survey and, possibly, of decreased precision (Kanninen et al., 2012).

Using taxon abundance to calculate weighted SSI values did not improve metric performance with our data. This seems to be the case for most indicator-based systems, at least when sufficiently diverse communities (not dominated by few taxa) are considered (Diekmann, 2003). On the other hand, compositional metrics incorporating abundance have been reported to perform better than presence/absence metrics in riverine macrophyte status assessment (Aguiar et al., 2011) and several studies have pointed to abundance being more sensitive to hydrological change than species occurrence (Nilsson and Keddy 1988; Coops and van der Velde, 1996; Hellsten, 2001). Our results, however, do not support the use of metrics incorporating abundance, which (including PMA) performed generally slightly

worse than presence-based metrics. One possible reason is the confounding effect of interannual variation, which may be greater for abundance than taxon presence (Diekmann, 2003). Also methodological noise, such as among-surveyor variation, may be higher in metrics considering abundance (Kanninen et al., 2012).

4.4. Metric relationships

The trophy-related SSIs were strongly correlated with each other for impacted lakes, but in reference conditions there was high variation and less evident correlation, which resulted in the minimum SSI being more probable ($p = 0.65$ for a REF-site EQR < 0.8) than the single best performing MTC – the BC index ($p = 0.27$) – in classifying sites as non-reference. In clearly impacted lakes, however, the minimum SSI vs. BC agreed better, as there was less variation among SSIs. The analysis gave support to our initial hypothesis of MTCs being able to identify impaired macrophyte status in most cases when any of the stress-specific metrics does. Importantly, MTCs measuring general biotic degradation responded consistently, to both nutrient enrichment and water level regulation.

Some multimetric assessment systems or indices (MMIs) – based on the indices of biological integrity (IBI; Karr, 1991; Beck and Hatch, 2009) – use stress-specific indices (e.g. Willby et al., 2009) or the presence of indicator species (Beck et al., 2010) as component metrics. While the rationale of using IBIs is intuitively appealing – the multiple, preferably non-correlated metrics measure different aspects of the structure of the biological community at hand (Karr, 1991) – combining metrics measuring dissimilar things on different scales is problematic. Similarly, the degree of change in, and status classifications by different SSIs is difficult to compare judiciously. For instance, we cannot compare EQR values (no matter whether rescaled - as here - or not) between WI and OTS in any ecologically relevant terms; the associated degree of compositional change per index unit can be quite variable, both within and among indices. Dodkins et al. (2005) attempted to overcome this problem by unifying the degree of community overturn by

standard deviation units in the CCA-ordination space constrained by differing stress-gradients. For the MMIs, elaborate metric combination rules have been developed (e.g. Willby et al., 2009) or the range of values of individual metrics is calibrated and rescaled to cover a certain pressure gradient (Minns et al., 1994; Beck and Hatch, 2009). An essential, both pragmatic and conceptual, advantage of MTCs is the lack of need for such elaborate and inevitably arbitrary rules for calibration across different types of disturbances or their combination.

The performance of SSIs, MMIs and MTCs has rarely been contrasted, as the comparison is usually hampered by differing methods in predicting biota under reference conditions (Hawkins et al., 2010b). Kilgour et al. (2004) documented that general descriptors of community change (PMA and ordination axis scores) revealed altered composition of benthic invertebrate assemblages among replicated cases with variable disturbances more consistently than any of the SSIs compared. A recent comparison (Hawkins et al., 2010b), showed that taxonomic completeness (OE index) performed better than MMIs, at reflecting known diatom community alteration. In one of the few, if not the only, macrophyte studies to compare conceptually differing assessment approaches, Aguiar et al. (2011) found a better performance of an MTC-type metric ('MAC') compared to an SSI (MTR; Holmes, 2009), although an MMI ('RVI') was considered the best performing assessment tool, on the basis of correlations with the observed disturbance.

Generally, comparing metric performance with the statistical criteria of accuracy, precision, sensitivity and correlation to stressor variables, while commonly applied (Hawkins et al., 2010b), is insufficient (e.g. Cao and Epifanio, 2010; Jyväsjärvi et al., 2010) and should be regarded cautiously and as secondary to the biological relevance of the metric (Aroviita et al., 2010). For instance, insofar as the aim is to detect changes in biota, rather than indicate a stressor using biota as a proxy, the shape and strength of association between the metric and a specific stressor variable is irrelevant. In the present study, the weaker correlation of MTCs with the trophic gradient (PCA 1) as compared to trophy-specific SSIs is what could be expected, as

the latter are tailored to indicate the gradient in question. MTCs simply measure the change in community composition, which can be caused by many, possibly uncorrelated disturbances (regulation and eutrophication in our case) and hence does not necessarily closely correlate with any single stressor. Therefore, ultimately, metrics should be selected on the basis of ecological criteria (Cao and Epifanio, 2010) and metrics used to evaluate ecosystem status should ideally be related to ecosystem function or other values (Aroviita et al., 2010; Cao and Epifanio, 2010; Smyth et al., 2007). Whether the MTCs treated here and others meet this criterion is arguable; yet it is reasonable to assume that an ecosystem harbouring flora and fauna natural in composition also functions naturally (Stoddard et al., 2006). Consistency with the relevant legislation is another important criterion in applied status assessment that is well met by the MTCs. A recent tendency in European water policy, however, is to establish SSIs and use them to conduct intercalibration of assessment systems among countries (Poikane et al., 2011). This may lead to the status of naturally nutrient rich and floristically diverse lakes being classified incorrectly, and even to discrepancies between different legislations aimed at conserving diversity and ecological status (Ecke et al., 2010).

4.5. Conclusions

Our results, together with some previous studies, demonstrate that stressor-specific community indicators are largely redundant with general community composition metrics in detecting biotic impairment. In addition to this 'technical' argument, there are more conceptual grounds, as discussed above, underpinning the use of the latter in ecological assessment. Generally, in our view, a clearer distinction should be made between metrics of system state and indicators of stressors or human pressures (see also Daan, 2005; Hatton-Ellis, 2008). Hence, non-stressor-specific measures of taxonomic composition should primarily be used in the assessment and classification of the changes in status of aquatic biota, and stress-specific indicators only *a posteriori*, as illustrated by Meador et al. (2008), to help identifying the likely causes of these changes.

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Figure captions

Fig. 1. Location of the study lakes in Finland. Symbols: white=REF, light grey=IMP_{Eut} and dark grey=IMP_{Reg} lakes.

Fig. 2. NMDS ordination bi-plot of a) reference lakes grouped by lake type and b) all lakes grouped by *a priori* status. For lake types in a), see text (sec. 2.2).

Fig 3. Responses of the metric EQRs of the stress-specific oligotrophy score (OTS; panels a and b) and water level fluctuation index (WI; c, d) and the general BC index (e, f) to the eutrophication gradient (PCA1) and winter water-level drawdown in different status groups of lakes. Linear fit lines shown. Correlation coefficients with significances are presented in Table 3.

Fig 4. Scatterplot matrix of macrophyte status metric EQRs. All metrics are based on count data of hydrophyte taxa. (O/E: $P_t=0.75$). Spearman's rho correlation coefficients are shown and significance flagged with asterisks: $*=p<0.05$, $**=p<0.01$ and $***=p<0.001$.

Fig 5. The minimum of stress-specific metrics ($EQR_{MIN SSI}$) plotted against the best performing measure of taxonomic composition, the BC index (EQR_{BC}). The class boundaries of High/Good (dashed line) and Good/Moderate (solid line) are indicated.

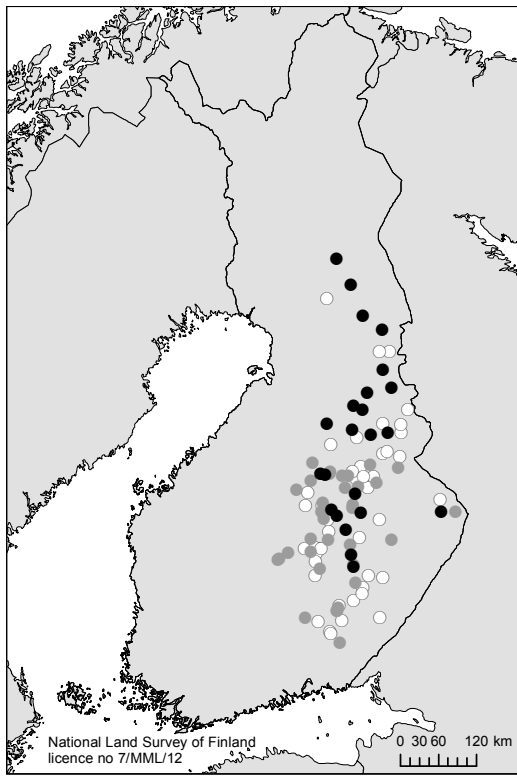


Fig 1.

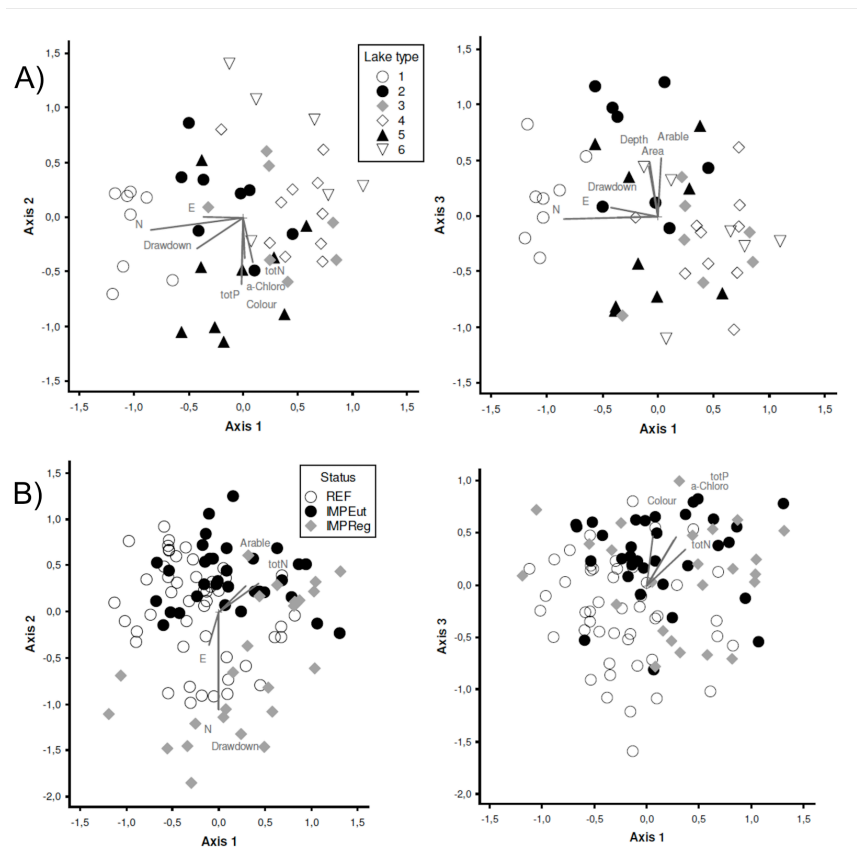


Fig. 2.

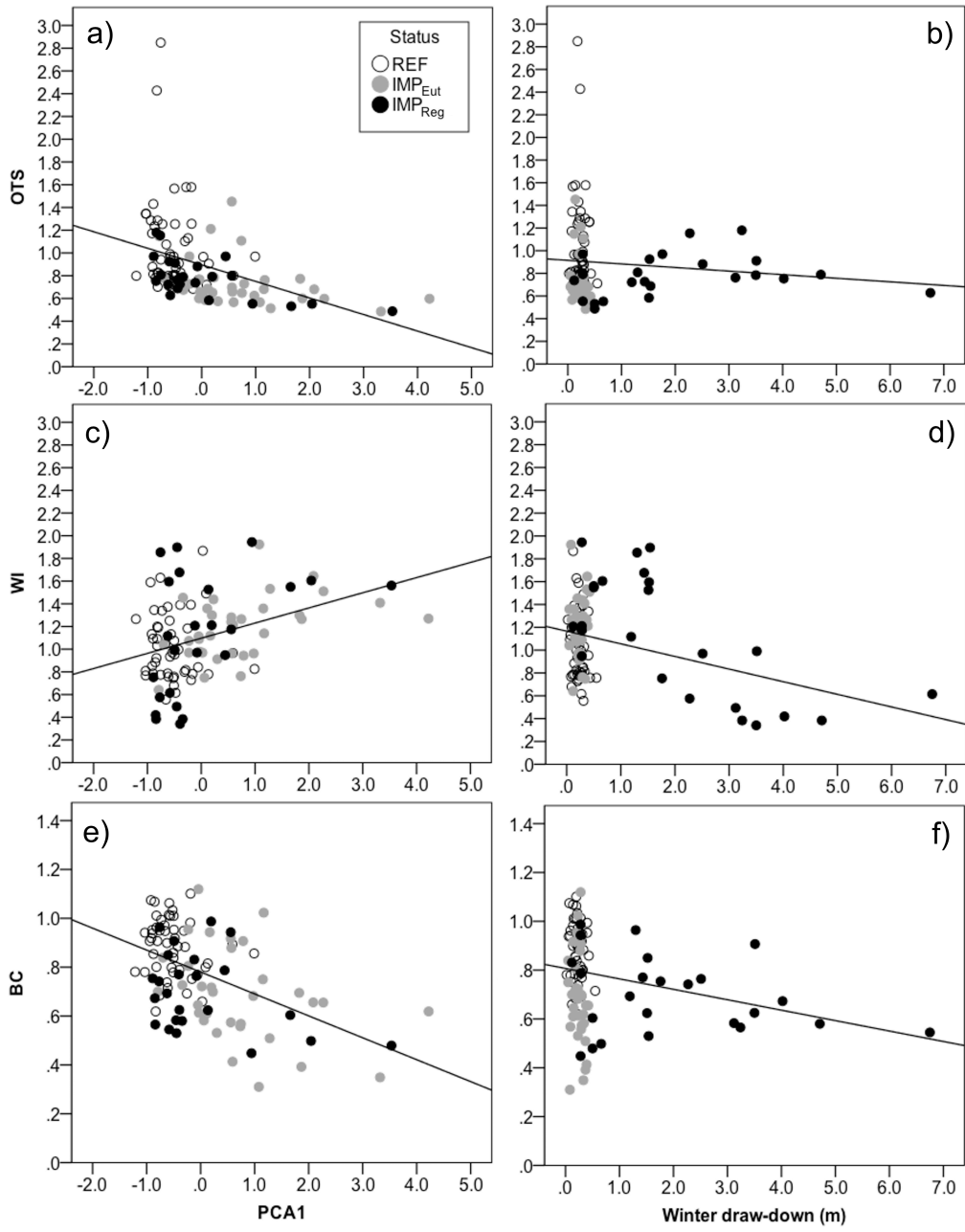


Fig 3.

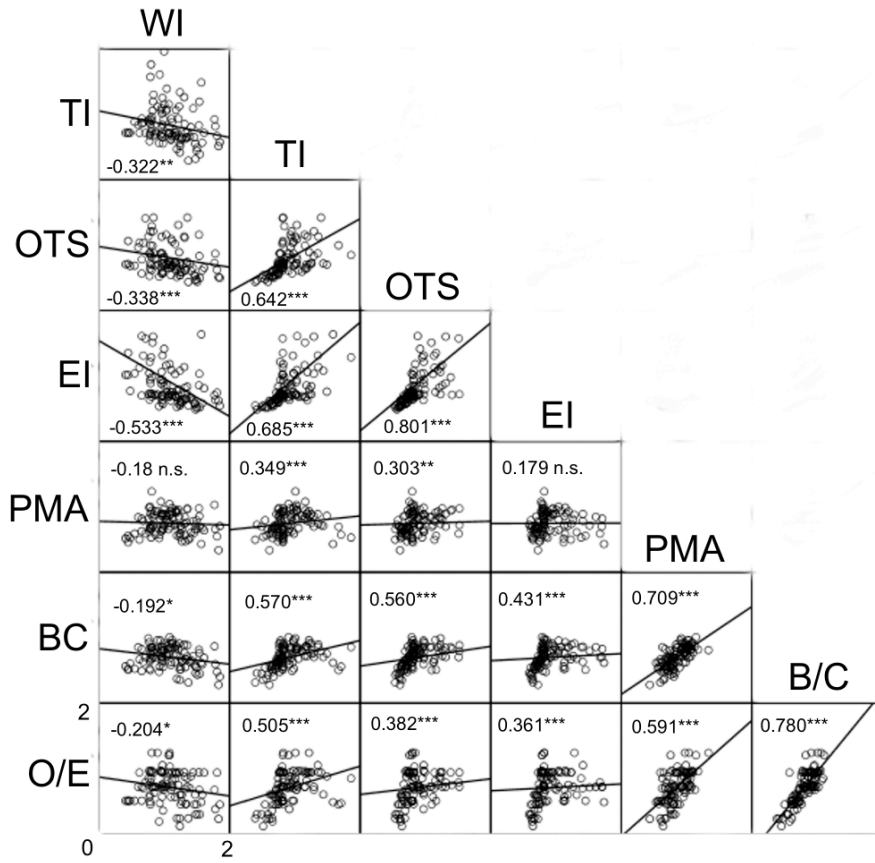


Fig. 4.

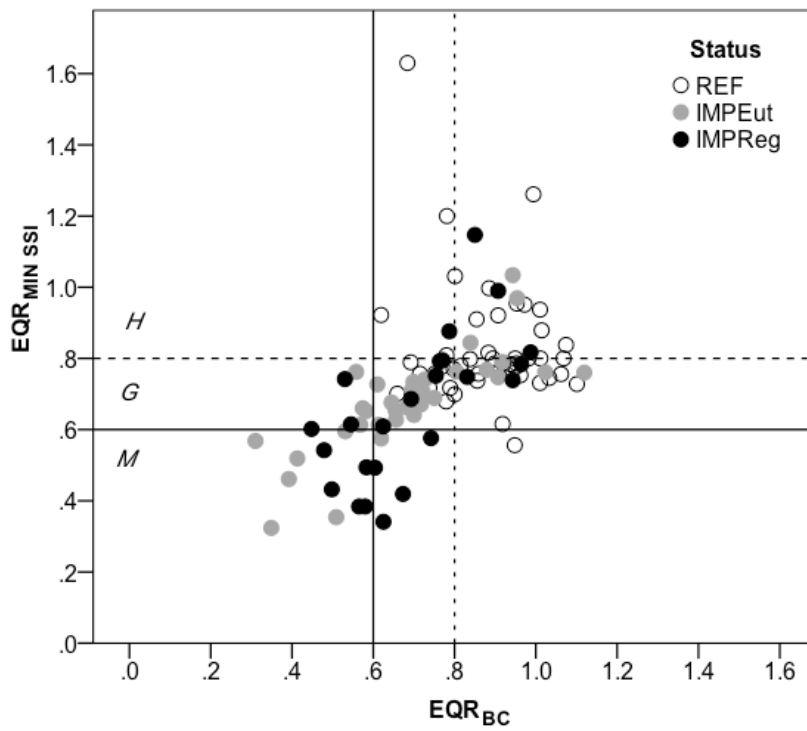


Fig. 5.

TABLES.

Table 1. Summary of some of the main hydro-morphological and water quality properties of lakes in different *a priori* status groups.

	<u>REF (n=48)</u>			<u>IMP_{Eut} (n=33)</u>			<u>IMP_{Reg} (n=24)</u>		
	<u>Mean</u>	<u>Min</u>	<u>Max</u>	<u>Mean</u>	<u>Min</u>	<u>Max</u>	<u>Mean</u>	<u>Min</u>	<u>Max</u>
Area (km ²)	25.9	0.4	393.5	5.0	0.3	75.8	108.3	9.1	878.1
Depth (m)	4.6	0.5	10.4	3.2	1.0	8.0	6.1	3.0	9.5
Colour (mg Pt/l)	67	5	180	94	15	300	76	40	160
Arable land (%)	2.1	0	10.7	9.2	0.2	30.8	3.9	0.2	12.5
Chl-a (µg/l)	6.3	1.5	32.0	19.0	2.4	52.0	11.8	3.0	41.0
Tot. P (µg/l)	12.1	4.0	28.0	28.7	8.0	99.0	21.6	8.0	80.0
Tot. N (µg/l)	394	220	635	623	300	1300	470	240	1100
Winter water level drawdown (m)	0.23	0.02	0.55	0.24	0.05	0.44	1.95	0.12	6.75

Table 2. Summary of the stress-specific indices with equations.

Index	Stressor	Indicator species	Equation
Ellenberg index (EI)	Nitrogen	Scores 1-9: From 1= indicator of extremely infertile to 9= extremely nutrient rich sites	$El_C = (\sum N_i * E_i) / \sum N_i$ $El_A = (\sum A_i * E_i) / \sum A_i$, where N_i is the number of taxa, A_i is the relative abundance and E_i is the numeric Ellenberg score of the i:th indicator group (1-9). El_C and El_A is the Ellenberg index with count and abundance data, respectively
Oligotrophy score (OTS)	General eutrophy	Scores 1-5: 1=eutrophy, 2=meso-eutrophy, 3=mesotrophy, 4=oligo-mesotrophy, 5=oligotrophy	$OTS_C = (\sum N_i * S_i) / \sum N_i$ $OTS_A = (\sum A_i * S_i) / \sum A_i$ where N_i is the number of taxa, A_i is the relative abundance and S_i is the numeric trophic score of the i:th indicator group (1-5) OTS_C and OTS_A is the Oligotrophy score with count and abundance data, respectively
Trophic Index (TI)	Phosphorus	Tolerant, Sensitive, Indifferent	$TI_C = ((N_S - N_T) * 100) / N_{S+T+I}$ $TI_A = ((\sum A_S - \sum A_T) * 100) / \sum A_{S+T+I}$ where N_S and A_S , N_T and A_T and N_I and A_I are the number and abundance of sensitive taxa, tolerant and indifferent taxa, respectively. TI_C and TI_A is the Trophic Index with count and abundance data, respectively
Water level Index (WI)	Water level fluctuation	Tolerant, Sensitive, Indifferent	$WI_C = ((N_S - N_T) * 100) / N_{S+T+I}$ $WI_A = ((\sum A_S - \sum A_T) * 100) / \sum A_{S+T+I}$ Abbreviations as in TI, except WI_C and WI_A is the Water level fluctuation Index with count and abundance data, respectively

Table 3. Summary of macrophyte status metric performance. The criteria used to assess metric performance were accuracy (mean of EQR_{CV}), precision (SD of EQR_{CV}), sensitivity (proportion of IMP-sites deemed as non-reference, i.e. $EQR < 0.8$, and/or impaired, i.e. $EQR < 0.6$) and correlation (Spearman's rho) to stressor variables (significance flagged with asterisks: *= $p < 0.05$ and **= $p < 0.01$). Columns P_{Eut} and P_{Reg} are the significances of the Mann-Whitney U-tests of difference between reference and IMP_{Eut} and IMP_{Reg} sites, respectively.

Metric	REF		IMP _{Eut}				IMP _{Reg}				Correlation to stressors		
	Mean EQR_{CV}	SD EQR_{CV}	% of EQR			P_{Eut}	% of EQR			P_{Reg}	% Arable land	PCA1	Winter drawdown
			≥ 0.8	0.6 – 0.799	< 0.6		≥ 0.8	0.6 – 0.799	< 0.6				
WI _C	1.004	0.187	91	9	0	0.009	67	8	25	0.481	0.429**	0.350**	-0.073
WI _A	1.004	0.176	79	21	0	0.247	63	13	25	0.599	0.365**	0.365**	-0.136
TI _C	1.001	0.079	30	52	18	<0.001	33	54	13	<0.001	-0.425**	-0.542**	-0.182
TI _A	1.003	0.143	30	61	9	<0.001	42	54	4	0.001	-0.396**	-0.560**	-0.188
OTS _{C-All}	1.002	0.105	18	64	18	<0.001	21	63	17	<0.001	-0.694**	-0.602**	-0.291**
OTS _{A-All}	1.002	0.116	21	48	30	<0.001	38	46	17	<0.001	-0.681**	-0.633**	-0.281**
OTS _{C-Hyd}	1.001	0.090	15	58	27	<0.001	33	46	21	<0.001	-0.686**	-0.587**	-0.175
OTS _{A-Hyd}	1.001	0.094	18	61	21	<0.001	13	67	21	<0.001	-0.619**	-0.559**	-0.272**
EI _{C-All}	1.001	0.107	30	70	0	<0.001	58	42	0	0.203	-0.625**	-0.603**	-0.016
EI _{A-All}	1.001	0.107	30	70	0	<0.001	58	42	0	0.203	-0.625**	-0.603**	-0.016
EI _{C-Hyd}	1.001	0.064	15	76	9	<0.001	58	29	13	0.049	-0.735**	-0.619**	0.022
EI _{A-Hyd}	1.002	0.132	15	73	12	<0.001	42	50	8	0.012	-0.558**	-0.626**	-0.086
PMA _{All}	0.915	0.105	52	45	3	0.015	29	54	17	<0.001	-0.180	-0.234*	-0.329**
PMA _{Hyd}	0.909	0.142	33	48	18	0.001	38	46	17	0.004	-0.195*	-0.250*	-0.238*
BC _{All}	0.933	0.063	18	67	15	<0.001	21	63	17	<0.001	-0.369**	-0.362**	-0.352**
BC _{Hyd}	0.925	0.089	27	42	30	<0.001	25	42	33	<0.001	-0.330**	-0.438**	-0.346**
O/E($P_t=0$) _{All}	0.909	0.201	88	12	0	0.021	88	8	4	0.010	0.415**	.336**	0.098
O/E($P_t=0$) _{Hyd}	0.892	0.242	70	15	15	0.954	79	17	4	0.126	0.096	0.001	0.006
O/E($P_t=0.25$) _{All}	0.929	0.181	70	21	9	0.672	75	25	0	0.595	0.181	0.064	-0.107

O/E($P_t=0.25$) _{Hyd}	0.929	0.224	45	27	27	0.022	42	42	17	0.038	-0.112	-0.219*	-0.205*
O/E($P_t=0.5$) _{All}	0.929	0.134	45	30	24	0.018	42	46	13	0.012	-0.072	-0.167	-0.235*
O/E($P_t=0.5$) _{Hyd}	0.917	0.166	42	21	36	0.003	33	33	33	0.007	-0.097	-0.238*	-0.255**
O/E($P_t=0.75$) _{All}	0.933	0.121	36	42	21	<0.001	42	50	8	0.003	-0.220*	-0.311**	-0.222*
O/E($P_t=0.75$) _{Hyd}	0.912	0.166	30	30	39	<0.001	29	29	42	<0.001	-0.226*	-0.420**	-0.287**

Appendix A. The macrophyte species of the study lakes. Taxon life-forms (B=Bryophytes, Ce=Ceratophyllids, Ch=Charophytes, F=Floating-leaved, E=Elodeids, I=Isoetids, H= Helophytes, L=Lemnids, S=Shore species) and division to emergents (Eme) and hydrophytes (Hyd) is indicated. Occurrence probabilities of taxa (P_k) in different status groups are calculated across all lakes (without division into lake types). The taxa list is organized according to the value O_k-P_k (calculated as average across lake types); which indicates the contribution of each taxon to the value of the BC index (van Sickle, 2008).

Taxon	Life-form	Hyd/Eme	EI	OTS	TI	WI	P_{REF}	P_{IMPEut}	P_{IMPReg}	Mean O_k-P_k
<i>Alisma plantago-aquatica</i> L.	H	Eme	8	2			0.396	0.629	0.818	0.500
<i>Potamogeton perfoliatus</i> L.	E	Hyd	8	1			0.396	0.800	0.864	0.485
<i>Eleocharis palustris</i> (L.) Roem. et Schult.	H	Eme	4				0.521	0.486	0.545	0.476
<i>Calla palustris</i> L.	S	Eme					0.333	0.686	0.364	0.450
<i>Utricularia vulgaris</i> L.	Ce	Hyd	4		I	T	0.604	0.600	0.455	0.443
<i>Carex acuta</i> L.	S	Eme	4	2			0.396	0.657	0.682	0.436
<i>Sparganium gramineum</i> Georgi	F	Hyd	3	3	I	I	0.667	0.571	0.773	0.423
<i>Schoenoplectus lacustris</i> (L.) Palla	H	Eme	6				0.500	0.800	0.273	0.422
<i>Nymphaea alba</i> ssp. <i>candida</i> (C. Presl & J. Persl) Korsh	F	Hyd	5		I	S	0.438	0.314	0.409	0.416
<i>Caltha palustris</i> L.	S	Eme	6	3			0.250	0.629	0.682	0.407
<i>Sparganium emersum</i> Rehmman	H	Eme	7			S	0.208	0.600	0.364	0.401
<i>Utricularia intermedia</i> Hayne	Ce	Hyd	1	4	I	I	0.292	0.143	0.409	0.400
<i>Nymphaea tetragona</i> Georgi	F	Hyd	4	4	I	T	0.583	0.686	0.455	0.395
<i>Lysimachia vulgaris</i> L.	S	Eme	5				0.292	0.457	0.455	0.390
<i>Potamogeton natans</i> L.	F	Hyd	5		I	S	0.604	0.886	0.636	0.389
<i>Juncus filiformis</i> L.	S	Eme					0.333	0.257	0.727	0.378
<i>Carex vesicaria</i> L.	S	Eme	5	2			0.146	0.371	0.818	0.375
<i>Cicuta virosa</i> L.	S	Eme	5	3			0.125	0.629	0.455	0.375

<i>Subuluria aquatica</i> L.	I	Hyd	1	4	S	T	0.354	0.171	0.545	0.375
<i>Lythrum salicaria</i> L.	S	Eme	5	3			0.292	0.514	0.409	0.374
<i>Lobelia dortmanna</i> L.	I	Hyd	1	5	S	S	0.896	0.429	0.455	0.362
<i>Isoetes lacustris</i> L.	I	Hyd	1	5	S	S	0.938	0.429	0.500	0.334
<i>Eleocharis acicularis</i> (L) Roem. et Schult.	I	Hyd	2	4	S	T	0.646	0.743	0.955	0.330
<i>Myriophyllum alterniflorum</i> DC.	E	Hyd	3	4	S	S	0.625	0.600	0.409	0.328
<i>Ranunculus peltatus</i> Schrank	E	Hyd	6		S	S	0.396	0.314	0.273	0.318
<i>Utricularia australis</i> R. Br.	Ce	Hyd	3		S		0.333	0.286	0.045	0.315
<i>Ranunculus reptans</i>	S	Eme		1			0.833	0.486	0.818	0.313
<i>Carex lasiocarpa</i> Ehrh.	S	Eme	3	4			0.708	0.629	0.227	0.305
<i>Sagittaria natans</i> Pall.	F	Hyd		2	I	S	0.104	0.286	0.682	0.297
<i>Menyanthes trifoliata</i> L.	S	Eme	3	4			0.250	0.229	0.000	0.287
<i>Elodea canadensis</i> Michx.	E	Hyd	7	2	I	S	0.167	0.371	0.091	0.282
<i>Fontinalis antipyretica</i> Hedw.	B	Hyd	5	4			0.104	0.343	0.227	0.257
<i>Warnstorfia trichophylla</i> (Warnst.) Tuom. & T. J. Kop.	B	Hyd		5			0.146	0.314	0.136	0.250
<i>Potamogeton berchtoldii</i> Fieber	E	Hyd	5	2	S	S	0.125	0.229	0.182	0.250
<i>Warnstorfia procera</i> (Renauld & Arnell) Tuom.	B	Hyd		4			0.208	0.000	0.091	0.248
<i>Potamogeton gramineus</i> L.	E	Hyd	5	3	S	I	0.125	0.143	0.318	0.247
<i>Persicaria amphibia</i> (L.) Delarbre	F	Hyd	4	2	I	S	0.125	0.200	0.500	0.241
<i>Drepanocladus longifolius</i> (Mitt.) Broth. ex Paris	B	Hyd		2			0.125	0.343	0.000	0.236
<i>Carex rostrata</i> Stokes	S	Eme	3				0.896	0.771	0.818	0.231
<i>Isoetes echinospora</i> Durieu	I	Hyd	1	5	S	I	0.917	0.714	0.727	0.229
<i>Utricularia ochroleuca</i> R. W. Hartm.	Ce	Hyd	1	4		T	0.167	0.000	0.091	0.224
<i>Juncus bulbosus</i> L.	E	Hyd	2	4		T	0.167	0.029	0.091	0.223
<i>Utricularia minor</i> L.	Ce	Hyd	2	4	I	T	0.146	0.057	0.091	0.215
<i>Lemna minor</i> L.	L	Hyd	6	2	T	S	0.021	0.400	0.273	0.208
<i>Sparganium angustifolium</i> Michx.	F	Hyd	1	5	S	T	0.083	0.200	0.227	0.199

<i>Carex aquatilis</i> Wahlenb.	S	Eme	4	2			0.063	0.143	0.636	0.199
<i>Fontinalis hypnoides</i> Hartm.	B	Hyd		2			0.042	0.343	0.273	0.186
<i>Elatine triandra</i> Schkuhr	I	Hyd	4	3	T	I	0.042	0.200	0.364	0.183
<i>Nuphar pumila</i> (Timm) DC.	F	Hyd	2	4	I	S	0.083	0.114	0.227	0.183
<i>Potamogeton obtusifolius</i> Mert. & W. D. J. Koch	E	Hyd	6	2	T	S	0.000	0.429	0.182	0.181
<i>Phalaris arundinacea</i> L.	S	Eme	7	2			0.063	0.200	0.227	0.170
<i>Hippuris vulgaris</i> L.	H	Eme	4	4		T	0.063	0.114	0.227	0.157
<i>Myriophyllum verticillatum</i> L.	E	Hyd	8	1	T	S	0.042	0.200	0.182	0.157
<i>Glyceria fluitans</i> (L.) R. Br.	H	Eme	7	2			0.083	0.057	0.182	0.157
<i>Carex elata</i> All.	S	Eme	5	4			0.083	0.057	0.182	0.149
<i>Iris pseudacorus</i> L.	H	Eme	7	2			0.042	0.200	0.182	0.142
<i>Scorpidium scorpioides</i> (Hedw.) Limpr.	B	Hyd		4			0.083	0.143	0.000	0.131
<i>Drepanocladus sendtneri</i> (Schimp. ex H. Müll) Warnst.	B	Hyd		2			0.063	0.143	0.091	0.131
<i>Calliergon megalophyllum</i> Mikut.	B	Hyd		3			0.042	0.200	0.091	0.121
<i>Nymphaea alba</i> ssp. <i>candida</i> x <i>tetragona</i>	F	Hyd					0.083	0.029	0.000	0.118
<i>Sagittaria sagittifolia</i> L.	H	Eme	6	2		S	0.000	0.143	0.318	0.114
<i>Phragmites australis</i> (Cav.) Trin. ex Steud.	H	Eme	7				0.979	0.943	0.682	0.101
<i>Hydrocharis morsus-ranae</i> L.	L	Hyd	6	1	T		0.000	0.171	0.182	0.095
<i>Nymphaea alba</i> ssp. <i>alba</i>	F	Hyd	5			S	0.063	0.029	0.000	0.091
<i>Littorella uniflora</i> (L.) Asch.	I	Hyd	2	4	S	S	0.063	0.000	0.091	0.090
<i>Potamogeton praelongus</i> Wulfen	E	Hyd	4	2	S		0.063	0.086	0.000	0.086
<i>Callitriche cophocarpa</i> Sendtn.	E	Hyd	4	3	T	S	0.000	0.143	0.182	0.086
<i>Elatine orthosperma</i> Düben	I	Hyd				I	0.021	0.000	0.273	0.085
<i>Callitriche palustris</i> L.	E	Hyd	3	3	I	T	0.021	0.057	0.182	0.082
<i>Typha latifolia</i> L.	H	Eme	8	2			0.042	0.114	0.000	0.082
<i>Elatine hydropiper</i> L.	I	Hyd	3	3	S	S	0.000	0.086	0.227	0.076

Warnstorfia exannulata (W. Gümbel) Loeske	B	Hyd		3			0.021	0.114	0.045	0.075
Nitella flexilis (Linné) Agardh	Ch	Hyd	4	2	S		0.042	0.086	0.000	0.074
Alopecurus aequalis Sobol.	H	Eme	9	3			0.021	0.029	0.182	0.073
Drepanocladus aduncus (Hedw.) Warnst.	B	Hyd		2			0.042	0.000	0.091	0.071
Eleocharis mamillata (H. Lindb.) H. Lindb. ex Dörfl.	H	Eme					0.021	0.000	0.182	0.067
Callitriche hamulata Kütz. ex W.D.J. Koch	E	Hyd		4		T	0.021	0.029	0.136	0.061
Fontinalis dalecarlica Bruch & Schimp.	B	Hyd		5			0.042	0.000	0.000	0.060
Callitriche hermaphroditica L.	E	Hyd	3	1	S	T	0.000	0.000	0.273	0.057
Nuphar lutea (L.) Sibth. & Sm.	F	Hyd	6		I	S	1.000	1.000	0.727	0.057
Sphagnum platyphyllum (Lindb. ex Braithw.) Sull. ex Warnst.	B	Hyd	3				0.021	0.057	0.045	0.057
Myriophyllum sibiricum Kom.	E	Hyd	4	1	S		0.021	0.057	0.045	0.053
Lysimachia thyrsoflora L.	H	Eme	4				0.979	0.971	0.909	0.052
Ceratophyllum demersum L.	Ce	Hyd	8	1	T		0.000	0.086	0.091	0.048
Sparganium hyperboreum Laest.	E	Hyd		5	S	T	0.021	0.000	0.091	0.046
Calliergonella cuspidata (Hedw.) Loeske	B	Hyd		3			0.021	0.029	0.045	0.044
Najas tenuissima (A. Braun) Magnus	E	Hyd	3	2			0.021	0.057	0.000	0.042
Comarum palustre L., (Potentilla palustris (L.) Scop.)	S	Eme	2				0.958	1.000	1.000	0.041
Calliergon cordifolium (Hedw.) Kindb.	B	Hyd					0.000	0.086	0.045	0.038
Lycopus europaeus L.	S	Eme	7	2			0.000	0.086	0.045	0.038
Ricciocarpos natans (L.) Corda	B	Hyd		2			0.000	0.029	0.136	0.038
Sparganium erectum L.	H	Eme	7	1			0.000	0.057	0.091	0.038

Sparganium natans L.	E	Hyd	3	4	I	S	0.000	0.057	0.091	0.038
Fontinalis dichelymoides Lindb.	B	Hyd		5			0.021	0.029	0.000	0.037
Chara fragilis Desvaux	Ch	Hyd			I		0.021	0.029	0.000	0.033
Riccia fluitans L.	B	Hyd		2			0.021	0.000	0.000	0.033
Potamogeton alpinus Balb.	E	Hyd	6		I	S	0.000	0.057	0.045	0.029
Potamogeton berchtoldii x obtusifolius	E	Hyd					0.021	0.000	0.000	0.027
Rorippa palustris (L.) Besser	S	Eme	8	2			0.021	0.000	0.000	0.027
Typha angustifolia L.	H	Eme	7	1			0.021	0.000	0.000	0.027
Equisetum fluviatile L.	H	Eme	5				0.979	1.000	1.000	0.026
Carex paniculata L.	S	Eme	4	1			0.021	0.000	0.000	0.026
Bidens radiata Thuill.	S	Eme		1			0.000	0.057	0.000	0.019
Bidens tripartita L.	S	Eme	8	2			0.000	0.029	0.045	0.019
Carex diandra Schrank	S	Eme	3	3			0.000	0.057	0.000	0.019
Nitella wahlbergiana Wallman	Ch	Hyd		2			0.000	0.000	0.091	0.019
Ranunculus lingua L.	H	Eme	7	2			0.000	0.057	0.000	0.019
Scirpus sylvaticus L.	S	Eme	4	3			0.000	0.029	0.045	0.019
Stratiotes aloides L.	Ce	Hyd	6	1	T		0.000	0.029	0.045	0.019
Bidens cernua L.	S	Eme	9	1			0.000	0.029	0.000	0.010
Calliergon giganteum (Schimp.) Kindb.	B	Hyd		2			0.000	0.029	0.000	0.010
Catabrosa aquatica (L.) P. Beauv.	S	Eme	8	1			0.000	0.000	0.045	0.010
Chiloscyphus polyanthos (L.) Corda	B	Hyd		4			0.000	0.029	0.000	0.010
Limosella aquatica L.	I	Hyd	3	2		T	0.000	0.000	0.045	0.010
Nitella opaca (Bruzelius) Agardh	CH	Hyd	2	2	S	I	0.000	0.029	0.000	0.010
Nuphar lutea x pumila	F	Hyd			S		0.000	0.029	0.000	0.010
Philonotis fontana (Hedw.) Brid.	B	Hyd					0.000	0.000	0.045	0.010
Rumex aquaticus L.	S	Eme	8	2			0.000	0.029	0.000	0.010
Sagittaria natans x sagittifolia	F	Hyd		2	T		0.000	0.029	0.000	0.010
Scolochloa festucacea (Willd.)	H	Eme		2			0.000	0.000	0.045	0.010

Link.							
Sphagnum riparium Ångstr.	B	Hyd	3	0.000	0.000	0.045	0.010
Typha angustifolia x latifolia	H	Eme		0.000	0.029	0.000	0.010
Warnstorfia fluitans (Hedw.) Loeske	B	Hyd		0.000	0.029	0.000	0.010

Appendix B. Spearman correlation coefficients of the environmental variables with ordination axes of NMDS analysis of reference and all study lakes.

Variable	REF lakes			All lakes		
	Axis 1	Axis 2	Axis 3	Axis 1	Axis 2	Axis 3
Area	-0.225	0.089	0.573	0.355	0.291	-0.486
Depth	-0.237	0.234	0.550	0.328	0.200	-0.210
Colour	-0.094	-0.598	-0.226	0.194	0.131	-0.054
<i>a</i> -Chloroph	0.103	-0.495	-0.069	0.414	0.296	0.189
totP	-0.104	-0.635	-0.085	0.435	0.289	0.080
totN	0.242	-0.521	0.174	0.498	0.316	0.344
N	-0.750	-0.269	-0.114	-0.031	0.017	-0.536
E	-0.491	0.055	-0.054	-0.242	-0.142	-0.312
Drawdown	-0.531	-0.434	0.237	-0.003	0.231	-0.390
Arable	0.143	0.179	0.593	0.408	0.441	0.249

V

**RESPONSE OF MACROPHYTE COMMUNITIES AND STATUS
METRICS TO NATURAL GRADIENTS AND LAND USE IN
BOREAL LAKES**

by

Janne Alahuhta, Antti Kanninen & Kari-Matti Vuori 2012

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Comparing long term sediment records to current biological quality element data – Implications for bioassessment and management of a eutrophic lake

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ABSTRACT

Defining reference conditions for lakes situated in areas of human settlement and agriculture is rarely straightforward, and is especially difficult within easily eroding and nutrient rich watersheds. We used diatoms, cyanobacterial akinetes, remains of green algae and chironomid head capsules from sediment samples of Lake Kirmanjärvi, Finland, to assess its deviation from the initial ecological status. These site-specific records of change were compared to current type-specific ecological status assessment. All paleolimnological data indicated deviation from natural conditions and mirrored the current, monitoring-based assessment of “moderate” ecological lake status. However, the sediment data showed that the lake should be re-typified as a naturally eutrophic lake. Sediment records as well as current monitoring data indicated temporary improvement in water quality in response to extensive fish manipulation. Our results suggest that paleolimnological records can be used to derive site-specific reference conditions and that extensive restoration efforts can result in gradual, observable improvements of water quality and ecological status.

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Introduction

Decision making in lake management is based on the interplay between current policy, the availability of resources and the environmental conditions at the site. One of the most challenging tasks for lake management and restoration is the selection of the target level, i.e. the desired level of attainable water quality. Target levels are often normatively specified in environmental legislation (e.g. U.S. Clean Water Act, EU Water Framework Directive, WFD) and set close to the estimated pre-disturbance conditions. In the European Union, the WFD (European Commission 2000) defines the management goal of ‘good status’ as a minor deviance of selected biological and abiotic quality elements from their respective reference states. In the ecological classification of lakes the assessment of this deviance is based on four biological quality elements (BQEs): phytoplankton, aquatic macrophytes and phytobenthos, benthic macroinvertebrates, and fish.

While intuitively simple, defining and quantifying pre-disturbance reference conditions in lakes is demanding because anthropogenic impacts often predate direct observations on the BQEs. Reference conditions can be based on data from lake type-specific pristine monitoring sites, predictive modelling or temporal changes i.e. historical changes or paleoreconstruction (European Commission 2003). Recent advances in studies of sediment records have promoted the use of paleolimnological methods to obtain information on past lake characteristics (Leira et al. 2006; Räsänen et al. 2006a). These quantitative paleoecological inference methods allow reconstructions of past lake nutrient levels, pH, conductivity, etc. and enable comparisons with direct observations. Current paleoecological inference models often use phytoplankton or chironomids as proxies. When compared to water samples, the sediment record of phytoplankton communities is often incomplete due to the poor preservation of certain types of algae. However, many important taxa deposit resistant, identifiable remains (e.g. Bacillariophyceae) which are sometimes identifiable to species level. Diatom assemblages are functionally important in many aquatic ecosystems and sensitive to environmental changes (e.g. Stoermer and Smol 1999). Similarly, cyanobacteria and green algae

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are important for practical lake management since their blooms often mirror eutrophication and indicate drastic changes in the overall composition of the phytoplankton community. While vegetative cyanobacterial cells do not preserve in lake sediments, certain cyanobacteria produce resting spores, i.e. akinetes, that are resistant to decomposition and have successfully been applied as a paleolimnological proxy (Cronberg 1986; van Geel et al. 1994; Kling 1998; Findlay et al. 1998; van Geel 2001; Eilers et al. 2004; Bradbury et al. 2004; Räsänen et al. 2006b). In Finnish freshwaters *Anabaena* spp. is the most common blooming cyanobacteria with well identifiable akinetes (Lepistö 1999) and *Scenedesmus* spp. and *Pediastrum* spp. are the paleolimnologically most extensively used green algae. A combined study of fossil diatoms, cyanobacteria, and green algae provides a fairly comprehensive picture of past phytoplankton assemblages.

Changes in lakes are not only reflected in phytoplankton but affect higher trophic levels such as chironomids as well. Subfossil head capsules of chironomids are present in all except highly anoxic sediments (e.g. Meriläinen et al. 2001). The composition of the profundal chironomid community is widely used to assess lake trophic status and several paleolimnological chironomid inference models have been constructed (e.g. Brodersen and Lindegaard 1999). In this study we relate past changes in the phytoplankton community to concurrent sediment data for the chironomid community.

The Iisalmi watercourse in Eastern Finland poses a unique challenge to the determination of lake management goals. Located in a geographic area where surface waters generally attain “good” or “excellent” status, these lakes are unexpectedly eutrophic and display phytoplankton blooms and extensive growth of macrophytes. The eutrophic state of these lakes may be related to watershed geology which is characterized by shallow basins and high proportions of fine-grained tills and fine grained sorted deposits. However, since these fertile soils rich in fines also attracted early human settlement and agriculture the main cause of the observed eutrophic state is difficult to determine.

The WFD-compliant ecological classification of Finnish surface waters is based on type-specific reference conditions derived from monitoring data of pristine reference sites for each surface water type (Vuori et al. 2009). The Finnish lake typology differentiates between 14 lake types mainly based on size, humic content, depth and natural eutrophy (Vuori et al. 2006). The exceptionally high amount of fine grained glacial and post glacial sediments in the watershed of the Iisalmi watercourse complicates the exact designation of individual lakes into lake types and, hence, the application of the correct type-specific reference conditions for ecological classification. Closely matching pristine reference lakes also are difficult to find for the region. We therefore used paleolimnological methods to assess the overall changes in the phytoplankton and chironomid community, identify algal blooms, and to infer the past nutrient status within one representative lake from the Iisalmi watercourse. By relating paleolimnological to contemporary monitoring data we further sought to (1) quantify the community changes caused by recent human activities, (2) assess the efficacy of past lake management actions and (3) evaluate the utility of sediment-based data in comparison to current data for reference condition-based ecological classification.

Methods

Study site

Lake Kirmanjärvi, located in Eastern Central Finland in the Iisalmi watercourse (Fig. 1), consists of two basins, the smaller Pieni-Kirma (31 ha, maximum depth 7 m) and the larger Kirmanjärvi (280 ha, maximum depth 9.7 m). The Finnish lake typology

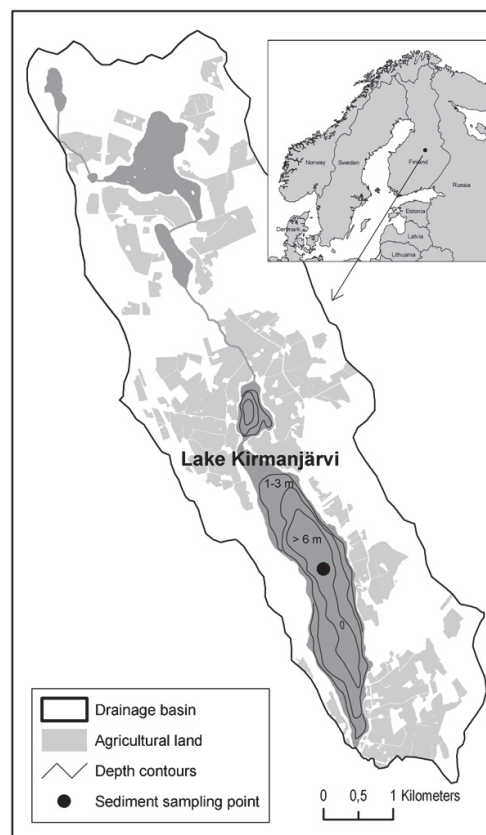


Fig. 1. A map of Lake Kirmanjärvi and its location in Fennoscandia.

system (Vuori et al. 2006) categorizes small lakes under 500 ha based on their humic content (measured by water colour), and mean depth. Naturally eutrophic lakes are identified based on their high epilimnetic turbidity ($FNU > 5$), high alkalinity ($> 0.4 \text{ mmol/l}$) or other information on natural eutrophy. Using the typology criteria of surface area, mean depth and water colour (area $< 500 \text{ ha}$, mean depth $> 3 \text{ m}$, water colour $30\text{--}90 \text{ mg Pt/l}$) Lake Kirmanjärvi is classified as a small humic lake in the Finnish system. Even though its geographic location suggests possible naturally high productivity Lake Kirmanjärvi does not meet the formal typology criteria for naturally eutrophic lakes. In addition to being representative for the Iisalmi watercourse the lake has regional importance because it serves as a reservoir for the production of drinking water and preserving good water quality (e.g. without toxic cyanobacterial blooms) attracts high public interest. Furthermore, the chemical water quality of the lake has been monitored intensively since 1986 and the lake has undergone different phases of restoration efforts, including intensive fishing period during 1987–1997 and hypolimnetic oxygenation since 1986. Further management efforts of the lake are currently being considered.

Coring, sampling and sediment dating

Sediment assemblages predating anthropogenic impacts were analyzed for algal remains and chironomid subfossils extracted from a 150 cm long composite sediment core sequence. Coring was performed from the lake ice on March 27th, 2006. We selected a

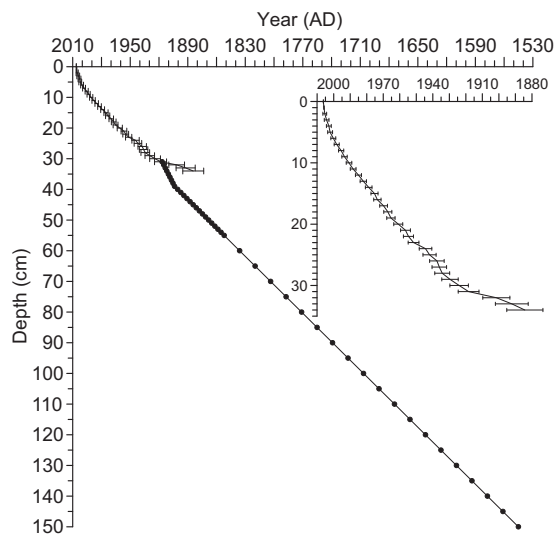


Fig. 2. Age-depth model for the Lake Kirmanjärvi core based on ^{210}Pb dating results and the CRS model (lines with error bars). The inset shows the results for the upper 34 cm in more detail. The dotted line shows an extension of the dating results based on an estimated 2-fold accumulation rate between 39 and 31 cm and average accumulation rate below 39 cm.

coring site from the middle part of the Kirmanjärvi basin (water depth 7.2 m, Fig. 1) because of the installation of a hypolimnetic oxygenator into the deepest part of the basin in 1986.

Sediments were first cored with a Limnos gravity corer which preserves the sediment-water interface (Kansanen et al. 1991). The 42 cm long Limnos core was sliced continuously into 1 cm thick slices that were stored in plastic bags in the field. A long continuous Kullenberg type PP core (712 cm, top 150 cm used here; Putkinen and Saarelainen 1998) was taken from the same area. The two cores were later correlated based on their loss on ignition and ^{137}Cs profiles. The PP core tubes and the Limnos core slices were stored in a cold room at +4 °C until subsampled for analyses.

The Limnos core was dated using artificial ^{137}Cs and naturally occurring ^{210}Pb radionuclides (Robbins 1978; Appleby and Oldfield 1992; Crusius and Anderson 1995; Appleby 1998; Mattila et al. 2006). The activities of radionuclides were determined by gamma-ray spectrometry (Klemola et al. 1996). The major source of the ^{137}Cs activities in the study area has been the fallout from the Chernobyl accident in 1986 and the highest measured activity concentration of ^{137}Cs in the sediment profile was used as a time marker for that year. Sediment cores were dated with the CRS (Constant Rate of Supply) model (Appleby and Oldfield 1978), but the simple "constant flux: constant sedimentation rate" model was also used to compare dating results. The final age model was constructed by extending the radiometrically obtained average sediment accumulation rate to the lower part of the core, with the exception of the 30–40 cm mineral rich section for which twice the average rate was used as a rough estimate in the absence of measured data (see Fig. 2).

Sediment chemistry

Analyses of sediment chemistry were employed to provide information on changes in sediment quality (due to e.g. erosion, organic inputs, resuspension) and nutrient or metal inputs. Sediment chemistry was analyzed from 85 samples down to 150 cm in sediment. Samples were digested with HNO_3 in a microwave

oven and the resulting extracts analyzed with the ICP-OES technique, a modification of the US EPA method 3051 (US EPA 2007). The digestion does not dissolve the major silicates but breaks down trioctahedral micas, clay minerals, carbonates (excluding magnesite), titanite, oxyhydroxide precipitates, etc. and liberates elements from the organic matter of lake sediments. Besides the determination of the 'contaminated fraction' that the extraction was originally developed for, the leach allows determination of total P concentrations and captures variation in the fine-grained mineral fraction as well as hydroxide precipitates and sulfide minerals.

Paleobiological analyses

Diatoms were analyzed from 27 samples of the sediment column with increasing sample spacing down-core (1 cm, 2 cm, 5 cm, and after that at 15 cm spacings, see Figs. 5 and 6). Diatom slides were prepared according to Battarbee et al. (2001) and analyzed with a Leitz Diaplan light microscope at 1250 \times final magnification using both phase contrast and bright field illumination. Identification of 300–350 valves/slide were performed to species level when possible following Krammer and Lange-Bertalot (1986, 1988, 1991a, 1991b).

A total of 85 samples were analyzed for cyanobacterial akinetes and fossil green algae. These remains were analyzed from the 42 cm Limnos core at 1 cm intervals and the 150 cm PP core at 2.5 cm intervals beginning from 45 cm. First, 0.5 ml of fresh sediment was weighted and diluted to 50 ml using distilled water. Well-mixed subsamples of 0.25 ml were taken for microscopy from this diluted sample. The akinetes and green algae remains were counted with an inverse microscope using phase-contrast illumination at 800 \times magnification applying Utermöhl's technique (Utermöhl 1958). Akinetes and green algae were counted from randomly selected fields, and the counting was stopped when 1000 fields were investigated using a technique modified from Kling (1998). The results were given as number of remains per gram of dry sediment.

Altogether 27 subsamples from the sediment core were taken for chironomid analysis with increasing sample spacing down core. Chironomid samples were deflocculated in 10% KOH solution at room temperature for 12 h, sieved (100 μm), preserved in 70% ethanol and picked under a stereomicroscope. The numbers of chironomid head capsules per subsample ranged from 21 to 70. The 1136 head capsules belonging to 27 chironomid taxa were identified mainly to species or genus level mainly according to Wiederholm (1983).

Current biological, physical and chemical monitoring data

Biological data used for the present ecological classification of Lake Kirmanjärvi was sampled in 2005–2006. In October 2005, six replicate Ekman grab samples were taken outside the effective range of the hypolimnetic oxygenation device. Ekman grabs were sampled from depths ranging between 7.6 and 8.0 meters. The benthic fauna was identified according to Wiederholm (1983).

Aquatic macrophytes were surveyed in the summer of 2006 from transects at 15 sites using a method described in detail by Leka et al. (2003). A vegetation abundance metric index (Ilmavirta and Toivonen 1986) was calculated from these transects using the taxonomical resolution used in Kuoppala et al. (2008).

Fish were sampled with Nordic gillnets in June–September 2006 using a stratified random sampling protocol described in detail by Olin et al. (2004). The total fishing effort was 15 gillnet nights (Kauppinen 2006).

Physical and chemical water quality parameters of Lake Kirmanjärvi have been monitored during the productive period since 1986. On 3–4 occasions annually, basic parameters e.g. total and

inorganic nutrients, conductivity, oxygen and COD were measured. In addition, surface water chlorophyll-*a* was measured during all summertime samplings. Phytoplankton species biomass estimates, calculated by the Utermöhl's technique (Utermöhl 1958), were available from 2001 to 2006 from the phytoplankton database of the Finnish Environment Institute.

Numerical methods

Results of the diatom analyses were visualized using ordination plots created with the CANOCO for Windows 4.5 software package (ter Braak and Šmilauer 2002) and stratigraphic diagrams created with C2 (Juggins 2003). Because an initial detrended correspondence analysis (DCA; Hill and Gauch 1980) run showed that the length of the species gradient was less than 2 SD units, the linear response-based method of principal components analysis (PCA) was used. Stratigraphically constrained cluster analysis was used to help divide the sediment diatom assemblages into local assemblage zones. Cluster analysis was run using the paired group method on Euclidean and chord distance dissimilarity measures (Past ver. 1.99; Hammer et al. 2001). To describe the deviation of the diatom phytoplankton community from the site specific reference conditions, a local diatom Percent Model Affinity curve (PMA; Novak and Bode 1992) was generated using average abundance data from the lowermost diatom zone (150–60 cm) as the model community from which percentage difference was calculated for each sample ($100 - 0.5 \cdot \sum (\text{taxon's relative abundance in model community} - \text{taxon's relative abundance in sample})$).

The past diatom-inferred epilimnetic total phosphorus (DI-TP) concentration was reconstructed using the calibration set and inference model presented by Kauppila et al. (2002). The model was generated with simple weighted averaging regression (ter Braak and van Dam 1989) and classical deshrinking. Lake Kirmanjärvi is located within the geographical area covered by the calibration set and the lake fulfils the selection criteria of the calibration set lakes (size, pH and water colour ranges).

The Shannon–Wiener diversity index (H) for the chironomid data used untransformed counts and the Benthic Quality Index (BQI) was calculated according to Wiederholm (1980). To assess whether the diatom zonation was reflected in the structure of the chironomid community as well we ran the non-parametric analysis of similarity (ANOSIM, Clarke 1993) on log-transformed abundance data. In addition, we tested whether the onset of intensive farming in the 1960s was reflected in the composition of the chironomid fauna. ANOSIM tests for differences between groups in multivariate data sets and is thus vaguely analogous to analysis of variance in univariate statistics. ANOSIM computes *R*, which indicates the magnitude of difference between groups. An *R* value of 1 indicates that the communities completely differ between defined groups, whereas an *R* of 0 indicates no difference. Bray–Curtis distance measures were used in ANOSIM and Bonferroni corrected *P*-values were used to assess the statistical significance of *R*. To evaluate the contribution of individual species to the dissimilarity between predefined groups we used SIMPER (Clarke 1993).

Ecological Quality Ratios (EQRs) were derived for each metric in the current monitoring data by comparing the observed (O) value of each metric to the expected (E) values from reference conditions according to the Finnish national ecological classification system (Vuori et al. 2009). The EQRs were calculated employing the reference values for both “small humic” and “naturally eutrophic” lake types. The resulting EQRs were divided into five classes: high, good, moderate, poor and bad (class boundaries 0.8, 0.6, 0.4, and 0.2). The classification system integrates multiple metrics and different biological quality elements (BQE) by scoring the individual metrics and the overall lake status class is given as a median score across different BQEs. A more thorough presentation of the Finnish eco-

logical classification system, with examples of application, is given by Alahuhta et al. (2009a).

Results

Sediment characterization and dating

The sediment profile (150 cm) consisted of homogenous grayish brown silty gyttja up to the depth of 50 cm. The mineral content of the sediment gradually increased to the depth of 35 cm where we observed a shift to higher organic content and from 31 cm upwards the sediment turned black due to sulfide coloring. Another increase in organic content was seen at 18 cm and patches of oxidized iron were observed at sediment depths between 26 and 12 cm. A thin veneer of oxidized iron was also observed at the sediment water interface in the gravity core. Fig. 2 shows the radiometric dating results and the estimated age-depth model for the core, indicating that the core section extends to mid sixteenth century. Both the Cs-137 and Pb-210 determinations were made from samples between 0 and 43 cm and a clear Chernobyl fallout peak was detected at the 12–15 cm slice. Pb-210 provided dates down to 36 cm below which concentrations declined to very low levels.

Sediment chemistry

To obtain information on external forcing events such as changes in sediment quality and possible metal or nutrient loading, changes in sediment chemistry are presented for selected elements in Fig. 3. Concentrations of the major redox-sensitive elements Fe and Mn, as well as phosphorus, which is often bound to Fe–Mn precipitates, were high in the lowest part of the sediment column, between 150 and 100 cm. In the 100–55 cm sediment section, concentrations of Fe, Mn and P were lower despite a peak in mineral matter content at the depth of 85 cm and the subsequent increase in indicator elements for fine grained mineral matter such as Al, K and V.

A second period of increasing mineral matter content was found from 55 cm upwards with a peak between 35 and 31 cm. Iron concentrations were high in these samples, while no concurrent peaks in Mn and P were present. In the topmost part of the sediment column the share of mineral matter declined (above 31 cm), whereas concentrations of redox and oxyhydroxide-related elements gradually increased. Manganese, which is more susceptible to reduction and mobilization in sediments than iron, peaked at 2–3 cm depth. Superimposed on this trend was an increase in elements often associated with fossil fuels and atmospheric deposition: Pb, S, Ni and V. The trends for Ni and V became evident only after the data was normalized using the K content to account for the fine grained minerals that typically also contain both Ni and V. Sulfur, a major constituent in poorly crystalline metal monosulfides of many sediments, peaked along with iron concentrations at 14–15 cm depth. High concentrations of elements indicative of air emissions were found in sediments deposited in the early 1980s. Subsequently, their fallout has reduced considerably because of reduced emissions from fossil fuel-based power generation and the ban of lead in gasoline.

Diatoms

Results of the diatom analyses are presented in Fig. 4 and summarized as a PCA plot in Fig. 5. The diatom stratigraphy was divided into three distinct local diatom assemblage zones (LDZ) with the aid of cluster analysis results. The lowermost diatom zone (LDZ 3) extended from 150 to 60 cm in the sediment core (~AD 1545–1836) and mirrored observed changes in sediment chemistry (150–100 cm, 100–55 cm). LDZ 3 was characterized

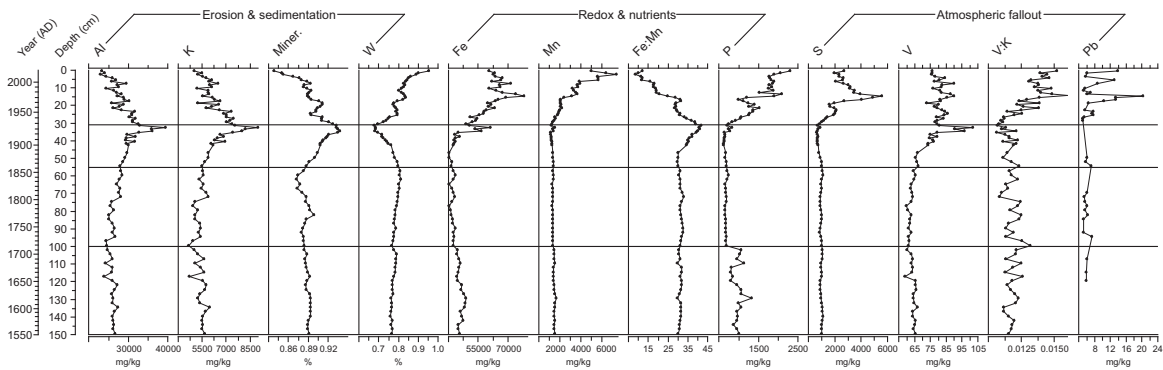


Fig. 3. Distribution of selected elements and element ratios in the core.

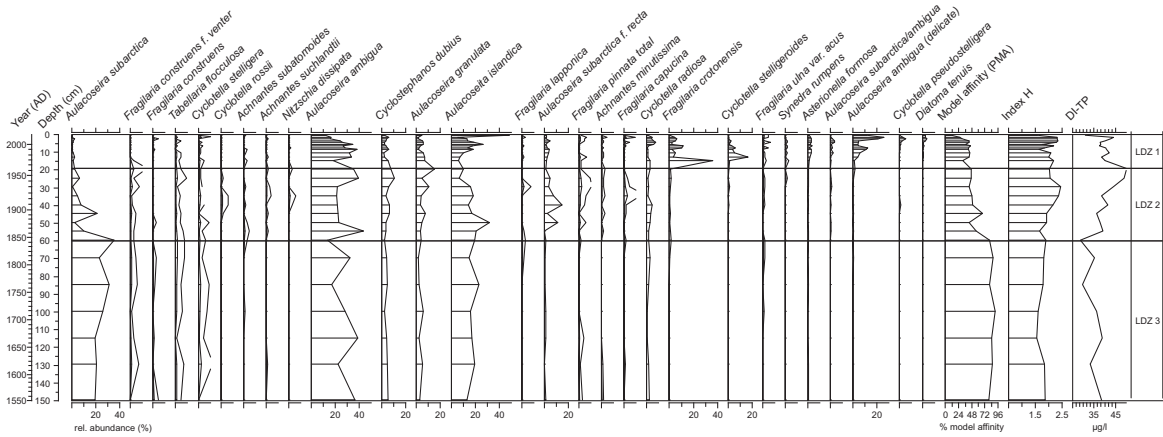


Fig. 4. Relative abundances of the most common diatom taxa in the Kirmanjärvi core. The species have been arranged according to their abundance weighted average depth of occurrence. Exaggeration line multiplier is 5. Percent model affinity, Shannon Index H, diatom-inferred lake water [TP] and Local diatom assemblage zones (LDZ) are also shown.

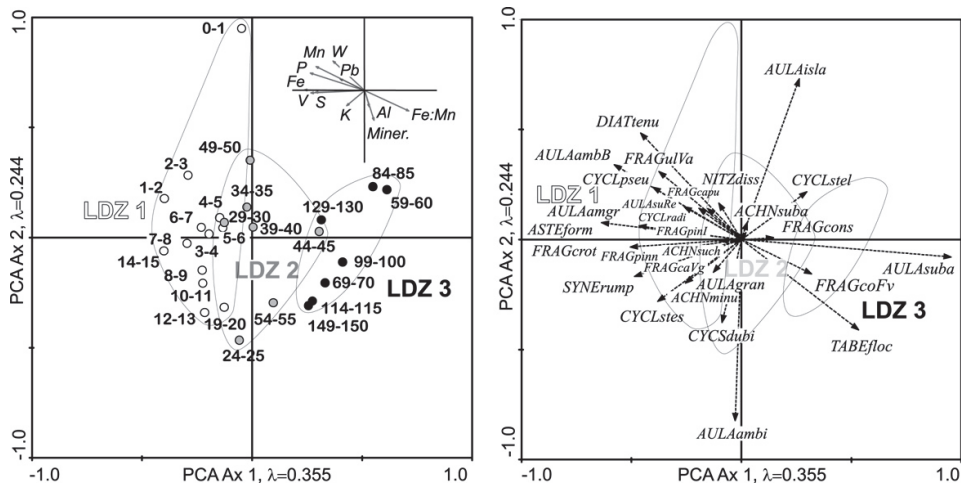


Fig. 5. A PCA plot of the diatom samples (left panel) and species (right panel). The diatom assemblage zones are indicated with symbol shading (black = LDZ 3, white = LDZ 2, medium grey = LDZ 1). The inset shows major directions in selected sediment variables superimposed on the diatom ordination.

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by high abundances of species that prefer mesotrophic waters such as *Aulacoseira subarctica* (~25%) and *A. islandica* (~18%). In addition, several taxa characteristic of eutrophic waters occurred in high numbers: *A. ambigua* (~30%), *A. granulata* (~5%), and *Cyclostephanos dubius* (~5%). The diatom-inferred TP concentration was in the eutrophic range for this zone (29–38 µg/l) and Shannon's Index H values ranged between 1.55 and 1.90. In general, species assemblages were fairly stable in LDZ 3, despite the decrease in sediment-bound P at 100 cm (~AD 1710) and a short-lived peak in mineral matter content at 85 cm (~AD 1755). The average PMA value for samples in LDZ 3 was 84%, which is the site-specific (local) reference PMA used in this study.

The delineation of the next diatom zone (LDZ 2, 55–19 cm, ~AD 1850–1965) coincides with the incipient increase in sediment mineral matter content. In this zone, the relative abundance of *A. subarctica* declined and a less thinly silicified form of the species increased. This delicate form of *A. subarctica* is often encountered in cases of eutrophication (e.g. Kauppila et al. 2002). *A. islandica*, another mesotrophic species, declined as well after an initial peak at 50 cm. With the decrease in the two abundant mesotrophic forms, species typical of eutrophic waters such as *A. ambigua*, *A. granulata* and *C. dubius* increased their share. At the same time, the proportions of certain non-planktonic *Achnanthes* and *Fragilaria* species increased. The diatom-inferred TP concentration increased to a maximum of 50 µg/l at 19–20 cm sediment depth. Even though the initial shift in species compositions was abrupt, the PCA plot indicates that the overall change in assemblages was moderate (Fig. 5). The PMA values for LDZ 2 ranged between 44 and 68% (i.e. 52–81% of the background PMA) and with the addition of taxa to the assemblages the values of Shannon's H increased to a maximum of 2.44 at 30 cm depth. In the ordination plot, ranges of LDZ 3 and 2 partly overlap although the first PCA axis separates the diatom zones fairly well.

The transition to LDZ 1 (19 cm – top, ~AD 1965–) coincided with a moderate peak in sediment mineral matter content. Similar to the LDZ 3–LDZ 2 transition, samples of LDZ 2 and LDZ 1 overlap somewhat in the ordination plot (Fig. 5). In the uppermost samples from LDZ 2, *A. ambigua*, *A. granulata*, and *C. dubius* increased in abundance while *A. islandica* continued to decline and this shift to more eutrophic species compositions continued in LDZ 1. Diatom-inferred TP subsequently declined from the very high values observed at the LDZ 2 to LDZ 1 transition but remained at a level of 40 µg/l. The division between LDZ 2 and LDZ 1 was the most distinctive separation in all of the cluster analysis runs and several 'new' species increased their proportion in the assemblages: *Fragilaria crotonensis*, *Cyclotella stelligeroides*, *Synedra rumpens*, *Asterionella formosa*, and a small form of *A. ambigua*. Concomitant with the increases in these eutrophic planktonic species was the decline in species such as *A. subarctica* and *Fragilaria construens* f. *venter*, and many of the *Achnanthes* and *Fragilaria* taxa that increased in LDZ 2. The species composition in the most recent samples revealed increases in *Fragilaria capucina* var. *gracilis*, *Cyclotella pseudostelligera* and *Diatoma tenuis*. Species shifts were reflected as decreases in the PMA curve, especially in the uppermost three samples (average PMA for the zone 39%, i.e. 47% relative to background PMA). In contrast, the Shannon diversity showed no corresponding declines and attained high values ($H = 2.30$ – 2.35) near the sediment surface.

Cyanobacterial akinetes and green algae

Results of the cyanobacterial akinete, *Pediastrum* spp. and *Scenedesmus* spp. analysis are presented in Fig. 6. Cyanobacteria akinetes were present in every sample studied. The majority of akinetes belonged to the genus *Anabaena* but also akinetes typical of less nutrient-rich waters such as the genus *Aphanizomenon* were

found. *Anabaena* spp. akinetes were most abundant at sediment depths from 90 to 50 cm (~AD 1740–1870). Towards the sediment surface their amount decreased. By contrast, *Aphanizomenon* spp. akinetes were present in large amounts at the bottom of the sediment core, decreased in abundance between sediment depths of 130 and 30 cm, whereafter they gradually became abundant again.

A large variation in the number of green algae remains was observed, as some samples contained no subfossils, while large amounts of remains were present in adjacent samples. Both *Pediastrum* spp. and *Scenedesmus* spp. remains displayed a slight increase in their abundance at sediment depths from 80 to 60 cm after which their amount first decreased, to increase again from 20 cm towards the sediment top.

Chironomid subfossils

An overview of the chironomid species composition along the sediment profile is presented in Fig. 7. The most abundant taxa throughout the sediment record were *Procladius* spp. and *Chironomus plumosus*, both indicative of eutrophic conditions, along with *Tanytarsus* spp., a genus containing mainly littoral species. Overall, the observed fauna was mainly littoral and indicative of meso-eutrophic conditions. Chironomid species indicative of oligotrophic conditions were lacking and indicators of mesotrophy (*Sergentia coracina*, *Stictochironomus rosenscholdii*) were scarce throughout the sediment profile.

The chironomid fauna did not follow the compositional changes of the local diatom assemblages, i.e. there were no significant differences between any of the LDZ-based sections (ANOSIM, $R < 0.22$, $P > 0.089$). Overall, SIMPER found the dissimilarity of the chironomid species composition to range between 38 and 40% between the diatom zones (LDZs). The most notable difference between the modern intensive farming period (0–19 cm, ~AD 1965– present) and the preindustrial (>55 cm; ~AD 1850–1545) era was the marked reduction in *Chironomus thummi* type (contribution to observed dissimilarity 6%) mirrored by the concurrent increase in *Chironomus plumosus* (contribution to dissimilarity 4%). Contrary to the results for the LDZ division, we noticed a distinct difference in chironomid communities only when comparing the modern intensive farming period (0–19 cm, ~AD 1965–) to the rest of the sediment column (ANOSIM, $R = 0.21$, $P < 0.0005$). SIMPER identified differences in densities of *Chironomus plumosus*, *Procladius* spp., *C. thummi* type, *Glyptotendipes* and *Tanytarsus* spp. to account for roughly half of the total observed dissimilarity of 41%.

Temporal changes in the number of chironomid head capsules relative to the sediment dry weight (g^{-1}) mirrored the diatom based zonation (Fig. 7). The BQI, however, did not differ between the LDZ (one way ANOVA, $F_{2,25} < 0.2$, $P > 0.81$), while Shannon–Wiener diversity and the total number of chironomid species differed significantly between the diatom zones (one way ANOVA $F_{2,24} > 7.7$, $P < 0.003$). We observed significantly fewer species in the LDZ 1 (mean = 7.2) compared to the LDZ 2 (mean = 11.5) and Shannon Wiener diversity was lower when comparing LDZ 1 to LDZ 2 or LDZ 3 (Post hoc Bonferroni $P < 0.004$; Fig. 7). A separate comparison of the modern era species richness and Shannon Wiener diversity to the rest of the sediment column also showed significant differences (one way Anova, $F_{1,25} > 14.8$, $P < 0.001$). The local chironomid model affinity remained high, above 80% of the local reference PMA of 76%, until the uppermost 5 cm, where it declined to 65–50% of the reference in three samples.

Ecological status according to current monitoring data

The modern biota of Lake Kirmanjärvi was dominated by taxa preferring meso-eutrophic conditions in all the taxonomic groups studied. Cyanobacteria were abundant in the pelagic phytoplank-

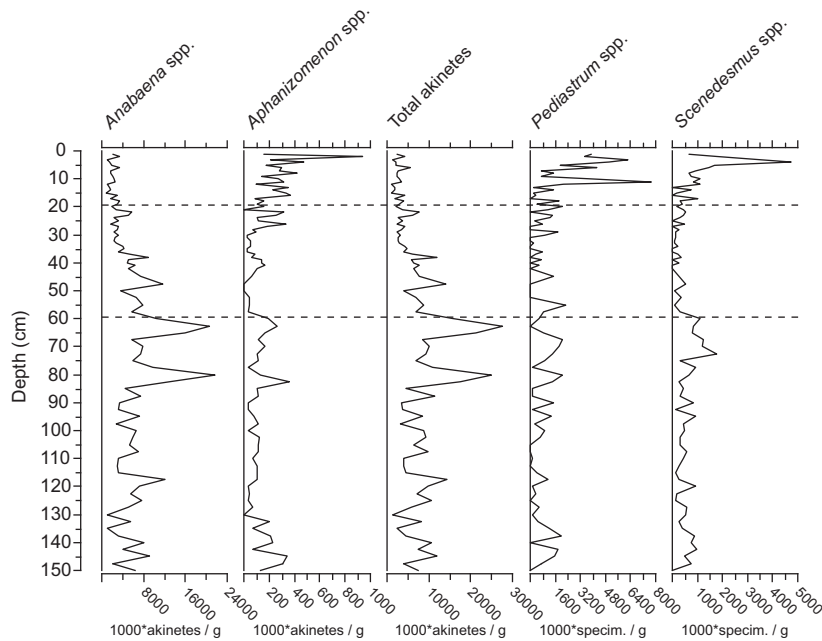


Fig. 6. Distribution of *Anabaena* and *Aphanizomenon* akinetes and chlorophyte remains in the Lake Kirmanjärvi core. Dashed lines show diatom zone boundaries for comparison.

ton community and the overall phytoplankton biomass was high (5.1 mg/l). A relatively high number of macrophyte species (total 48 taxa) were recorded, of which 37% (18 taxa) are considered indicators of meso-eutrophy or eutrophy (Leka et al. 2008). Chironomid larvae dominated the benthos (42% of individuals) with *Procladius* sp. and *Chironomus anthracinus* being the two most abundant taxa. The second most abundant benthic group were the oligochaetes (40%), which are indicative of low oxygen conditions. The cyprinid fish like roach (*Rutilus rutilus*) and bream (*Abramis brama*) accounted for 51% of biomass per unit effort (BPUE). Per-

cids like perch (*Perca fluviatilis*) and ruffe (*Gymnocephalus cernuus*) made up 37% of the total BPUE.

Lake Kirmanjärvi was classified to be in a “moderate” ecological state using type specific reference conditions for either the small humic lake type or the naturally eutrophic lake type (Table 1). The choice of typology influenced the resulting overall lake EQR only minutely. Designating the lake to the small humic lake type produced a slightly lower EQR than designation to the naturally eutrophic lakes. This difference was small and resulted from type specific results for the fish community. The benthic macroinver-

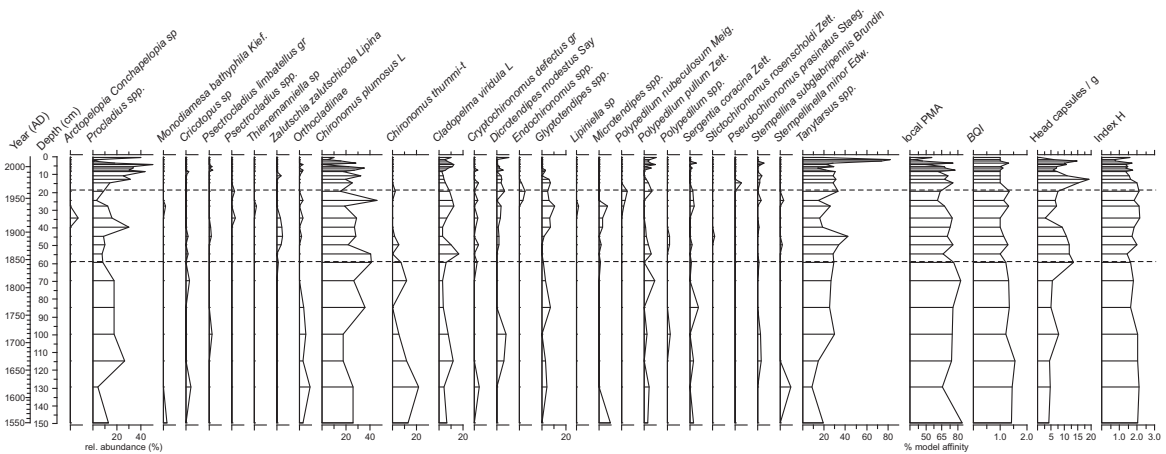


Fig. 7. Relative abundances of the most common chironomid taxa in the Kirmanjärvi core. Percent Model Affinity (site-specific), Benthic Quality Index, concentration of head capsules and Shannon Index H are also shown. Dashed lines show diatom zone boundaries for comparison.

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Table 1

Lake Kirmanjärvi ecological status according to the Finnish ecological classification system, given separately for Small Humic and Naturally Eutrophic lake types. H = High, G = Good, M = Moderate, P = Poor. *Observed values differ for two of the macrophyte metrics depending on lake type assignment. N/BPUE = number/biomass per unit fishing effort.

Classification metrics			Ecological classification using type-specific reference conditions of small humic lakes			Ecological classification using type-specific reference conditions of naturally eutrophic lakes			
BQE	Metric (unit)	Observed value	Metric EQR score/status class	BQE median EQR score/status class	Lake status EQR score/class	Metric EQR score/status class	BQE median EQR score/status class	Lake status EQR score/status class	
Phytoplankton	Biomass (mg/l)	5.1	0.5/M	0.5/M	0.5/M	N.A.	0.5/M	0.6/M	
	Cyanobacteria (%)	19.8	0.7/G			N.A.			
	<i>a</i> -chlorophyll (μ g/l)	26.5	0.5/M			0.5/M			
Macrophytes	Proportion of type-specific-species (%)*	29/35	0.3/P	0.5/M		0.5/M	0.5/M		
	RI (-)	14.29	0.7/G			0.7/G			
	PMA (-)*	38.27/30.04	0.5/M			0.5/M			
Macro-invertebrates	BQI-1 (-)	0.86	0.9/H	0.9/H		0.9/H	0.9/H		
	PMA (-)	0.615	0.9/H			N.A.			
Fish	BPUE (g/night)	1423	0.7/G	0.6/M		0.9/H	0.7/G		
	NPUE (n/night)	131.3	0.3/P			0.5/M			
	Cyprinids (%)	50.3	0.9/H			0.9/H			
	Indicator species (expert judgement)	Normal pop. structure of <i>P. fluviatilis</i> , <i>E. lucius</i> and/or <i>R. rutilus</i>		0.5/M			0.5/M		

tebrate community classified Lake Kirmanjärvi to “high” ecological status irrespective of the classification criteria used. Phytoplankton and macrophytes indicated an elevated level of primary production and consequently classified the lake to “moderate” status for both typologies.

Responses to past management measures

A decrease in chlorophyll-*a* and total phosphorous concentration, as well as a subtle increase in secchi depth, can be observed in the water quality data from the late 1980s to 1998 (Fig. 8). The changes coincided with the onset of the hypolimnetic oxygenation (since 1986) and a period of intensive fishing (during 1989–1997; Kauppinen 2006). At the end of the biomanipulation period, the chlorophyll-*a* content improved to “good” (20 μ g/l) status irrespective of lake type designation. Following the cessation of intensive fishing in 1997, the chlorophyll-*a* and phosphorous concentrations gradually rose while secchi depth decreased (Fig. 8). In the sediment record effects of the lake management period correspond to the upper 12 cm (Fig. 2).

Discussion

Indications of external forcing

The chemical and physical properties of the sediment suggested certain periods of external forcing on Lake Kirmanjärvi that could have affected lake biota and status class. The earliest period of external forcing started around AD 1710 (at 100 cm) with a sudden change in sediment phosphorus binding mode. While the exact reasons leading to these results are unknown, early human settlement has the potential to cause observable changes in lakes (e.g. Miettinen et al. 2002; Heinsalu and Aliksaar 2009). The next disturbance phase manifested itself as a short-lived peak in mineral matter contents at 85 cm (~AD 1755). However, no drastic biological changes were observed at these sediment depths. Major

increases in mineral matter inputs started at 55 cm and peaked at 35 cm (~AD 1850–1910), corresponding to changes in diatom assemblages. Changes in the sediment chemistry were most likely related to deforestation and agriculture in the catchment which increased overall nutrient inputs to lakes. Sediment phosphorus concentrations increased from 40 to 30 cm (~AD 1900–1925) with the intensification of agriculture. The sediment nutrient trend did not mirror the rapid changes in biota at 19 cm but stable and high P concentrations found in the top 12 cm of sediment correspond to the period of artificial hypolimnetic aeration, and may indicate an increase in the sediment P binding capacity. Trends for Pb, S, Ni and V in the early 1980s clearly indicate an increase in fossil fuel-related atmospheric fallout, but this type of forcing is likely to have a negligible effect in eutrophic lakes such as Lake Kirmanjärvi.

Phytoplankton

Paleoecological diatom data show marked changes in diatoms species assemblages from the site specific reference conditions, coinciding with the stages of external forcing caused by human activity. However, even the oldest sediment level analyzed contained species assemblages indicative of high nutrient concentrations (modeled lake water [TP] of 29–38 μ g/l), suggesting that the lake may be best classified to the naturally eutrophic type. Changes observed in the reference period sediment properties at 100 cm and 85 cm depths did not cause detectable species shifts. In contrast, the increase in inferred phosphorus concentrations coincided with the first signs of agriculture at 55 cm (~AD 1850) and resulted in a major shift in diatom species compositions (PMA ~50%, i.e. 60% of the reference PMA = moderate class) but no decline in species diversity. Model affinity is not employed as a classification metric for phytoplankton in the Finnish system (Table 1) but we interpreted the local diatom-PMA results analogously to other metrics for which it is employed. Also the next major shift in diatom species composition (19 cm, ~AD 1965; PMA 47% of the reference PMA = moderate class) coincided with a peak in mineral matter and

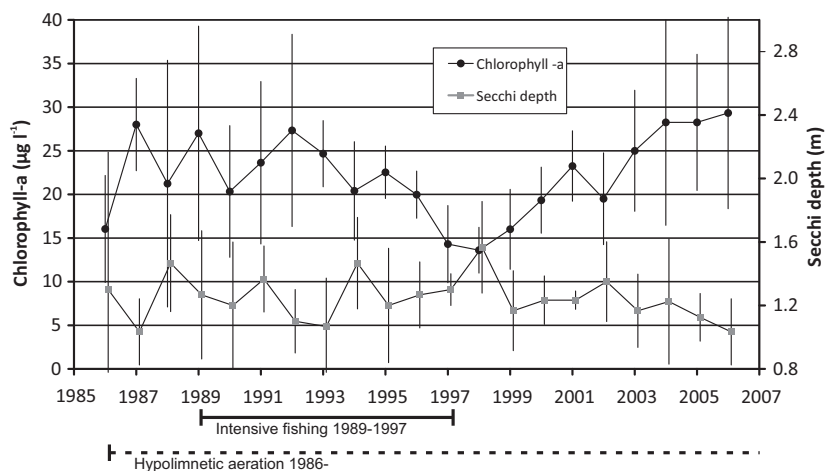


Fig. 8. Changes in chlorophyll-*a* content and secchi depth (annual mean \pm SD) during the productive period (May–September) in Lake Kirmanjärvi between 1986 and 2006.

a transient decline in sediment P concentrations with respect to the phosphorus binding elements Fe and Mn. Overall, the site-specific history of these externally forced changes in the diatom assemblages support the assignment of the 'moderate' status class to Lake Kirmanjärvi based on modern observational data. The situation was not reversed by the management efforts, even though the increases in *Asterionella formosa* in the uppermost 13 cm may be related to the hypolimnetic aeration which is known to favor early blooming species (Webb et al. 1997). It has been shown that oxic sediment surfaces resulting from hypolimnetic aeration may not be effective in enhancing permanent phosphorus retention in lakes (Gächter and Müller 2003; Gächter and Wehrli 1998).

Long term changes in the non-diatom phytoplankton community were substantial in the upper 20 cm of the sediment core (corresponding to LDZ 1). In accordance with the diatom record, the green algae *Pediastrum* spp. and *Scenedesmus* spp. increased, suggesting gradual eutrophication since the 1960s. *Anabaena* akinetes declined simultaneously, a phenomenon which has been attributed to progressive eutrophication (van Geel et al. 1994). However, a concomitant increase rather than a decrease was observed for *Aphanizomenon*, a cyanobacterial species that thrives in lower nutrient concentrations than *Anabaena* but gains a competitive advantage in shaded conditions (de Nobel et al. 1998). Regardless of the cause of these shifts, major deviations from the reference conditions were observed for cyanobacteria and green algae in sediments corresponding to LDZ 1. Elevated concentrations of akinetes and remains of green algae were also found in older sediments, at a depth of 80–60 cm (~AD 1770–1840). These increased concentrations were mirrored only by the chironomid proxy as slightly elevated proportions of *Chironomus plumosus*. The changes may be anthropogenic in origin, but climatic causes cannot be ruled out either. The observed deviations thus support the view of lower than good status class for the study lake based on phytoplankton assemblages. However, akinetes especially of the bloom forming taxon *Anabaena* spp. were found throughout the core, which suggests that also cyanobacteria were present and eutrophic conditions prevailed during the whole study period.

Benthic macroinvertebrates

Dominant chironomid taxa in all samples included *Procladius* spp, *Chironomus plumosus* and *C. thummi* type (probably *C. anthracinus*), and *Tanytarsus* spp. In addition, samples included also

other taxa often associated with shallow and eutrophic lakes (e.g. Brodersen and Quinlan 2006).

We found no concordant changes of the chironomid fauna with the diatom zones i.e. adjacent LDZs did not differ from another with respect to chironomids. This finding is not uncommon as diatom trophic trends can differ from patterns observed for chironomids (Meriläinen et al. 2001, 2003). However when we compared the period of intensive fertilizer-based farming (~ AD 1965–; equivalent to LDZ 1) to the rest of the sediment core distinct changes in the chironomid fauna became obvious. Modern sediments displayed a clear reduction in the abundances of *Chironomus thummi* type and an increase in *Procladius* spp. pointing to prolonged periodic hypolimnetic anoxia. However, the local relative PMA for chironomids suggested a lower than high status only for three samples in the upper 5 cm of sediment, which is in fairly good agreement with the status based on lake type specific PMA data ('good' status, Table 1). Other paleoecological studies of currently eutrophic lakes (e.g. Brodersen et al. 2001; Meriläinen et al. 2000; Kansanen 1986) have shown substantial changes in lake chironomid communities due to anthropogenic eutrophication. In Lake Kirmanjärvi the significant reduction of chironomid diversity in the modern sediments was unparalleled in the BQI values. Meriläinen and Hamina (1993) noted a similar pattern for Lake Puruvesi resulting from indicator taxa substitution (see also Brodersen and Quinlan 2006). In our case, species substitution does not influence the results and given the low pre-disturbance BQI for Lake Kirmanjärvi we suspect that only very severe and continuous hypoxia could have resulted in a noticeable reduction in the modern BQI. Lake morphology can be a major determinant of lake trophic status (Vollenweider 1975) and in a recent study of Finnish lakes, Jyväsjärvi et al. (2009) revealed a distinct link between lake morphometry, in particular depth, and the BQI. Using the morphometric relationship of Jyväsjärvi et al. (2009) slightly overestimated the actual pre-disturbance BQI of Lake Kirmanjärvi (modelBQI = 2 vs. paleoBQI = 1.43). The paleolimnological chironomid data therefore support findings for the phytoplankton community and indicates that Kirmanjärvi is indeed a naturally meso-eutrophic lake.

Classification of ecological status

The ability to set adequate objectives for lake restoration and management is intimately linked to the correct assignment of the correct lake type. The paleolimnological approach taken in

this study revealed Lake Kirmanjärvi's naturally high productivity. Naturally eutrophic lakes have been identified in other paleolimnological studies (e.g. Räsänen et al. 2006a) and these lakes form a unique lake type in the Finnish typology system (Vuori et al. 2006). Using only the standard typology criteria, Lake Kirmanjärvi would have been erroneously considered a small humic lake. In the present case, individual biological quality element-assigned class boundaries differed between the two types only for the fish community which classified the lake as either "moderate" or "good" for the small humic or naturally eutrophic lake type assumption, respectively. Differences in the status assessment for this BQE did not change the overall "moderate" ecological status. This is because contrary to the One-out, All-out (OoAo) principle (European Commission 2005), the Finnish classification scheme (Vuori et al. 2009) utilizes the median score across all BQEs to determine the overall lake ecological status. The outcomes and benefits of these two differing approaches have been discussed in detail by Alahuhta et al. (2009b) and Rask et al. (2010). In Lake Kirmanjärvi, all biological quality elements largely agreed on the ecological status so that the nature of the classification approach (OoAo vs. median) would not have impacted the overall outcome.

In addition to producing comparable overall status class, site-specific (paleolimnological) status assessment mirrored the type-specific ecological status classification at the level of individual BQEs, with the site-specific diatom-PMA and other algal data suggesting moderate status while chironomid-PMA and cyanobacterial akinete data mostly indicated high status. The type-specific assessment method resulted in the conclusion that changes in ecological status had occurred for several BQEs and management efforts are needed to bring the lake back to good ecological status. While this overall assessment was in agreement with the paleoecological classification, the monitoring data approach did not indicate the marked changes in the subfossil chironomid or phytoplankton community observed in the paleobiological record. Even though the metrics and BQEs used to assess a lake's deviation from its type-specific and site-specific historical reference states are slightly different, our results show that paleobiological data provide a viable – and in our case more informative – alternative to the typology-based method of assessing lakes status. The paleolimnological approach employed is especially useful in validating the deterioration of ecological status in cases where there are few pristine sites left to allow monitoring-based type-specific reference conditions to be set.

Implications for lake management

Coinciding with lake management efforts, monitoring detected clear changes in the biomass of phytoplankton, indicated by concentration of chlorophyll-*a*, and epilimnetic phosphorous concentration. Between 1987 and 1997 a total of 200 t of fish, corresponding to an average annual fish catch of 22 t (71 kg/ha/a) were removed from the lake (Kauppinen 2006). The fishing effort was targeted to cyprinids but the species composition of the catch has not been reported. Effects of drastically reduced fish predation on zooplankton with a cascading effect on the phytoplankton are well documented (Mittelbach and Chesson 1987). The current fish community of Lake Kirmanjärvi is dominated by cyprinids, suggesting their substantial role in the functioning of the lake ecosystem. The intensive fishing effort mainly targeted cyprinids which may have reduced internal loading through the reduction of feeding activity-related bioturbation. Unfortunately, we lack monitoring or proxy data on zooplankton and fish to assess the direct or indirect effects of biomanipulation. However the declining chlorophyll-*a* and phosphorous concentrations during intensive lake management efforts and their subsequent increase after the termination of the intensive fishing, suggest that the intensive fishing effort

was effective. Hypolimnetic oxygenation of Lake Kirmanjärvi may have enhanced the effects of biomanipulation by decreasing the release of nutrients during hypoxic conditions (Kauppinen 2006). The increased sediment phosphorus concentrations in the recent sediments support this conclusion. Further management efforts are necessary to restore Lake Kirmanjärvi from its present "moderate" to "good" ecological status. Given the good present day status of the benthic community we believe the objective of attaining "good" ecological status of Lake Kirmanjärvi is possible, although it will require intensive in-lake and catchment restoration measures and follow up monitoring.

Conclusions

- The paleoecological proxies suggested that Lake Kirmanjärvi has been eutrophic with cyanobacterial blooms for several centuries and should be classified to the Naturally Eutrophic lake type instead of the Small Humic lake type. However, this shift did not affect the final ecological status assessment for the lake.
- In this study, ecological status assessment based on site-specific paleoecological proxy data largely agreed with the monitoring-based type-specific status assessment, even at the level of individual BQIs. However, since this may not be the case for every lake, paleoecological data can provide a more detailed picture of past ecological changes for WFD-based ecological assessment.
- While the diatom-based lake status declined to "moderate" already in the mid-1800s, lake status deteriorated most drastically for all proxies during the mid-1960s with the onset of intensive, fertilizer-based, agriculture.
- The effects of lake management actions were observed both in the monitoring data and in the paleolimnological record. The results suggest that further in-lake and catchment management efforts and monitoring are required to reach the goal of "good" ecological status.

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VI

COMPARING LONG-TERM SEDIMENT RECORDS TO CURRENT BIOLOGICAL QUALITY ELEMENT DATA - IMPLICATIONS FOR BIOASSESSMENT AND MANAGEMENT OF A EUTROPHIC LAKE

by

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Comparing long term sediment records to current biological quality element data – Implications for bioassessment and management of a eutrophic lake

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ABSTRACT

Defining reference conditions for lakes situated in areas of human settlement and agriculture is rarely straightforward, and is especially difficult within easily eroding and nutrient rich watersheds. We used diatoms, cyanobacterial akinetes, remains of green algae and chironomid head capsules from sediment samples of Lake Kirmanjärvi, Finland, to assess its deviation from the initial ecological status. These site-specific records of change were compared to current type-specific ecological status assessment. All paleolimnological data indicated deviation from natural conditions and mirrored the current, monitoring-based assessment of “moderate” ecological lake status. However, the sediment data showed that the lake should be re-typified as a naturally eutrophic lake. Sediment records as well as current monitoring data indicated temporary improvement in water quality in response to extensive fish manipulation. Our results suggest that paleolimnological records can be used to derive site-specific reference conditions and that extensive restoration efforts can result in gradual, observable improvements of water quality and ecological status.

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Introduction

Decision making in lake management is based on the interplay between current policy, the availability of resources and the environmental conditions at the site. One of the most challenging tasks for lake management and restoration is the selection of the target level, i.e. the desired level of attainable water quality. Target levels are often normatively specified in environmental legislation (e.g. U.S. Clean Water Act, EU Water Framework Directive, WFD) and set close to the estimated pre-disturbance conditions. In the European Union, the WFD (European Commission 2000) defines the management goal of ‘good status’ as a minor deviance of selected biological and abiotic quality elements from their respective reference states. In the ecological classification of lakes the assessment of this deviance is based on four biological quality elements (BQEs): phytoplankton, aquatic macrophytes and phytobenthos, benthic macroinvertebrates, and fish.

While intuitively simple, defining and quantifying pre-disturbance reference conditions in lakes is demanding because anthropogenic impacts often predate direct observations on the BQEs. Reference conditions can be based on data from lake type-specific pristine monitoring sites, predictive modelling or temporal changes i.e. historical changes or paleoreconstruction (European Commission 2003). Recent advances in studies of sediment records have promoted the use of paleolimnological methods to obtain information on past lake characteristics (Leira et al. 2006; Räsänen et al. 2006a). These quantitative paleoecological inference methods allow reconstructions of past lake nutrient levels, pH, conductivity, etc. and enable comparisons with direct observations. Current paleoecological inference models often use phytoplankton or chironomids as proxies. When compared to water samples, the sediment record of phytoplankton communities is often incomplete due to the poor preservation of certain types of algae. However, many important taxa deposit resistant, identifiable remains (e.g. Bacillariophyceae) which are sometimes identifiable to species level. Diatom assemblages are functionally important in many aquatic ecosystems and sensitive to environmental changes (e.g. Stoermer and Smol 1999). Similarly, cyanobacteria and green algae

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are important for practical lake management since their blooms often mirror eutrophication and indicate drastic changes in the overall composition of the phytoplankton community. While vegetative cyanobacterial cells do not preserve in lake sediments, certain cyanobacteria produce resting spores, i.e. akinetes, that are resistant to decomposition and have successfully been applied as a paleolimnological proxy (Cronberg 1986; van Geel et al. 1994; Kling 1998; Findlay et al. 1998; van Geel 2001; Eilers et al. 2004; Bradbury et al. 2004; Räsänen et al. 2006b). In Finnish freshwater systems *Anabaena* spp. is the most common blooming cyanobacteria with well identifiable akinetes (Lepistö 1999) and *Scenedesmus* spp. and *Pediastrum* spp. are the paleolimnologically most extensively used green algae. A combined study of fossil diatoms, cyanobacteria, and green algae provides a fairly comprehensive picture of past phytoplankton assemblages.

Changes in lakes are not only reflected in phytoplankton but affect higher trophic levels such as chironomids as well. Subfossil head capsules of chironomids are present in all except highly anoxic sediments (e.g. Meriläinen et al. 2001). The composition of the profundal chironomid community is widely used to assess lake trophic status and several paleolimnological chironomid inference models have been constructed (e.g. Brodersen and Lindegaard 1999). In this study we relate past changes in the phytoplankton community to concurrent sediment data for the chironomid community.

The Iisalmi watercourse in Eastern Finland poses a unique challenge to the determination of lake management goals. Located in a geographic area where surface waters generally attain “good” or “excellent” status, these lakes are unexpectedly eutrophic and display phytoplankton blooms and extensive growth of macrophytes. The eutrophic state of these lakes may be related to watershed geology which is characterized by shallow basins and high proportions of fine-grained tills and fine grained sorted deposits. However, since these fertile soils rich in fines also attracted early human settlement and agriculture the main cause of the observed eutrophic state is difficult to determine.

The WFD-compliant ecological classification of Finnish surface waters is based on type-specific reference conditions derived from monitoring data of pristine reference sites for each surface water type (Vuori et al. 2009). The Finnish lake typology differentiates between 14 lake types mainly based on size, humic content, depth and natural eutrophy (Vuori et al. 2006). The exceptionally high amount of fine grained glacial and post glacial sediments in the watershed of the Iisalmi watercourse complicates the exact designation of individual lakes into lake types and, hence, the application of the correct type-specific reference conditions for ecological classification. Closely matching pristine reference lakes also are difficult to find for the region. We therefore used paleolimnological methods to assess the overall changes in the phytoplankton and chironomid community, identify algal blooms, and to infer the past nutrient status within one representative lake from the Iisalmi watercourse. By relating paleolimnological to contemporary monitoring data we further sought to (1) quantify the community changes caused by recent human activities, (2) assess the efficacy of past lake management actions and (3) evaluate the utility of sediment-based data in comparison to current data for reference condition-based ecological classification.

Methods

Study site

Lake Kirmanjärvi, located in Eastern Central Finland in the Iisalmi watercourse (Fig. 1), consists of two basins, the smaller Pieni-Kirma (31 ha, maximum depth 7 m) and the larger Kirmanjärvi (280 ha, maximum depth 9.7 m). The Finnish lake typology

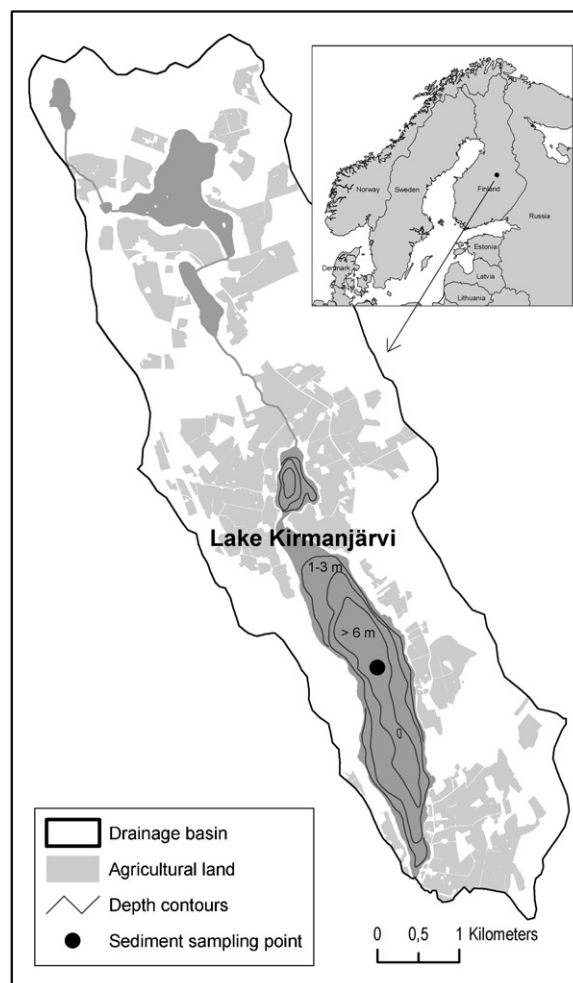


Fig. 1. A map of Lake Kirmanjärvi and its location in Fennoscandia.

system (Vuori et al. 2006) categorizes small lakes under 500 ha based on their humic content (measured by water colour), and mean depth. Naturally eutrophic lakes are identified based on their high epilimnetic turbidity (FNU > 5), high alkalinity (>0.4 mmol/l) or other information on natural eutrophy. Using the typology criteria of surface area, mean depth and water colour (area < 500 ha, mean depth > 3 m, water colour 30–90 mg Pt/l) Lake Kirmanjärvi is classified as a small humic lake in the Finnish system. Even though its geographic location suggests possible naturally high productivity Lake Kirmanjärvi does not meet the formal typology criteria for naturally eutrophic lakes. In addition to being representative for the Iisalmi watercourse the lake has regional importance because it serves as a reservoir for the production of drinking water and preserving good water quality (e.g. without toxic cyanobacterial blooms) attracts high public interest. Furthermore, the chemical water quality of the lake has been monitored intensively since 1986 and the lake has undergone different phases of restoration efforts, including intensive fishing period during 1987–1997 and hypolimnetic oxygenation since 1986. Further management efforts of the lake are currently being considered.

Coring, sampling and sediment dating

Sediment assemblages predating anthropogenic impacts were analyzed for algal remains and chironomid subfossils extracted from a 150 cm long composite sediment core sequence. Coring was performed from the lake ice on March 27th, 2006. We selected a

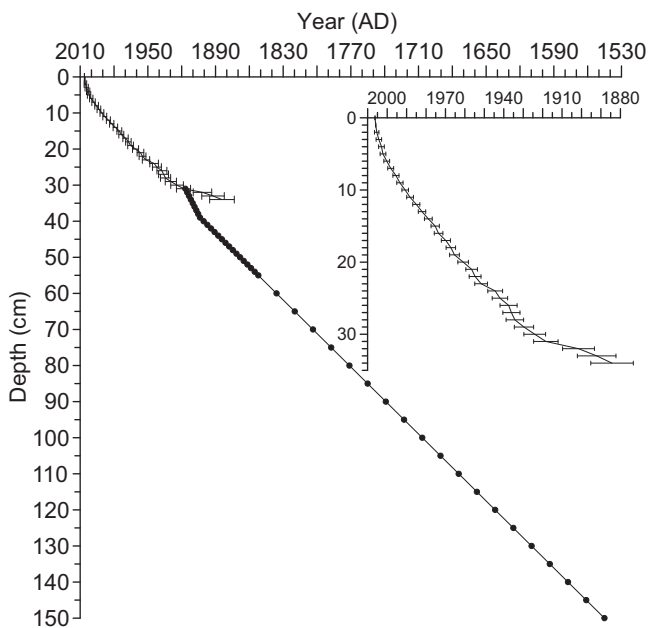


Fig. 2. Age-depth model for the Lake Kirmanjärvi core based on ^{210}Pb dating results and the CRS model (lines with error bars). The inset shows the results for the upper 34 cm in more detail. The dotted line shows an extension of the dating results based on an estimated 2-fold accumulation rate between 39 and 31 cm and average accumulation rate below 39 cm.

coring site from the middle part of the Kirmanjärvi basin (water depth 7.2 m, Fig. 1) because of the installment of a hypolimnetic oxygenator into the deepest part of the basin in 1986.

Sediments were first cored with a Limnos gravity corer which preserves the sediment-water interface (Kansanen et al. 1991). The 42 cm long Limnos core was sliced continuously into 1 cm thick slices that were stored in plastic bags in the field. A long continuous Kullenberg type PP core (712 cm, top 150 cm used here; Putkinen and Saarelainen 1998) was taken from the same area. The two cores were later correlated based on their loss on ignition and ^{137}Cs profiles. The PP core tubes and the Limnos core slices were stored in a cold room at $+4^\circ\text{C}$ until subsampled for analyses.

The Limnos core was dated using artificial ^{137}Cs and naturally occurring ^{210}Pb radionuclides (Robbins 1978; Appleby and Oldfield 1992; Crusius and Anderson 1995; Appleby 1998; Mattila et al. 2006). The activities of radionuclides were determined by gamma-ray spectrometry (Klemola et al. 1996). The major source of the ^{137}Cs activities in the study area has been the fallout from the Chernobyl accident in 1986 and the highest measured activity concentration of ^{137}Cs in the sediment profile was used as a time marker for that year. Sediment cores were dated with the CRS (Constant Rate of Supply) model (Appleby and Oldfield 1978), but the simple “constant flux: constant sedimentation rate” model was also used to compare dating results. The final age model was constructed by extending the radiometrically obtained average sediment accumulation rate to the lower part of the core, with the exception of the 30–40 cm mineral rich section for which twice the average rate was used as a rough estimate in the absence of measured data (see Fig. 2).

Sediment chemistry

Analyses of sediment chemistry were employed to provide information on changes in sediment quality (due to e.g. erosion, organic inputs, resuspension) and nutrient or metal inputs. Sediment chemistry was analyzed from 85 samples down to 150 cm in sediment. Samples were digested with HNO_3 in a microwave

oven and the resulting extracts analyzed with the ICP-OES technique, a modification of the US EPA method 3051 (US EPA 2007). The digestion does not dissolve the major silicates but breaks down trioctahedral micas, clay minerals, carbonates (excluding magnesite), titanite, oxyhydroxide precipitates, etc. and liberates elements from the organic matter of lake sediments. Besides the determination of the ‘contaminated fraction’ that the extraction was originally developed for, the leach allows determination of total P concentrations and captures variation in the fine-grained mineral fraction as well as hydroxide precipitates and sulfide minerals.

Paleobiological analyses

Diatoms were analyzed from 27 samples of the sediment column with increasing sample spacing down-core (1 cm, 2 cm, 5 cm, and after that at 15 cm spacings, see Figs. 5 and 6). Diatom slides were prepared according to Battarbee et al. (2001) and analyzed with a Leitz Diaplan light microscope at $1250\times$ final magnification using both phase contrast and bright field illumination. Identification of 300–350 valves/slide were performed to species level when possible following Krammer and Lange-Bertalot (1986, 1988, 1991a, 1991b).

A total of 85 samples were analyzed for cyanobacterial akinetes and fossil green algae. These remains were analyzed from the 42 cm Limnos core at 1 cm intervals and the 150 cm PP core at 2.5 cm intervals beginning from 45 cm. First, 0.5 ml of fresh sediment was weighted and diluted to 50 ml using distilled water. Well-mixed subsamples of 0.25 ml were taken for microscopy from this diluted sample. The akinetes and green algae remains were counted with an inverse microscope using phase-contrast illumination at $800\times$ magnification applying Utermöhl’s technique (Utermöhl 1958). Akinetes and green algae were counted from randomly selected fields, and the counting was stopped when 1000 fields were investigated using a technique modified from Kling (1998). The results were given as number of remains per gram of dry sediment.

Altogether 27 subsamples from the sediment core were taken for chironomid analysis with increasing sample spacing down core. Chironomid samples were deflocculated in 10% KOH solution at room temperature for 12 h, sieved ($100\ \mu\text{m}$), preserved in 70% ethanol and picked under a stereomicroscope. The numbers of chironomid head capsules per subsample ranged from 21 to 70. The 1136 head capsules belonging to 27 chironomid taxa were identified mainly to species or genus level mainly according to Wiederholm (1983).

Current biological, physical and chemical monitoring data

Biological data used for the present ecological classification of Lake Kirmanjärvi was sampled in 2005–2006. In October 2005, six replicate Ekman grab samples were taken outside the effective range of the hypolimnetic oxygenation device. Ekman grabs were sampled from depths ranging between 7.6 and 8.0 meters. The benthic fauna was identified according to Wiederholm (1983).

Aquatic macrophytes were surveyed in the summer of 2006 from transects at 15 sites using a method described in detail by Leka et al. (2003). A vegetation abundance metric index (Ilmavirta and Toivonen 1986) was calculated from these transects using the taxonomical resolution used in Kuoppala et al. (2008).

Fish were sampled with Nordic gillnets in June–September 2006 using a stratified random sampling protocol described in detail by Olin et al. (2004). The total fishing effort was 15 gillnet nights (Kauppinen 2006).

Physical and chemical water quality parameters of Lake Kirmanjärvi have been monitored during the productive period since 1986. On 3–4 occasions annually, basic parameters e.g. total and

inorganic nutrients, conductivity, oxygen and COD were measured. In addition, surface water chlorophyll-*a* was measured during all summertime samplings. Phytoplankton species biomass estimates, calculated by the Utermöhl's technique (Utermöhl 1958), were available from 2001 to 2006 from the phytoplankton database of the Finnish Environment Institute.

Numerical methods

Results of the diatom analyses were visualized using ordination plots created with the CANOCO for Windows 4.5 software package (ter Braak and Šmilauer 2002) and stratigraphic diagrams created with C2 (Juggins 2003). Because an initial detrended correspondence analysis (DCA; Hill and Gauch 1980) run showed that the length of the species gradient was less than 2 SD units, the linear response-based method of principal components analysis (PCA) was used. Stratigraphically constrained cluster analysis was used to help divide the sediment diatom assemblages into local assemblage zones. Cluster analysis was run using the paired group method on Euclidean and chord distance dissimilarity measures (Past ver. 1.99; Hammer et al. 2001). To describe the deviation of the diatom phytoplankton community from the site specific reference conditions, a local diatom Percent Model Affinity curve (PMA; Novak and Bode 1992) was generated using average abundance data from the lowermost diatom zone (150–60 cm) as the model community from which percentage difference was calculated for each sample ($100 - 0.5 * \sum | \text{taxon's relative abundance in model community} - \text{taxon's relative abundance in sample} |$).

The past diatom-inferred epilimnetic total phosphorus (DI-TP) concentration was reconstructed using the calibration set and inference model presented by Kauppila et al. (2002). The model was generated with simple weighted averaging regression (ter Braak and van Dam 1989) and classical deshrinking. Lake Kirmanjärvi is located within the geographical area covered by the calibration set and the lake fulfils the selection criteria of the calibration set lakes (size, pH and water colour ranges).

The Shannon–Wiener diversity index (*H*) for the chironomid data used untransformed counts and the Benthic Quality Index (BQI) was calculated according to Wiederholm (1980). To assess whether the diatom zonation was reflected in the structure of the chironomid community as well we ran the non-parametric analysis of similarity (ANOSIM, Clarke 1993) on log-transformed abundance data. In addition, we tested whether the onset of intensive farming in the 1960s was reflected in the composition of the chironomid fauna. ANOSIM tests for differences between groups in multivariate data sets and is thus vaguely analogous to analysis of variance in univariate statistics. ANOSIM computes *R*, which indicates the magnitude of difference between groups. An *R* value of 1 indicates that the communities completely differ between defined groups, whereas an *R* of 0 indicates no difference. Bray–Curtis distance measures were used in ANOSIM and Bonferroni corrected *P*-values were used to assess the statistical significance of *R*. To evaluate the contribution of individual species to the dissimilarity between predefined groups we used SIMPER (Clarke 1993).

Ecological Quality Ratios (EQRs) were derived for each metric in the current monitoring data by comparing the observed (*O*) value of each metric to the expected (*E*) values from reference conditions according to the Finnish national ecological classification system (Vuori et al. 2009). The EQRs were calculated employing the reference values for both “small humic” and “naturally eutrophic” lake types. The resulting EQRs were divided into five classes: high, good, moderate, poor and bad (class boundaries 0.8, 0.6, 0.4, and 0.2). The classification system integrates multiple metrics and different biological quality elements (BQE) by scoring the individual metrics and the overall lake status class is given as a median score across different BQEs. A more thorough presentation of the Finnish eco-

logical classification system, with examples of application, is given by Alahuhta et al. (2009a).

Results

Sediment characterization and dating

The sediment profile (150 cm) consisted of homogenous grayish brown silty gyttja up to the depth of 50 cm. The mineral content of the sediment gradually increased to the depth of 35 cm where we observed a shift to higher organic content and from 31 cm upwards the sediment turned black due to sulfide coloring. Another increase in organic content was seen at 18 cm and patches of oxidized iron were observed at sediment depths between 26 and 12 cm. A thin veneer of oxidized iron was also observed at the sediment water interface in the gravity core. Fig. 2 shows the radiometric dating results and the estimated age-depth model for the core, indicating that the core section extends to mid sixteenth century. Both the Cs-137 and Pb-210 determinations were made from samples between 0 and 43 cm and a clear Chernobyl fallout peak was detected at the 12–15 cm slice. Pb-210 provided dates down to 36 cm below which concentrations declined to very low levels.

Sediment chemistry

To obtain information on external forcing events such as changes in sediment quality and possible metal or nutrient loading, changes in sediment chemistry are presented for selected elements in Fig. 3. Concentrations of the major redox-sensitive elements Fe and Mn, as well as phosphorus, which is often bound to Fe–Mn precipitates, were high in the lowest part of the sediment column, between 150 and 100 cm. In the 100–55 cm sediment section, concentrations of Fe, Mn and P were lower despite a peak in mineral matter content at the depth of 85 cm and the subsequent increase in indicator elements for fine grained mineral matter such as Al, K and V.

A second period of increasing mineral matter content was found from 55 cm upwards with a peak between 35 and 31 cm. Iron concentrations were high in these samples, while no concurrent peaks in Mn and P were present. In the topmost part of the sediment column the share of mineral matter declined (above 31 cm), whereas concentrations of redox and oxyhydroxide-related elements gradually increased. Manganese, which is more susceptible to reduction and mobilization in sediments than iron, peaked at 2–3 cm depth. Superimposed on this trend was an increase in elements often associated with fossil fuels and atmospheric deposition: Pb, S, Ni and V. The trends for Ni and V became evident only after the data was normalized using the K content to account for the fine grained minerals that typically also contain both Ni and V. Sulfur, a major constituent in poorly crystalline metal monosulfides of many sediments, peaked along with iron concentrations at 14–15 cm depth. High concentrations of elements indicative of air emissions were found in sediments deposited in the early 1980s. Subsequently, their fallout has reduced considerably because of reduced emissions from fossil fuel-based power generation and the ban of lead in gasoline.

Diatoms

Results of the diatom analyses are presented in Fig. 4 and summarized as a PCA plot in Fig. 5. The diatom stratigraphy was divided into three distinct local diatom assemblage zones (LDZ) with the aid of cluster analysis results. The lowermost diatom zone (LDZ 3) extended from 150 to 60 cm in the sediment core (~AD 1545–1836) and mirrored observed changes in sediment chemistry (150–100 cm, 100–55 cm). LDZ 3 was characterized

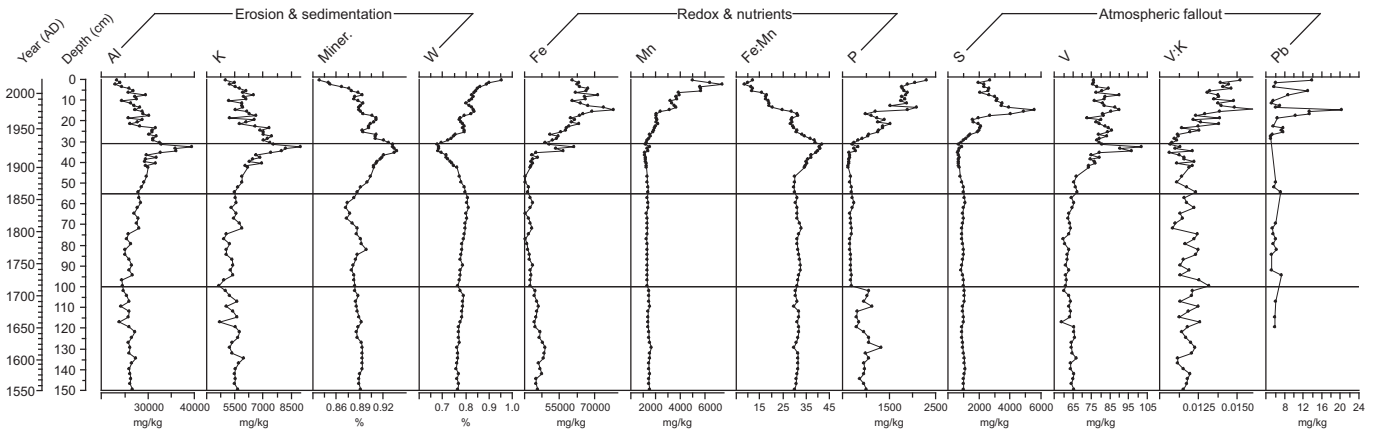


Fig. 3. Distribution of selected elements and element ratios in the core.

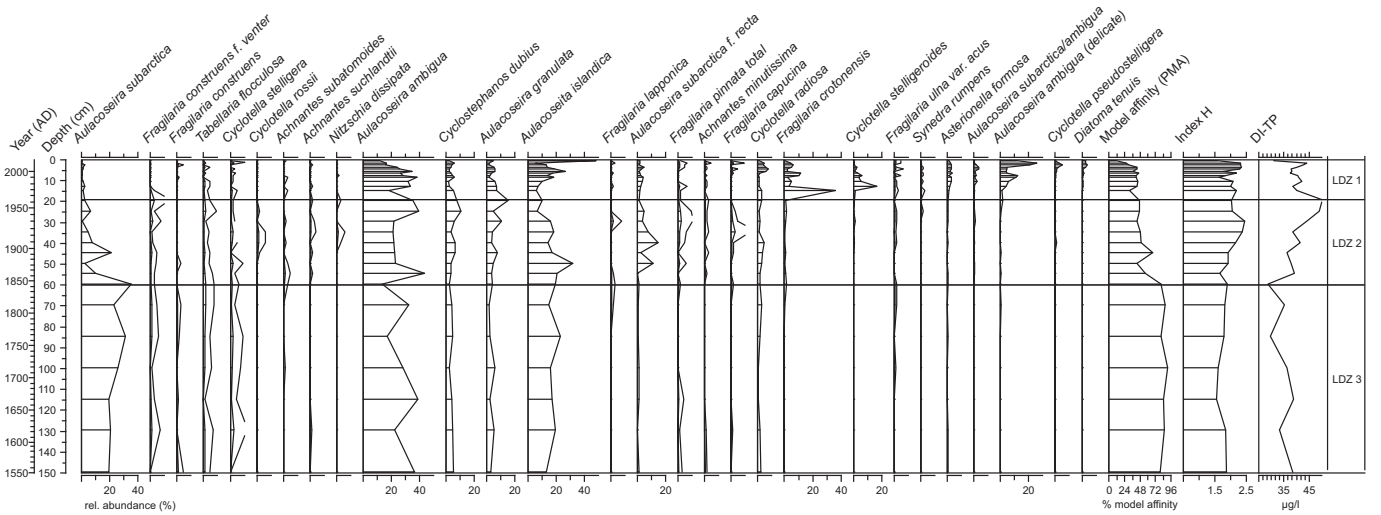


Fig. 4. Relative abundances of the most common diatom taxa in the Kirmanjärvi core. The species have been arranged according to their abundance weighted average depth of occurrence. Exaggeration line multiplier is 5. Percent model affinity, Shannon Index H, diatom-inferred lake water [TP] and Local diatom assemblage zones (LDZ) are also shown.

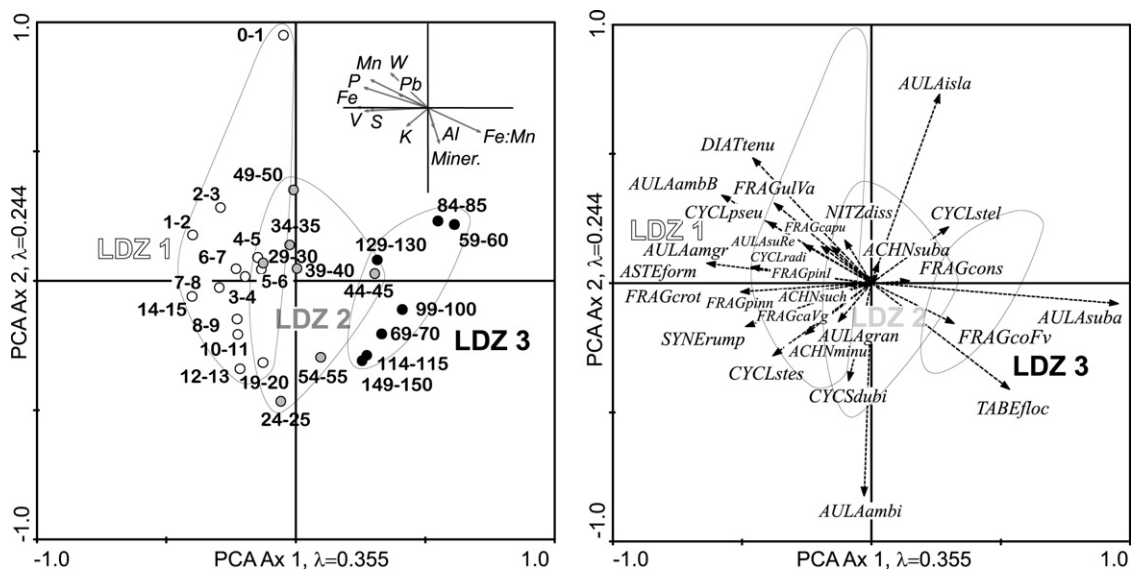


Fig. 5. A PCA plot of the diatom samples (left panel) and species (right panel). The diatom assemblage zones are indicated with symbol shading (black = LDZ 3, white = LDZ 2, medium grey = LDZ 1). The inset shows major directions in selected sediment variables superimposed on the diatom ordination.

by high abundances of species that prefer mesotrophic waters such as *Aulacoseira subarctica* (~25%) and *A. islandica* (~18%). In addition, several taxa characteristic of eutrophic waters occurred in high numbers: *A. ambigua* (~30%), *A. granulata* (~5%), and *Cyclostephanos dubius* (~5%). The diatom-inferred TP concentration was in the eutrophic range for this zone (29–38 µg/l) and Shannon's Index H values ranged between 1.55 and 1.90. In general, species assemblages were fairly stable in LDZ 3, despite the decrease in sediment-bound P at 100 cm (~AD 1710) and a short-lived peak in mineral matter content at 85 cm (~AD 1755). The average PMA value for samples in LDZ 3 was 84%, which is the site-specific (local) reference PMA used in this study.

The delineation of the next diatom zone (LDZ 2, 55–19 cm, ~AD 1850–1965) coincides with the incipient increase in sediment mineral matter content. In this zone, the relative abundance of *A. subarctica* declined and a less thinly silicified form of the species increased. This delicate form of *A. subarctica* is often encountered in cases of eutrophication (e.g. Kauppila et al. 2002). *A. islandica*, another mesotrophic species, declined as well after an initial peak at 50 cm. With the decrease in the two abundant mesotrophic forms, species typical of eutrophic waters such as *A. ambigua*, *A. granulata* and *C. dubius* increased their share. At the same time, the proportions of certain non-planktonic *Achnanthes* and *Fragilaria* species increased. The diatom-inferred TP concentration increased to a maximum of 50 µg/l at 19–20 cm sediment depth. Even though the initial shift in species compositions was abrupt, the PCA plot indicates that the overall change in assemblages was moderate (Fig. 5). The PMA values for LDZ 2 ranged between 44 and 68% (i.e. 52–81% of the background PMA) and with the addition of taxa to the assemblages the values of Shannon's H increased to a maximum of 2.44 at 30 cm depth. In the ordination plot, ranges of LDZ 3 and 2 partly overlap although the first PCA axis separates the diatom zones fairly well.

The transition to LDZ 1 (19 cm – top, ~AD 1965–) coincided with a moderate peak in sediment mineral matter content. Similar to the LDZ 3–LDZ 2 transition, samples of LDZ 2 and LDZ 1 overlap somewhat in the ordination plot (Fig. 5). In the uppermost samples from LDZ 2, *A. ambigua*, *A. granulata*, and *C. dubius* increased in abundance while *A. islandica* continued to decline and this shift to more eutrophic species compositions continued in LDZ 1. Diatom-inferred TP subsequently declined from the very high values observed at the LDZ 2 to LDZ 1 transition but remained at a level of 40 µg/l. The division between LDZ 2 and LDZ 1 was the most distinctive separation in all of the cluster analysis runs and several 'new' species increased their proportion in the assemblages: *Fragilaria crotonensis*, *Cyclotella stelligeroides*, *Synedra rumpens*, *Asterionella formosa*, and a small form of *A. ambigua*. Concomitant with the increases in these eutrophic planktonic species was the decline in species such as *A. subarctica* and *Fragilaria construens* f. *venter*, and many of the *Achnanthes* and *Fragilaria* taxa that increased in LDZ 2. The species composition in the most recent samples revealed increases in *Fragilaria capucina* var. *gracilis*, *Cyclotella pseudostelligera* and *Diatoma tenuis*. Species shifts were reflected as decreases in the PMA curve, especially in the uppermost three samples (average PMA for the zone 39%, i.e. 47% relative to background PMA). In contrast, the Shannon diversity showed no corresponding declines and attained high values ($H = 2.30$ – 2.35) near the sediment surface.

Cyanobacterial akinetes and green algae

Results of the cyanobacterial akinete, *Pediastrum* spp. and *Scenedesmus* spp. analysis are presented in Fig. 6. Cyanobacteria akinetes were present in every sample studied. The majority of akinetes belonged to the genus *Anabaena* but also akinetes typical of less nutrient-rich waters such as the genus *Aphanizomenon* were

found. *Anabaena* spp. akinetes were most abundant at sediment depths from 90 to 50 cm (~AD 1740–1870). Towards the sediment surface their amount decreased. By contrast, *Aphanizomenon* spp. akinetes were present in large amounts at the bottom of the sediment core, decreased in abundance between sediment depths of 130 and 30 cm, whereafter they gradually became abundant again.

A large variation in the number of green algae remains was observed, as some samples contained no subfossils, while large amounts of remains were present in adjacent samples. Both *Pediastrum* spp. and *Scenedesmus* spp. remains displayed a slight increase in their abundance at sediment depths from 80 to 60 cm after which their amount first decreased, to increase again from 20 cm towards the sediment top.

Chironomid subfossils

An overview of the chironomid species composition along the sediment profile is presented in Fig. 7. The most abundant taxa throughout the sediment record were *Procladius* spp. and *Chironomus plumosus*, both indicative of eutrophic conditions, along with *Tanytarsus* spp., a genus containing mainly littoral species. Overall, the observed fauna was mainly littoral and indicative of meso-eutrophic conditions. Chironomid species indicative of oligotrophic conditions were lacking and indicators of mesotrophy (*Sergentia coracina*, *Stictoichonimus rosenscholdii*) were scarce throughout the sediment profile.

The chironomid fauna did not follow the compositional changes of the local diatom assemblages, i.e. there were no significant differences between any of the LDZ-based sections (ANOSIM, $R < 0.22$, $P > 0.089$). Overall, SIMPER found the dissimilarity of the chironomid species composition to range between 38 and 40% between the diatom zones (LDZs). The most notable difference between the modern intensive farming period (0–19 cm, ~AD 1965– present) and the preindustrial (>55 cm; ~AD 1850–1545) era was the marked reduction in *Chironomus thummi* type (contribution to observed dissimilarity 6%) mirrored by the concurrent increase in *Chironomus plumosus* (contribution to dissimilarity 4%). Contrary to the results for the LDZ division, we noticed a distinct difference in chironomid communities only when comparing the modern intensive farming period (0–19 cm, ~AD 1965–) to the rest of the sediment column (ANOSIM, $R = 0.21$, $P < 0.0005$). SIMPER identified differences in densities of *Chironomus plumosus*, *Procladius* spp., *C. thummi* type, *Glyptotendipes* and *Tanytarsus* spp. to account for roughly half of the total observed dissimilarity of 41%.

Temporal changes in the number of chironomid head capsules relative to the sediment dry weight (g^{-1}) mirrored the diatom based zonation (Fig. 7). The BQI, however, did not differ between the LDZ (one way ANOVA, $F_{2,25} < 0.2$, $P > 0.81$), while Shannon–Wiener diversity and the total number of chironomid species differed significantly between the diatom zones (one way ANOVA $F_{2,24} > 7.7$, $P < 0.003$). We observed significantly fewer species in the LDZ 1 (mean = 7.2) compared to the LDZ 2 (mean = 11.5) and Shannon Wiener diversity was lower when comparing LDZ 1 to LDZ 2 or LDZ 3 (Post hoc Bonferroni $P < 0.004$; Fig. 7). A separate comparison of the modern era species richness and Shannon Wiener diversity to the rest of the sediment column also showed significant differences (one way Anova, $F_{1,25} > 14.8$, $P < 0.001$). The local chironomid model affinity remained high, above 80% of the local reference PMA of 76%, until the uppermost 5 cm, where it declined to 65–50% of the reference in three samples.

Ecological status according to current monitoring data

The modern biota of Lake Kirmanjärvi was dominated by taxa preferring meso-eutrophic conditions in all the taxonomic groups studied. Cyanobacteria were abundant in the pelagic phytoplank-

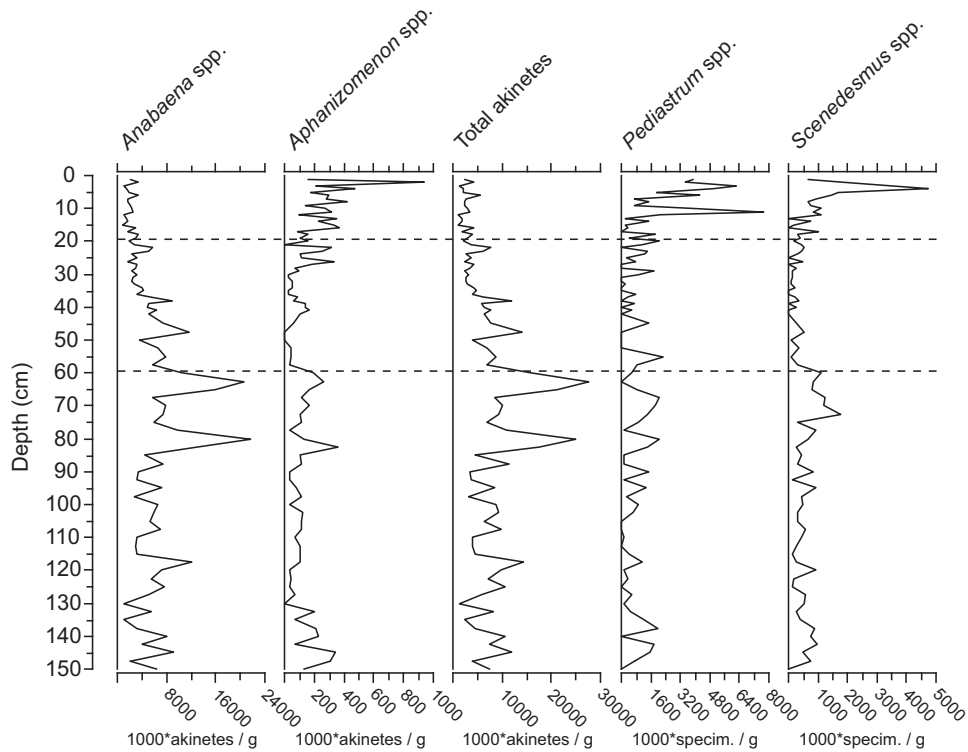


Fig. 6. Distribution of *Anabaena* and *Aphanizomenon* akinetes and chlorophyte remains in the Lake Kirmanjärvi core. Dashed lines show diatom zone boundaries for comparison.

ton community and the overall phytoplankton biomass was high (5.1 mg/l). A relatively high number of macrophyte species (total 48 taxa) were recorded, of which 37% (18 taxa) are considered indicators of meso-eutrophy or eutrophy (Leka et al. 2008). Chironomid larvae dominated the benthos (42% of individuals) with *Procladius* sp. and *Chironomus anthracinus* being the two most abundant taxa. The second most abundant benthic group were the oligochaetes (40%), which are indicative of low oxygen conditions. The cyprinid fish like roach (*Rutilus rutilus*) and bream (*Abramis brama*) accounted for 51% of biomass per unit effort (BPUE). Per-

cids like perch (*Perca fluviatilis*) and ruffe (*Gymnocephalus cernuus*) made up 37% of the total BPUE.

Lake Kirmanjärvi was classified to be in a “moderate” ecological state using type specific reference conditions for either the small humic lake type or the naturally eutrophic lake type (Table 1). The choice of typology influenced the resulting overall lake EQR only minutely. Designating the lake to the small humic lake type produced a slightly lower EQR than designation to the naturally eutrophic lakes. This difference was small and resulted from type specific results for the fish community. The benthic macroinver-

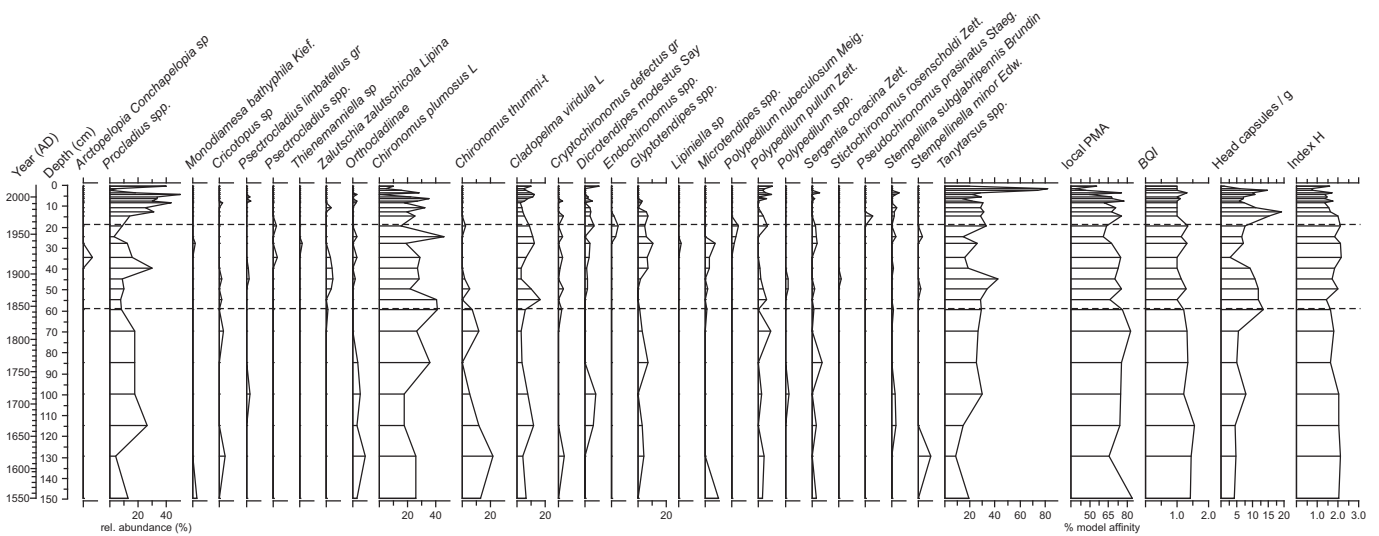


Fig. 7. Relative abundances of the most common chironomid taxa in the Kirmanjärvi core. Percent Model Affinity (site-specific), Benthic Quality Index, concentration of head capsules and Shannon Index H are also shown. Dashed lines show diatom zone boundaries for comparison.

Table 1
Lake Kirmanjärvi ecological status according to the Finnish ecological classification system, given separately for Small Humic and Naturally Eutrophic lake types. H = High, G = Good, M = Moderate, P = Poor. *Observed values differ for two of the macrophyte metrics depending on lake type assignment. N/BPUE = number/biomass per unit fishing effort.

Classification metrics			Ecological classification using type-specific reference conditions of small humic lakes			Ecological classification using type-specific reference conditions of naturally eutrophic lakes			
BQE	Metric (unit)	Observed value	Metric EQR score/status class	BQE median EQR score/status class	Lake status EQR score/class	Metric EQR score/status class	BQE median EQR score/status class	Lake status EQR score/status class	
Phytoplankton	Biomass (mg/l)	5.1	0.5/M	0.5/M	0.5/M	N.A.	0.5/M	0.6/M	
	Cyanobacteria (%)	19.8	0.7/G			N.A.			
	<i>a</i> -chlorophyll ($\mu\text{g/l}$)	26.5	0.5/M			0.5/M			
Macrophytes	Proportion of type-specific-species (%)*	29/35	0.3/P	0.5/M		0.5/M	0.5/M		
	RI (-)	14.29	0.7/G			0.7/G			
	PMA (-)*	38.27/30.04	0.5/M			0.5/M			
Macro-invertebrates	BQI-1 (-)	0.86	0.9/H	0.9/H		0.9/H	0.9/H		
	PMA (-)	0.615	0.9/H			N.A.			
Fish	BPUE (g/night)	1423	0.7/G	0.6/M		0.9/H	0.7/G		
	NPUE (n/night)	131.3	0.3/P			0.5/M			
	Cyprinids (%)	50.3	0.9/H			0.9/H			
	Indicator species (expert judgement)	Normal pop. structure of <i>P. fluviatilis</i> , <i>E. lucius</i> and/or <i>R. rutilus</i>		0.5/M			0.5/M		

tebrate community classified Lake Kirmanjärvi to “high” ecological status irrespective of the classification criteria used. Phytoplankton and macrophytes indicated an elevated level of primary production and consequently classified the lake to “moderate” status for both phylogenies.

Responses to past management measures

A decrease in chlorophyll-*a* and total phosphorous concentration, as well as a subtle increase in secchi depth, can be observed in the water quality data from the late 1980s to 1998 (Fig. 8). The changes coincided with the onset of the hypolimnetic oxygenation (since 1986) and a period of intensive fishing (during 1989–1997; Kauppinen 2006). At the end of the biomanipulation period, the chlorophyll-*a* content improved to “good” (20 $\mu\text{g/l}$) status irrespective of lake type designation. Following the cessation of intensive fishing in 1997, the chlorophyll-*a* and phosphorous concentrations gradually rose while secchi depth decreased (Fig. 8). In the sediment record effects of the lake management period correspond to the upper 12 cm (Fig. 2).

Discussion

Indications of external forcing

The chemical and physical properties of the sediment suggested certain periods of external forcing on Lake Kirmanjärvi that could have affected lake biota and status class. The earliest period of external forcing started around AD 1710 (at 100 cm) with a sudden change in sediment phosphorus binding mode. While the exact reasons leading to these results are unknown, early human settlement has the potential to cause observable changes in lakes (e.g. Miettinen et al. 2002; Heinsalu and Alliksaar 2009). The next disturbance phase manifested itself as a short-lived peak in mineral matter contents at 85 cm (~AD 1755). However, no drastic biological changes were observed at these sediment depths. Major

increases in mineral matter inputs started at 55 cm and peaked at 35 cm (~AD 1850–1910), corresponding to changes in diatom assemblages. Changes in the sediment chemistry were most likely related to deforestation and agriculture in the catchment which increased overall nutrient inputs to lakes. Sediment phosphorus concentrations increased from 40 to 30 cm (~AD 1900–1925) with the intensification of agriculture. The sediment nutrient trend did not mirror the rapid changes in biota at 19 cm but stable and high P concentrations found in the top 12 cm of sediment correspond to the period of artificial hypolimnetic aeration, and may indicate an increase in the sediment P binding capacity. Trends for Pb, S, Ni and V in the early 1980s clearly indicate an increase in fossil fuels-related atmospheric fallout, but this type of forcing is likely to have a negligible effect in eutrophic lakes such as Lake Kirmanjärvi.

Phytoplankton

Paleoecological diatom data show marked changes in diatoms species assemblages from the site specific reference conditions, coinciding with the stages of external forcing caused by human activity. However, even the oldest sediment level analyzed contained species assemblages indicative of high nutrient concentrations (modeled lake water [TP] of 29–38 $\mu\text{g/l}$), suggesting that the lake may be best classified to the naturally eutrophic type. Changes observed in the reference period sediment properties at 100 cm and 85 cm depths did not cause detectable species shifts. In contrast, the increase in inferred phosphorus concentrations coincided with the first signs of agriculture at 55 cm (~AD 1850) and resulted in a major shift in diatom species compositions (PMA ~50%, i.e. 60% of the reference PMA = moderate class) but no decline in species diversity. Model affinity is not employed as a classification metric for phytoplankton in the Finnish system (Table 1) but we interpreted the local diatom-PMA results analogously to other metrics for which it is employed. Also the next major shift in diatom species composition (19 cm, ~AD 1965; PMA 47% of the reference PMA = moderate class) coincided with a peak in mineral matter and

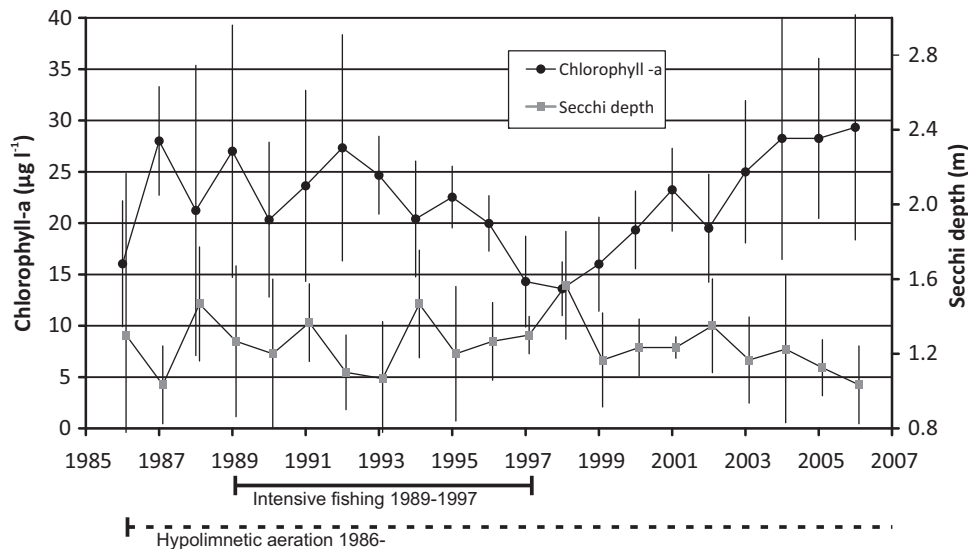


Fig. 8. Changes in chlorophyll-*a* content and secchi depth (annual mean \pm SD) during the productive period (May–September) in Lake Kirmanjärvi between 1986 and 2006.

a transient decline in sediment P concentrations with respect to the phosphorus binding elements Fe and Mn. Overall, the site-specific history of these externally forced changes in the diatom assemblages support the assignment of the 'moderate' status class to Lake Kirmanjärvi based on modern observational data. The situation was not reversed by the management efforts, even though the increases in *Asterionella formosa* in the uppermost 13 cm may be related to the hypolimnetic aeration which is known to favor early blooming species (Webb et al. 1997). It has been shown that oxic sediment surfaces resulting from hypolimnetic aeration may not be effective in enhancing permanent phosphorus retention in lakes (Gächter and Müller 2003; Gächter and Wehrli 1998).

Long term changes in the non-diatom phytoplankton community were substantial in the upper 20 cm of the sediment core (corresponding to LDZ 1). In accordance with the diatom record, the green algae *Pediastrum* spp. and *Scenedesmus* spp. increased, suggesting gradual eutrophication since the 1960s. *Anabaena* akinetes declined simultaneously, a phenomenon which has been attributed to progressive eutrophication (van Geel et al. 1994). However, a concomitant increase rather than a decrease was observed for *Aphanizomenon*, a cyanobacterial species that thrives in lower nutrient concentrations than *Anabaena* but gains a competitive advantage in shaded conditions (de Nobel et al. 1998). Regardless of the cause of these shifts, major deviations from the reference conditions were observed for cyanobacteria and green algae in sediments corresponding to LDZ 1. Elevated concentrations of akinetes and remains of green algae were also found in older sediments, at a depth of 80–60 cm (~AD 1770–1840). These increased concentrations were mirrored only by the chironomid proxy as slightly elevated proportions of *Chironomus plumosus*. The changes may be anthropogenic in origin, but climatic causes cannot be ruled out either. The observed deviations thus support the view of lower than good status class for the study lake based on phytoplankton assemblages. However, akinetes especially of the bloom forming taxon *Anabaena* spp. were found throughout the core, which suggests that also cyanobacteria were present and eutrophic conditions prevailed during the whole study period.

Benthic macroinvertebrates

Dominant chironomid taxa in all samples included *Procladius* spp, *Chironomus plumosus* and *C. thummi* type (probably *C. anthracinus*), and *Tanytarsus* spp. In addition, samples included also

other taxa often associated with shallow and eutrophic lakes (e.g. Brodersen and Quinlan 2006).

We found no concordant changes of the chironomid fauna with the diatom zones i.e. adjacent LDZs did not differ from another with respect to chironomids. This finding is not uncommon as diatom trophic trends can differ from patterns observed for chironomids (Meriläinen et al. 2001, 2003). However when we compared the period of intensive fertilizer-based farming (~ AD 1965–; equivalent to LDZ 1) to the rest of the sediment core distinct changes in the chironomid fauna became obvious. Modern sediments displayed a clear reduction in the abundances of *Chironomus thummi* type and an increase in *Procladius* spp. pointing to prolonged periodic hypolimnetic anoxia. However, the local relative PMA for chironomids suggested a lower than high status only for three samples in the upper 5 cm of sediment, which is in fairly good agreement with the status based on lake type specific PMA data ('good' status, Table 1). Other paleoecological studies of currently eutrophic lakes (e.g. Brodersen et al. 2001; Meriläinen et al. 2000; Kansanen 1986) have shown substantial changes in lake chironomid communities due to anthropogenic eutrophication. In Lake Kirmanjärvi the significant reduction of chironomid diversity in the modern sediments was unparalleled in the BQI values. Meriläinen and Hamina (1993) noted a similar pattern for Lake Puruvesi resulting from indicator taxa substitution (see also Brodersen and Quinlan 2006). In our case, species substitution does not influence the results and given the low pre-disturbance BQI for Lake Kirmanjärvi we suspect that only very severe and continuous hypoxia could have resulted in a noticeable reduction in the modern BQI. Lake morphology can be a major determinant of lake trophic status (Vollenweider 1975) and in a recent study of Finnish lakes, Jyväsjärvi et al. (2009) revealed a distinct link between lake morphometry, in particular depth, and the BQI. Using the morphometric relationship of Jyväsjärvi et al. (2009) slightly overestimated the actual pre-disturbance BQI of Lake Kirmanjärvi (modelBQI = 2 vs. paleoBQI = 1.43). The paleolimnological chironomid data therefore support findings for the phytoplankton community and indicates that Kirmanjärvi is indeed a naturally meso-eutrophic lake.

Classification of ecological status

The ability to set adequate objectives for lake restoration and management is intimately linked to the correct assignment of the correct lake type. The paleolimnological approach taken in

this study revealed Lake Kirmanjärvi's naturally high productivity. Naturally eutrophic lakes have been identified in other paleolimnological studies (e.g. Räsänen et al. 2006a) and these lakes form a unique lake type in the Finnish typology system (Vuori et al. 2006). Using only the standard typology criteria, Lake Kirmanjärvi would have been erroneously considered a small humic lake. In the present case, individual biological quality element-assigned class boundaries differed between the two types only for the fish community which classified the lake as either “moderate” or “good” for the small humic or naturally eutrophic lake type assumption, respectively. Differences in the status assessment for this BQE did not change the overall “moderate” ecological status. This is because contrary to the One-out, All-out (OoAo) principle (European Commission 2005), the Finnish classification scheme (Vuori et al. 2009) utilizes the median score across all BQEs to determine the overall lake ecological status. The outcomes and benefits of these two differing approaches have been discussed in detail by Alahuhta et al. (2009b) and Rask et al. (2010). In Lake Kirmanjärvi, all biological quality elements largely agreed on the ecological status so that the nature of the classification approach (OoAo vs. median) would not have impacted the overall outcome.

In addition to producing comparable overall status class, site-specific (paleolimnological) status assessment mirrored the type-specific ecological status classification at the level of individual BQEs, with the site-specific diatom-PMA and other algal data suggesting moderate status while chironomid-PMA and cyanobacterial akinete data mostly indicated high status. The type-specific assessment method resulted in the conclusion that changes in ecological status had occurred for several BQEs and management efforts are needed to bring the lake back to good ecological status. While this overall assessment was in agreement with the paleoecological classification, the monitoring data approach did not indicate the marked changes in the subfossil chironomid or phytoplankton community observed in the paleobiological record. Even though the metrics and BQEs used to assess a lake's deviation from its type-specific and site-specific historical reference states are slightly different, our results show that paleobiological data provide a viable – and in our case more informative – alternative to the typology-based method of assessing lakes status. The paleolimnological approach employed is especially useful in validating the deterioration of ecological status in cases where there are few pristine sites left to allow monitoring-based type-specific reference conditions to be set.

Implications for lake management

Coinciding with lake management efforts, monitoring detected clear changes in the biomass of phytoplankton, indicated by concentration of chlorophyll-*a*, and epilimnetic phosphorous concentration. Between 1987 and 1997 a total of 200 t of fish, corresponding to an average annual fish catch of 22 t (71 kg/ha/a) were removed from the lake (Kauppinen 2006). The fishing effort was targeted to cyprinids but the species composition of the catch has not been reported. Effects of drastically reduced fish predation on zooplankton with a cascading effect on the phytoplankton are well documented (Mittelbach and Chesson 1987). The current fish community of Lake Kirmanjärvi is dominated by cyprinids, suggesting their substantial role in the functioning of the lake ecosystem. The intensive fishing effort mainly targeted cyprinids which may have reduced internal loading through the reduction of feeding activity-related bioturbation. Unfortunately, we lack monitoring or proxy data on zooplankton and fish to assess the direct or indirect effects of biomanipulation. However the declining chlorophyll-*a* and phosphorous concentrations during intensive lake management efforts and their subsequent increase after the termination of the intensive fishing, suggest that the intensive fishing effort

was effective. Hypolimnetic oxygenation of Lake Kirmanjärvi may have enhanced the effects of biomanipulation by decreasing the release of nutrients during hypoxic conditions (Kauppinen 2006). The increased sediment phosphorus concentrations in the recent sediments support this conclusion. Further management efforts are necessary to restore Lake Kirmanjärvi from its present “moderate” to “good” ecological status. Given the good present day status of the benthic community we believe the objective of attaining “good” ecological status of Lake Kirmanjärvi is possible, although it will require intensive in-lake and catchment restoration measures and follow up monitoring.

Conclusions

- The paleoecological proxies suggested that Lake Kirmanjärvi has been eutrophic with cyanobacterial blooms for several centuries and should be classified to the Naturally Eutrophic lake type instead of the Small Humic lake type. However, this shift did not affect the final ecological status assessment for the lake.
- In this study, ecological status assessment based on site-specific paleoecological proxy data largely agreed with the monitoring-based type-specific status assessment, even at the level of individual BQIs. However, since this may not be the case for every lake, paleoecological data can provide a more detailed picture of past ecological changes for WFD-based ecological assessment.
- While the diatom-based lake status declined to “moderate” already in the mid-1800s, lake status deteriorated most drastically for all proxies during the mid-1960s with the onset of intensive, fertilizer-based, agriculture.
- The effects of lake management actions were observed both in the monitoring data and in the paleolimnological record. The results suggest that further in-lake and catchment management efforts and monitoring are required to reach the goal of “good” ecological status.

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