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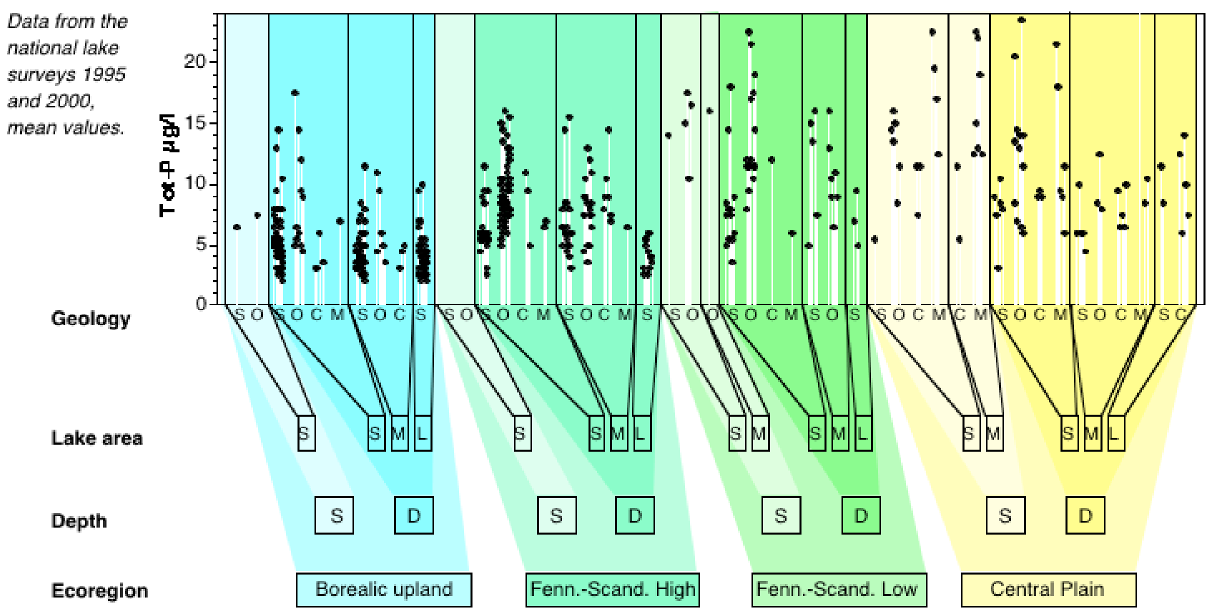

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Background

The Water Framework Directive (WFD)

On December 22nd, 2000, the Water Framework Directive (or Directive 2000/60/EC of the European Parliament and the Council of the 23rd October 2000 'establishing a framework for Community action in the field of water policy') was published in the Official Journal of the European Communities and thereby entered into force. The WFD establishes a framework for the protection of all waters (including inland surface waters, transitional waters, coastal waters and groundwater) which:

- Prevents further deterioration and protects and enhances the status of water resources;
- Promotes sustainable water use based on long-term protection of water resources;
- Aims at enhancing protection and improvement of the aquatic environment through specific measures for the progressive reduction of discharges, emissions and losses of priority hazardous substances and the cessation or phasing-out of discharges, emissions and losses of the priority hazardous substances;
- Ensures the progressive reduction of pollution of groundwater and prevents further pollution; and
- Contributes to mitigating the effects of floods and droughts.

Activities to support the implementation of the WFD are under way in both Member States, and in countries that are candidates for accession to the European Union. Examples of activities include consultation of the public, development of national guidance, pilot activities for testing specific elements of the WFD or the overall planning process, discussions on the institutional framework or launching of research programmes dedicated to the WFD.

The EU Member States, Norway and the European Commission have jointly developed a common strategy for supporting the implementation of the WFD. The main aim of this "Common Implementation Strategy" is to ensure a coherent and harmonious implementation of the Directive. Focus is on methodological questions related to a common understanding of the technical and scientific implications of the WFD.

In the context of this Common Implementation Strategy, a series of working groups and joint activities have been launched for the development and testing of non-legally binding guidance documents. Recommendations regarding typology for inland surface waters are given in the guidance documents of working group 2.3 (REFCOND - Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters) and 2.5 (Intercalibration - Guidance on establishment of the intercalibration network and on the process of the intercalibration exercise). Final versions of these guidance documents are to be published in the beginning of 2003. No common European typology for inland surface waters are suggested in these documents, instead, Member States are encouraged to engage activities to harmonise typology for inland waters on the most appropriate regional scale (e.g. the Nordic countries) in order to facilitate the selection of sites to be included in the intercalibration network.

A typology for lakes and rivers according to the WFD

The aim of typology

WFD requires that Member States differentiate the relevant surface water bodies with respect to type, and that Member States establish type-specific reference conditions for these types. Deviation from reference conditions are then used as the basis for the classification of ecological status of the surface water bodies. The classifications will subsequently be included in the River Basin Management Plans that will be reported to the EU Commission for the first time 2009. The purpose of surface water body types in this case will be to facilitate comparisons of the classifications between different Member States. Another important use of types is for the selection of intercalibration sites (draft register 2003, final register 2004) to be included in the intercalibration exercise during 2005-2006. The purpose of intercalibration in the WFD is to ensure comparable ecological quality assessment systems and harmonised ecological quality criteria for surface waters in the EU Member States. This will be done by establishing harmonised Ecological Quality Ratio (EQR) values for the two key quality class boundaries high/good and good/moderate.

System A or System B?

The types shall be differentiated using either ”System A” or ”System B”. The two systems are similar in that the same obligatory factors are to be used in both: geographic position, altitude, size, geology and, for lakes, depth. The difference is that System A prescribes how water bodies shall be aggregated spatially (ecoregions) and with respect to specific altitude, size and depth intervals, and that System B, besides lacking this prescription, permits the use of additional factors. It is up to Member States to decide on what system to use, and most Member States have indicated that they prefer to use System B, including the Nordic countries.

The Directive requires that System B, if used, must achieve at least the same degree of differentiation as would be achieved with System A. This could be interpreted to mean that if System B is used, it should result in no greater degree of variability in type specific reference conditions than if System A had been used. However, if it can be demonstrated that the same or a lower degree of variability in reference condition values may be achieved with a lower number of types, this should be acceptable, since the main purpose of typing is to establish type-specific reference conditions as precisely as possible.

System B provides, as indicated above, a greater degree of freedom in how types are designated and, subsequently, the possibility to designate type-specific conditions with, possibly, a lower variability. The starting point is a sound database. It should contain the obligatory factors and those other factors that the Member State may find useful. All or some of the Directives optional factors may be included here, but other factors may also be included. In the Nordic countries, for example, the highest coastline during the last glaciation and the tree line are considered ecologically more relevant than using the fixed altitude classes, 200m and 800m, prescribed in System A. Furthermore, some chemical measure of humic substances (colour, absorbance or TOC) and calcareous impact (Ca or alkalinity) are considered more relevant than the prescribed geology classes “organic” and “calcareous”.

Delineation of types

Based on the data available, types may be delimited using various mathematical-statistical clustering methods or using more intuitive methods, including expert opinion. The combination of factors used should, of course, result in as ecologically meaningful types as possible, which includes a low degree of variability within types, without an excessive number of types being designated

Whether mathematical methods or more intuitive methods are used, it is important to realise that the designation of types involves the consideration of, sometimes, mutually incompatible factors and that the result is a compromise between these. What may be considered good types with respect to one quality variable may not be the ideal ones with respect to another variable. This is easily demonstrated by the use of various multivariable techniques. Hence, a typology common to all variables is likely to be "the least bad alternative" for the purpose of deriving reference conditions of a multitude of variables.

Small water bodies

The purpose of the WFD is to establish a framework for the protection of all waters including inland surface waters, transitional waters, coastal waters and groundwater. Member States must ensure that the implementation of the WFD's provisions achieves this purpose. However, surface waters include a large number of very small water bodies for which the administrative burden of management may be enormous.

The Directive does not include a threshold for very small "water bodies". However, the Directive sets out two systems for differentiating water bodies into types, System A and System B. Only the System A typology specifies values for size descriptors for rivers and lakes. The smallest size range for a System A river type is 10 – 100 km^2 catchment area. The smallest size range for a System A lake type is 0.5 – 1 km^2 surface area. No sizes for small transitional and coastal waters are given. The application of system B must achieve, at least, the same level of differentiation as system A. It is therefore recommended to use the size of small rivers and lakes according to system A. However, it is recognised that in some regions where there are many small water bodies, this general approach will need to be adapted. Having said that, it may be appropriate to aggregate water bodies into groups for certain purposes in order to avoid unnecessary administrative burden (this procedure is outlined in a horizontal guidance document on the application of the term "water body" in the context of the WFD). However, there are still large numbers of discrete rivers and lakes that are smaller than these thresholds. A possible approach for the protection of these waters is outlined in the horizontal guidance document.

Member States have flexibility to decide whether the purposes of the Directive, which apply to all surface waters, can be achieved without the identification of every minor but discrete and significant element of surface water as a water body.

The Swedish national survey – a unique means to test the relevance of the typology

The Swedish national lake and stream surveys of 1995 and 2000 included both chemical and biological (benthic macroinvertebrate) measures.

The selection of lakes for sampling was made by a stratified random selection from the Swedish lake register (Swedish Meteorological and Hydrological Survey). In its present digitised form it contains about 100 000 lakes. Stratification was made with respect to

- Previous knowledge about water chemistry variation
- Number of lakes in each EMEP50 grid (50x50 km)
- The sample as defined by criteria 1 and 2 should result in a sample representing between 1% and 8% of the lake population
- The proportions of the different size classes 0.04–0.1; 0.1–1; 1–10; 10–100 and > 100 km^2 should be 1:1:4:8 and all lakes in the largest size class.

Thus we could reasonably well cover the whole country, with some emphasis on more polluted (mainly long-range transported pollutants) areas. Also the more important, larger lakes were focused without neglecting the small ones. Nordic co-ordination of the lake survey in 1995 required selection of lakes > 0.04 km^2. Since the smallest size in the Swedish lake register was 0.01 km^2 we had to divide the smallest size class. For the number of lakes of that size class to be sampled in each area a random selection was made from the lake register. Then the actual area was measured on maps and those within that size requirement were sampled. Based on this work covering several hundreds of lakes we estimated that half of those in the class 0.04–0.1 km^2 had areas above 0.04 km^2. Thus we obtained an estimate of the lake population in the class 0.04–0.1 km^2. The sample size was initially determined to be about 3000 lakes and the above exercise resulted in 3025 lakes. Out of this sample every third was selected for trace metal determinations and randomly out of that subset 700 for sampling of littoral fauna (700 lakes).

Sampling sites for benthic macroinvertebrates in streams were randomly selected from the Swedish Hydrological and Meteorological Institute's watercourse and catchment register as part of the Swedish National Stream Survey of 1995 (Wilander et al., 1998). Sites were stratified according to size; 350 within catchments of 15-50 km^2 and 350 within the size 50-250 km^2. Site selection was constrained to within 100-600 meters upstream of the randomly selected sampling coordinates and a 50-meter reach (sampling site), of homogenous substratum (preferably hard-bottom) and flow was chosen in each stream. Benthic macroinvertebrate samples were taken in a ten-meter sampling area situated in the downstream part of the sampled reach.

Samples of benthic macroinvertebrates were collected in exposed litoral zone and in streams using standardised kick sampling (European Committee for Standardisation 1994) with five samples (1 m x 1 min) taken at each site using a handnet (500-m mesh). Lake water chemistry was sampled at 0.5 m depth at the centre of the lake. For rivers, water chemistry was sampled at the site for collecting benthic fauna. The large temporal variation of water chemistry in streams makes the results from one single sample from a stream less representative than that taken for a lake. All chemical analysis were performed by certified laboratories according to EN or ISO standards.

The aim of this report

The aim of this report is to present a suggestion for a typology of Swedish lakes and rivers, that can be a part of a Nordic common typology according to the WFD. The second aim is to present a method to compare different alternatives of ordinating rivers and lakes into classes by using data from a survey of water chemistry and bottom fauna in lakes and rivers. First we evaluate whether there is a relationship between the potential descriptors for a typology according to systems A and B within the WFD, and other ecologically relevant variables. Then we test different suggestions to class borders and number of classes. Based on the evaluation we will give a suggestion of a typology for Swedish lakes and rivers. Finally we want to test the suggested typology to see if the types have well defined reference values for two selected quality element metrics.

Statistical methods

To evaluate whether a suggested descriptor was relevant for constructing a typology we mainly plotted water quality metrics, such as concentrations and benthic fauna indices against the descriptor. Only for ecoregion, that is a nominal variable, did we tested the difference between means with a t-test and Tukey-Kramer correction for multiple tests.

For testing how well a classification could separate water bodies into groups, we used discriminant function analysis (DFA). DFA is a multivariate technique where the discriminant function is a linear combination of the input variables that gives the best separation of the predefined groups. To measure of how well the groups were separated, we calculated the percentage of waterbodies that were predicted to the correct class by the DFA. This was done by cross validation. The input variables we used were three BF indices: Number of taxa, ASPT and Medins index, and 9 water chemistry variables: SO_4, Cl, NH_4, NO_3, Tot-N, Tot-P, $AbsF_{420/5}$, Si, and ANC. Several water chemistry variables were excluded, since they were too intercorrelated.

The environmental variables suggested for the typology were also tested by canonical correspondence analysis (CCA) where the abundance of all BF species was used in the analysis. CCA is a multivariate statistical method that combines ordination with multiple linear regression. Using the forward selection option in CCA, one can test which of the included environmental variables can best explain (i.e., partition) the variability in the benthic macroinvertebrate dataset.

Criteria for exclusion of waters with non-reference conditions.

The aim of this study was to test different alternatives of classifying waters into types with data from water bodies with reference conditions. However, the data from the national surveys contains a large number of sites that are, to a degree, affected by human activities. Our first task was therefore to identify the polluted sites.

Eutrophication

Increased input of nutrients from agriculture land, sewage and industries have caused eutrophication in a large number of lakes and rivers, particularly in southern Sweden. Since the primary production in most freshwaters is controlled by phosphorous, we have used elevated concentrations of Tot-P as the criteria for eutrophication. For lakes we used the principle of calculating a background level of Tot-P as in the Environmental Quality Criteria

(SEPA 1999). The main natural source of phosphorus to a lake is then assumed to be humic substances, here measured as absorbance at 420 nm on filtered sample in a 5 cm cuvette ($AbsF_{420/5}$). However, the relationship between Tot-P and $AbsF_{420/5}$ has been shown to change over time. Therefore, we did not use the formula in the Environmental Quality Criteria that is based on date from the 1980's. Instead we used mean values from 1997-2000 of $AbsF_{420/5}$ and Tot-P from 139 reference lakes with <0,1% agricultural soils and no point sources of phosphorus in the catchment. We then established the linear relationship:

$$\text{Tot-P}_{ref} = 27.7 * AbsF_{420/5} + 13.3$$

where the slope was estimated by Theils slope and the intercept as the 95 percentile of the residuals. Tot-P_{ref} was then calculated for all lakes in the lake survey and those with a Tot-P concentration exceeding Tot-P_{ref} were regarded as eutrophied and excluded from the evaluation of the typology.

In rivers, the variation in water chemistry is much larger than in lakes, and the use of one sample for assessing the water quality is more doubtful. Instead, we regarded rivers with >10% agricultural soil in the catchment as eutrophied. 408 lakes and 138 rivers were regarded as eutrophied, and were removed from the database.

Acidification

Acidification has been a serious problem in Sweden for several decades with decreases in pH, alkalinity and ANC followed by changes in biota. The acidified lakes were identified using a steady state water chemistry model which compared the present water chemistry with the preindustrial conditions that are estimated from the present chemistry. We used the alternative method in the Swedish Environmental Quality Criteria where the assessment is based on ANC. The reason for using ANC instead of alkalinity, is that the estimation was only made on one sample, and that ANC changes less over time compared to alkalinity. A lake was regarded as acidified when the $ANC_{present}/ ANC_{preindustrial}$ was less than 0.75. For rivers, where the water chemistry is much more variable than in lakes, the method we used for lakes gave unrealistic results. Instead we used excedance of critical load as the criteria for acidification. 159 lakes and 19 rivers were regarded to be acidified.

Liming

One way to relieve the harmful effects of acidification on biota in surface waters by has been to lime the waters. Although liming may stop further damages to organisms by acidification, the recovery of species earlier extinguished may take several years, so the present species composition may not be equal to the pristine condition. Further, the liming often changes the alkalinity Ca concentration far above the natural level. We have thus excluded all limed waters from the dataset, as well as waters affected by liming in their catchments. Information on which waters are affected by liming has been collected from the local governments that are responsible for the liming programs. The outcome was that 854 lakes and 188 rivers were affected by lime.

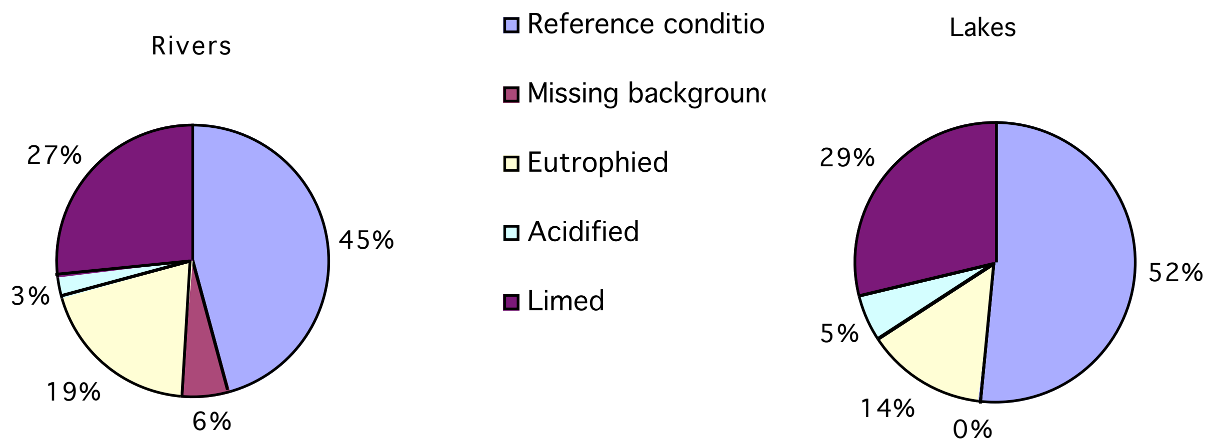


Figure 1 a and b. Distribution of anthropogenic impact on rivers (a) and lakes (b) from the national surveeys of lakes and rivers 1995 and 2000.

Other factors not considered

Eutrophication, acidification and liming were the only factors we regarded when excluding water bodies affected by human activities from the dataset. Another major impact on the fresh water ecosystem has been physical impact, for example, hydroelectric power dams, ditching of forest land, other forestry activities and clearing of watercourses for timber transport. The knowledge of the ecological effects of these activities is poor, but the influence on water chemistry is probably negligible. For benthic fauna the major influence from those impacts would have been the loss of habitats. Since the sampling of benthic fauna is restricted to a well defined habitat, the influence of physical impact on the species composition on the sampling sites is probably not important.

Other anthropogenic influences not regarded are organic pollutants, heavy metals and introduction of alien species.

Testing of descriptors and class limits for a typology by data on water chemistry and bottom fauna.

The WFD gives two alternatives for establishing a typology for water bodies, system A and system B. System A is a rigid predefined system whereas system B is more flexible. However in system B most variables used in system A are included as obligatory variables in system B, only with ecoregion replaced by latitude and longitude. Any use of optional factors will then result in at least a twofold number of classes in system B compared to system A. In Sweden, system A theoretically gives 108 river types and 324 lake types. Many of these types however do not exist or only contain a few number of objects, mainly because altitude is strongly correlated to ecoregion and lake depth is strongly correlated to lake area. Since all types are considered to be represented by a network of reference stations, where all quality elements should be monitored, the cost for a typology with so many types will be tremendous (SEPA, 2002). Therefore, our first aim is to reduce the number of types and to avoid types with very few objects. The typology must then be as simple as possible and any additional factor is only accepted if it markedly decreases the within-type variation. Another restriction when creating a typology is that many of the optional factors in system B are practically not possible to measure for a large number of objects and are thus not included in this report.

Table 1. Factors for differentiating water bodies into types according to systems A and B in the WFD.

Rivers

System A

Ecoregion: Ecoregions shown on Map A in Annex XI
Altitude: <200 m, 200 to 800 m and >800 m
Catchment area: 10 - 100 km^2, 100 - 1 000 km^2, 1 000 - 10 000 km^2 and > 10 000 km^2.
Geology: Calcareous, Siliceous and organic.

System B

Obligatory factors altitude
latitude
longitude
geology
size

Optional Factors
distance from river source
energy of flow (function of flow and slope)
mean water width
mean water depth
mean water slope
form and shape of main river bed
river discharge (flow) category
valley shape
transport of solids
acid neutralising capacity
mean substratum composition
chloride
air temperature range
mean air temperature
precipitation

Lakes

System A

Ecoregion: Ecoregions shown on Map A in Annex XI
Altitude: <200 m, 200 - 800 m and >800 m
Mean Depth: <3 m, 3 - 15 m and >15 m.
Lake surface area: 0.5 - 1 km^2, 1 - 10 km^2, 10 - 100 km^2 and >100 km^2.
Geology: Calcareous, Siliceous and organic.

System B

Obligatory factors altitude
latitude
longitude
depth
geology
size

Optional Factors mean water depth
lake shape
residence time
mean air temperature

air temperature range
mixing characteristics (e.g. monomictic, dimictic, polymictic)
acid neutralising capacity
background nutrient status
mean substratum composition
water level fluctuation

Ecoregions and geographical parameters

The spatial variation of ecological parameters as species compositions and water chemistry is mainly related to climatic gradients. The variation with latitude and altitude is obvious, but there is often also a variation with longitude. In the Nordic countries the longitudinal variation is related the distance to the Atlantic or the Baltic Sea. While this geographical variation is often continuous, sometimes more discontinuous changes are found, justifying the division of the landscape into ecoregions with differing ecological characteristics. One such border is the "*limes norlandicus*" which coincide with the border between the ecoregions Central plain and the Fenno-Scandian shield in Sweden.

Ecoregions

At the top of the hierarchy for system A is Ecoregion according to map A in Annex XI in the WFD. Sweden is divided into three ecoregions, Central plains, Fenno-Scandian shield and Borealis uplands. However, in the other Nordic countries most of the land surface is covered only by one ecoregion. In terms of water body types and reference stations, this means that Sweden will have three times more water types, which means three times more reference stations. This highlights the advantage for Sweden of co-operating with neighbouring countries, since several countries can share reference stations if they have the same water type.

The relevance of ecoregions for dividing water bodies into types is exemplified in Figures 2 a and b. All ecoregions are significantly different according to the number of taxa of Benthic fauna and Tot-P (t-test with Tukey-Kramer) although the overlap between the groups is large.

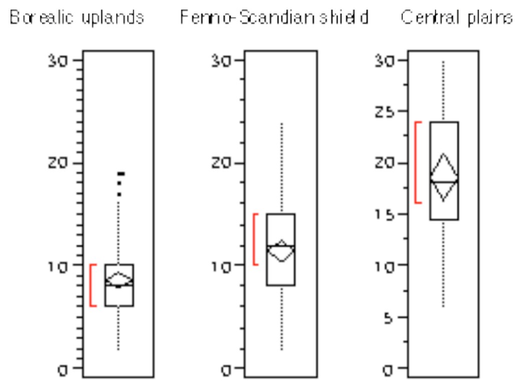


Figure 2a Number of taxa of benthic fauna in Swedish non-polluted lakes separated by ecoregions

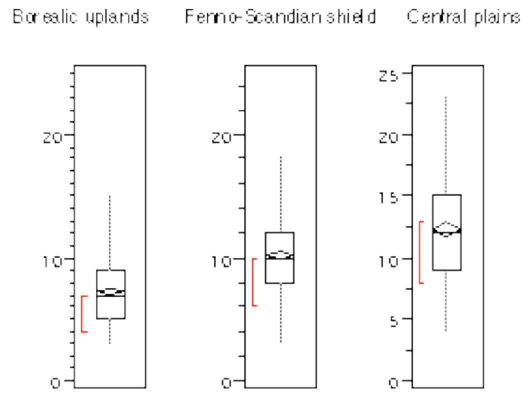


Figure 2b Total phosphorus in Swedish non-polluted lakes separated by ecoregions

Latitude

Latitude is one of the optional factors in System B. The latitudinal variation of WC often shows a tendency of decreasing concentrations from south to north although with a large variation. There are no sharp gradients supporting any limits for classes. Exceptions are ANC, where regions of calcareous soils give high values around the latitudes 58° and 63°.

For benthic fauna, a sharp gradient around 62°N is found. North of that latitude, no communities with more than 20 taxa are found. North of 62°N communities with less than 10 taxa becomes much more frequent due to the lakes in the ecoregion boreal highlands. This border is close to the *limes norlandicus* i.e. the border between the ecoregions central plain and Fenno-Scandian shield.

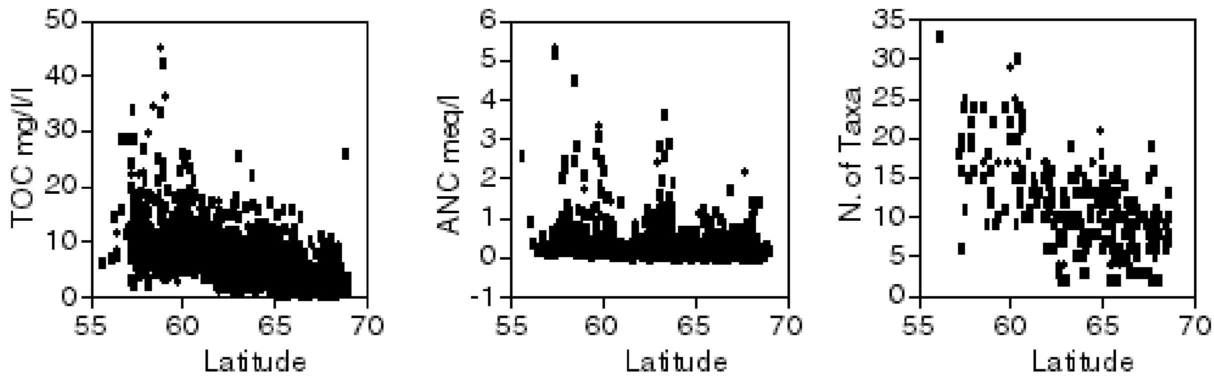


Figure 3. TOC, ANC and number of taxa of benthic fauna in Swedish non-polluted lakes plotted against Latitude.

Longitude

The variation with longitude is totally different in northern Sweden compared to southern Sweden. In northern Sweden there is a change of altitude with high elevation in the west down to the Baltic Sea in the east. Since this gradient is not perpendicular to the north-south axis, it will be poorly represented by a plot of ecological variables against the longitude. Classifying by Ecoregion or Altitude is then more relevant. In southern Sweden an east-west gradient in climate is found due to the distance to the Atlantic Sea. Plots of concentrations and BF indices show some east-west variation for several variables in southern Sweden, but do not suggest any natural border to separate the part of Sweden belonging to the central plain into further regions. For fish populations however lakes and rivers draining to the Atlantic sea shows a much lower species composition, since the high marine salt content prevents fish from migrating between river systems (SEPA, 2002). It might therefore be relevant to divide the region of the Central plain into two regions with one draining to the west cost north of Öresund.

Altitude

Altitude is included in system A with 200m and 800m as borders between types. In system B, altitude is one of the obligatory factors. The relevance of the factor is obvious when the ecological variables are plotted by altitude. Number of taxa of BF and most concentrations decreases by increasing altitude as well as the variations. The changes are however more or less continuous. Another problem with altitude is that it is correlated to the ecoregions. The 800 m level is close to the tree limit and the ecoregion Borealic uplands (see Figure 5). However, in the northernmost part of Sweden, the tree limit is below the 800 m line while it is the opposite case in central Sweden. The altitude of 800 m as a demarcation for the alpine region will be compared to the ecologically defined alternatives below. The 200 m limit will also be compared with the highest coastline since the last glaciation (HC) further on.

The border between the Central plain and the Fenno-Scandian shield

The ecoregions used in system A, mainly describes climatic and geological differences between regions, which is mostly relevant for e.g. terrestrial plants and benthic fauna. For fish, where the migration history is important for the species composition, other regions may be more relevant. One example of this is the border between ecoregions Central plains and Fenno-Scandian shield, that in eastern Sweden is defined by the river Dalälven. In a suggestion for Swedish fish regions (SVEFIRE) by the Swedish National Board of Fisheries an alternative stretch of the divide between southern and northern Sweden was suggested, following the southern border of the catchment area of Dalälven and the northern border of the catchment area of Klarälven in the west (Figure 4). A disadvantage is that the north western part of the southern ecoregion will reach up in the alpine region. Although an ecoregion defined by river basins will be more relevant for fish where migration is important for the species composition, it is not obvious that this preferable for BF and WC. The possibilities of testing the differences between the two definitions of a northern and a southern part of Sweden by DFA on BF and WC is limited, since the difference is only one river and two lakes in the data set.

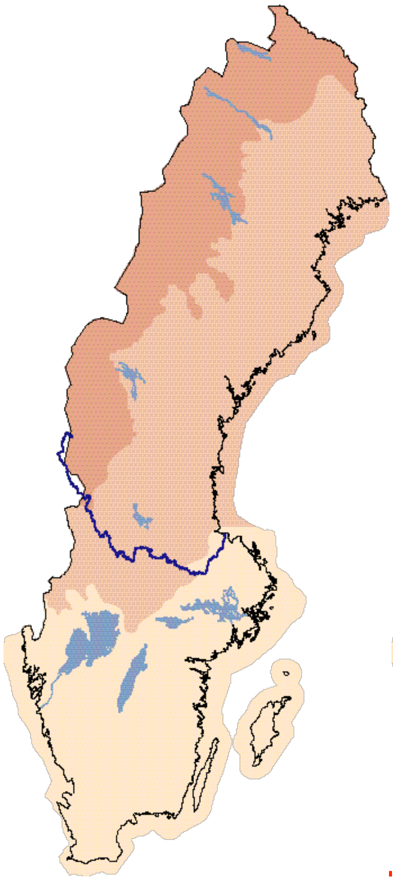


Figure 4. Ecoregions in Sweden and an alternative stretche of the limit between the the Central Plain and the Fenno-Scandian shield.

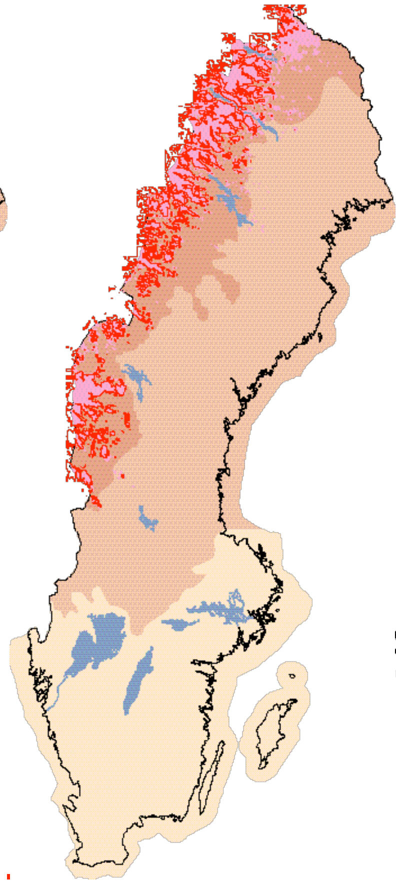


Figure 5. Tree limit (pink), 800m limit (red) and ecoregions in Sweden.

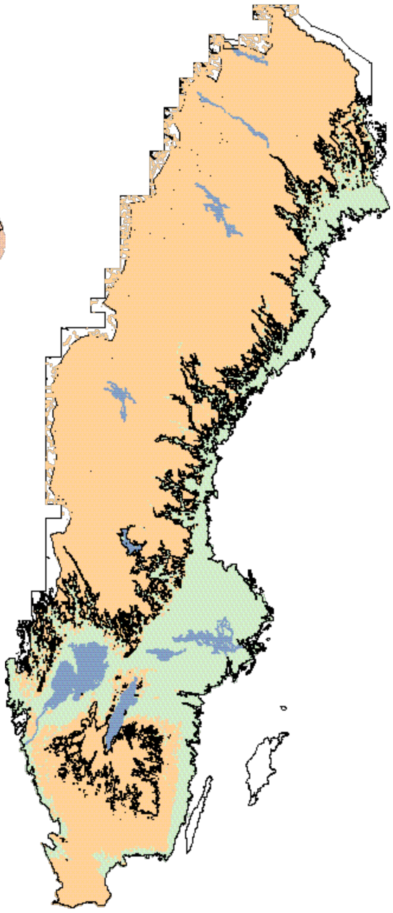


Figure 6 The Highest coastline (black line) and 200 limit (colour) in Sweden.

Demarcation of the alpine region.

As an alternative to the Borealic upland, the Tree limit has been suggested as a demarcation for the alpine region. The altitude of 800 that is used for classifying by altitude in System A, can also be seen as an alternative to the Ecoregion (Figure 6). To test whether the Ecoregions, the Tree limit or the 800m altitude is preferable for typing freshwater bodies, we undertook a discriminant function analysis (DFA) on a selection of chemistry variables and BF indices. We only used data above an altitude of 200m in the ecoregions Borealic uplands and Fenno-Scandian shield. The Tree limit was defined using the general map at 1:250 000.The analysis showed that for lakes, the Ecoregions gives a higher proportion of correct estimations than the Tree limit (Figure 5). The 800m level gave even poorer prediction for lakes. For rivers, there was no difference between Ecoregion and Tree limit. The analysis could not be performed for the 800m altitude for rivers since only one observation had an elevation above 800m.

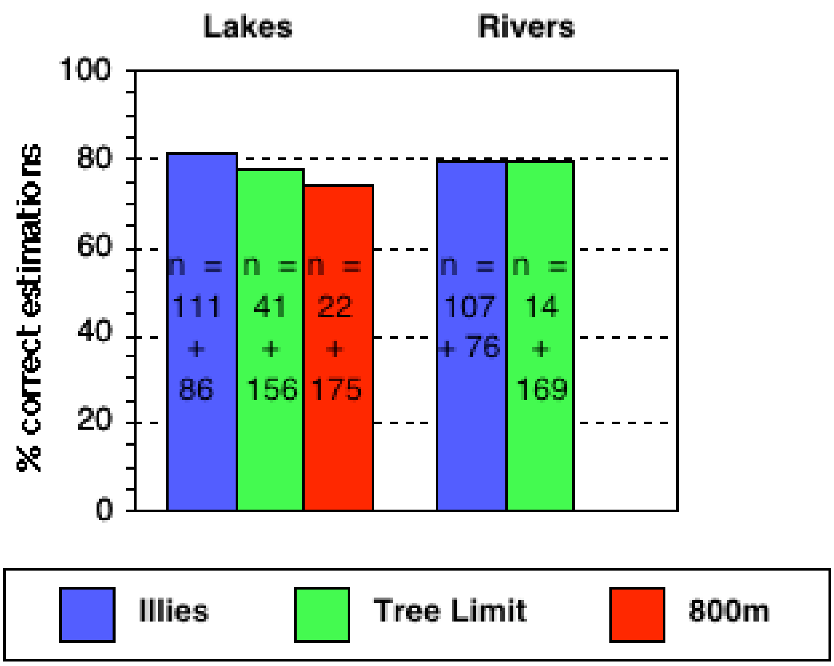


Figure 7. Proportion of correct estimations of lakes and rivers in northern Sweden. The number of objects each group are given within the columns with the highest elevated class first.

As an alternative to the 200m limit, the highest coastline since the last glaciation (HC) can be used. The altitude HC varies from close to the sea level in southernmost Sweden to about 200 m a.s.l in northern Sweden. In areas below the HC, the geology is affected by having been submerged, which is reflected in the morphometry of lakes and rivers and also in the water chemistry. The HC is also important for fish populations, related to migration history. We can test whether HC or the 200m limit is better at separating the water bodies into groups by DFA according to WC and BF. We did this separately for the ecoregions Central plains and Fenno-Scandian shield.

The DFA indicate a higher precision for altitude 200m than using the HC according to WC and BF. The difference is most pronounced for lakes. In southern Sweden however there are few sampled objects in the higher elevated types and no rivers at all above 200m, so the results are very uncertain for that region.

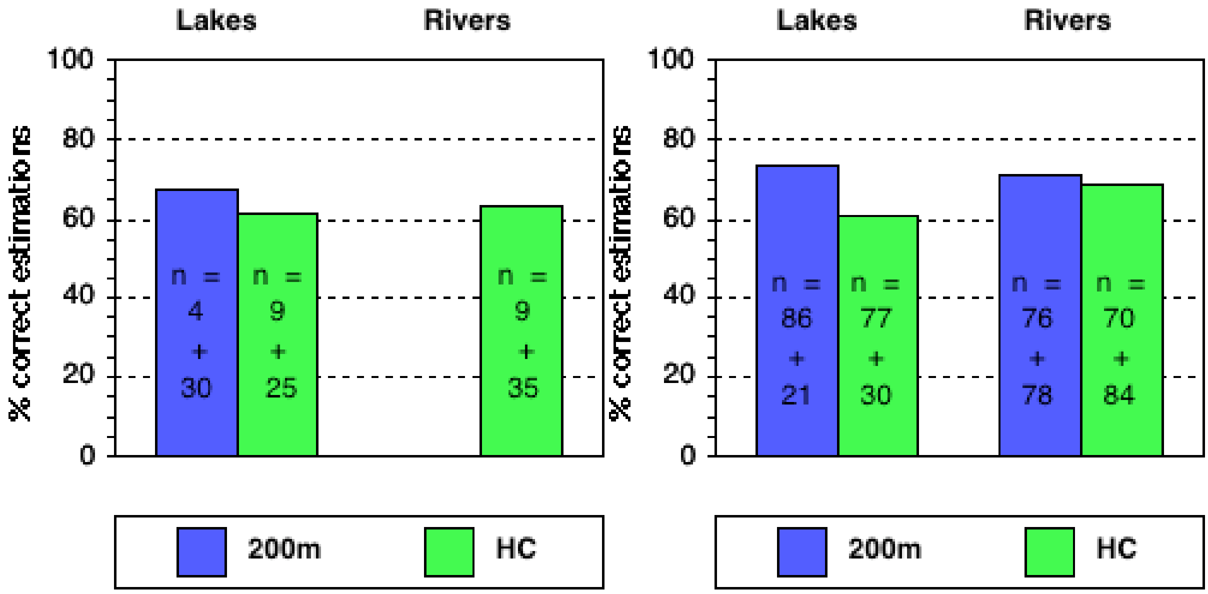


Figure 8 a and b. Proportion of correct estimations of lakes and rivers into types by altitude in southern Sweden (a) and northern Sweden with the Borealic uplands excluded (b). The water bodies are typed either by altitude 200m or by the highest coastline (HC). The number of objects each group are given within the columns with the highest elevated class first.

Differentiation of southern Sweden by connection to the Atlantic Sea

To test whether the eastern and western parts of the Central plain in Sweden are different according to water chemistry and bottom fauna we run a DFA to see if the water chemistry and BF can be used to predict whether a water body belongs to either of the two groups. For lakes, the number of correctly estimated objects is only 50%, which means that the two groups cannot be separated (Figure 9). This might however be caused by the few objects with BF data in this region. If only chemistry is used in the analysis, many more objects are present, and 74% are correctly estimated. For rivers an even higher prediction rate was observed, 77%, with BF included. Since WC is included in the analysis, the difference might be caused by the higher levels of marine SO_4 and Cl with little ecological relevance.

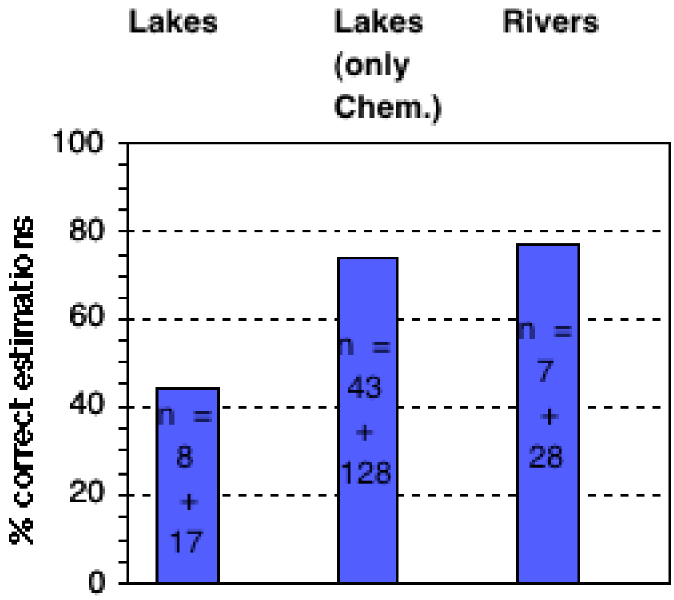


Figure 9. Proportion of correct estimations of lakes and rivers in southern Sweden below the highest coastland grouped into one east region and one west region. The number of objects each group are given within the columns with the highest elevated class first.

Conclusions

The geographical variables are strongly correlated to each other, and a strict application of either System A or the obligatory factors of System B in the WFD will result in a vast number of types where most will only contain a few object and the types will have very small differences in reference conditions. Based on the results presented above we suggest a division of Sweden into 4 regions: 1) The Borealic upland, defined by Annex XI in WFD. 2) The Fenno-Scandian shield above 200 m. 3) The Fenno-Scandian shield below 200m. 4) Central plain. The central plain may then optionally be divided into one Eastern part and one Western part.

Geological factors.

Calcareous

Geology is both a factor in system A and an obligatory factor in system B. According to system A one type should be defined by the presence of calcareous bedrock in the catchment. A high content of calcareous material in the catchment results in hard water systems with a characteristic flora of macrophytes and phytoplankton. Hard waters are also resistant against acidification. The problem with the Nordic countries, especially in Sweden, is that the soil material covering the bedrock was transported from distant locations during the last glaciation. Since it is rather the soil than the bedrock that defines the chemistry in runoff it is more relevant to look for the presence of calcareous minerals in the soil. This is however not possible to do at a larger scale since there are no detailed maps with soil mineral content available. If we use the best available maps of geology and estimations of regions with calcareous soils, we find a poor correlation between geology and water chemistry. For example 47% of the water bodies with >10% calcareous geology in the catchment had less than 0.4 meq/l Ca/l, while 15% of the objects with less than 10% calcareous geology in the catchment had a Ca concentration >0.4 meq/l. We therefore suggest that the impact of a calcareous geology is defined by water chemical variables, such as Ca, ANC or alkalinity. There is a drawback of using a measured variable for typing since it may be affected by anthropogenic impact. Acidification decreases the ANC and alkalinity whereas the Ca content is elevated during increasing acidification and may get lower than the preindustrial levels during recovery. In Sweden liming has elevated both Ca, ANC and alkalinity in many waters. The water chemical variables can however be corrected for this impact, and it is the calculated preindustrial concentrations that should be used in the classification.

In literature, a limit of 1 meq/l Ca is found as the limit for hard water lakes (Wetzel, 1983). According to the lake survey water data only 4 % of the non-polluted lakes in Sweden are regarded as hard or 6% if all lakes in the survey 1995 are considered. In the Finnish suggestion of a typology of water bodies according to WFD, a limit of 0.4 meq/l alkalinity was suggested. This would include 17 % of all lakes. The lower limit would be more relevant to acidification, since the ecology in waters with >0.4 meq/l are not likely to be acidified at the present levels of deposition of acidifying pollutants. For water bodies with an alkalinity close to the 0.4 meq/l-limit, the influence of acidification and liming on the alkalinity may be significant, and the preindustrial value has to be estimated. There has also been a suggestion of an even lower threshold, 0.2 meq/l alkalinity. The ecologically most relevant threshold is probably dependant on what variables that are considered. A lower value is probably more relevant for fish whereas a higher value will differentiate hard water lakes with a macrophyte community dominated by *chara* species. If we look at the BF data we get some qualitative support for the 0.4 meq/l limit. Below that level there is a much higher chance of finding communities with few species and a low Medin index (an acidification index) (Figure 10).

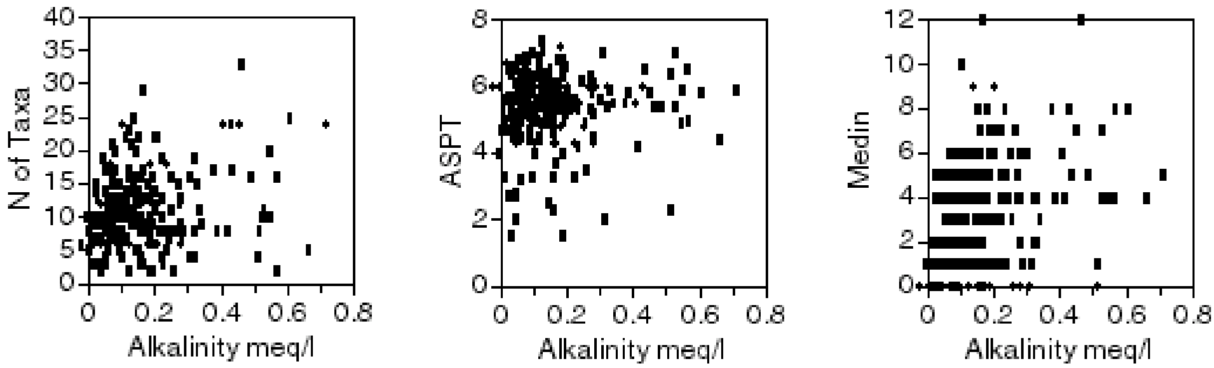


Figure 10 BF indices plotted against alkalinity in non-polluted lakes within the Swedish national survey 1995.

If we analyse how well two classes can be separated by BF data with DFA, we can find an even stronger support for the 0.4 meq/l limit (Figure 11). The results are somewhat biased though, since there is a higher probability of obtaining a correct estimation when the distribution between the groups is skewed.

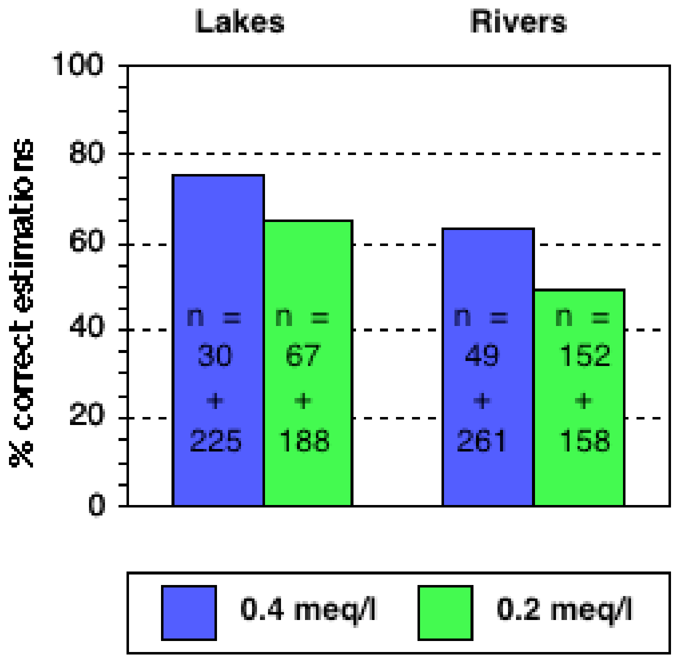


Figure 11 Proportion of correct estimations of lakes and rivers into classes by alkalinity with two threshold values. No. of objects in each class is found within the columns with the high alkalinity group first.

Besides Ca and alkalinity, ANC can be used for classifying waters according to impact of calcareous geology in the catchment. In waters with a high humic content, organic acids have consumed a part of the alkalinity delivered by minerals in the catchment. Since ANC can be defined as the sum of alkalinity and organic acids, it can be seen as a measure of how the alkalinity would have been without organic acids. ANC also has the advantage of having a lower temporal variability than Ca and alkalinity and, therefore is a more reliable variable. When comparing the two surveys, 1995 and 2000, the concentrations of Ca and alkalinity were on average c. 0,030 meq/l lower in the latter survey, due to high flow conditions, whereas ANC was only 0,015 meq/l lower. Finally, Ca is the most easily defined variable with no methodological ambiguities. We tested the three suggested variables by running DFA on BF data from the non-polluted lakes (No. of Taxa, ASPT and Medin). We did not include WC variables in the analysis since those were used for the classification. We then compared the results from classifying the data by alkalinity, Ca and ANC. The class borders were set so that there were equal numbers of objects in the two groups for all three variables. This was obtained by using alkalinity 0.40 meq/l, Ca 0.49 meq/l and ANC 0.52 meq/l as limits. The results indicated no difference between the three variables for separating BF indices into groups. 76% of the lakes were correctly classified in all three cases.

There is need for a broader discussion including other biological variables than BF before a decision can be made on which variable and what threshold value should be used for classifying the impact of calcareous geology in the catchment. Our preliminary suggestion is to use Ca as the defining variable for impact of calcareous geology. The Ca concentration is directly related to weathering of calcareous minerals and is not affected by the presence of organic acids. We further suggest the use of 0.5 meq/l as the threshold, this is comparable to an alkalinity of 0.4.

Organic soils

An important ecological factor for freshwater is the concentration of natural organic matter (NOM), mainly humic substances with terrestrial origin from the catchment. A high humic content in the water gives a more bacteria dominated food chain and also reduces the light penetration depth, with effects on the macrophyte communities and temperature distribution in the lake. The main source of NOM are wetlands, where the organic content in the soils is high. It is however not possible to relate, for example, the water colour or absorbance to the distribution of mires in the catchment, which is shown in Figure 12. There are several reasons for this. Firsly, that large areas of forests in Sweden are wetland forests or ditched wetlands, not regarded as mires on the maps used for the analysis. Another reason is that the contribution of NOM from wetlands to runoff depends on the location within the catchment. Narrow strips of riparian wetlands, not shown on maps, can be the major source for NOM in some catchments. Finally, a high turnover time of the water in a lake or in lakes upstream in the catchment highly reduces the content of NOM by biodegradation, photodegradation and precipitation. At the present time, we don't see a possibility to predict the expected content of NOM from catchment characteristics. Instead the classification has to be done using water colour or the absorbance at 420 nm on filtered sample in a 5 cm cuvette ($AbsF_{420/5}$). The colour or absorbance is preferable to TOC, since the latter is more affected by human impact through eutrophication. A risk with using water colour/ $AbsF_{420/5}$ for typing lakes is that the leaching of NOM may have changed through human activities, such as forestry and ditching of wetlands. At the moment, though, we don't see any alternative to using the water colour for classifying the water according to impact from organic soil.

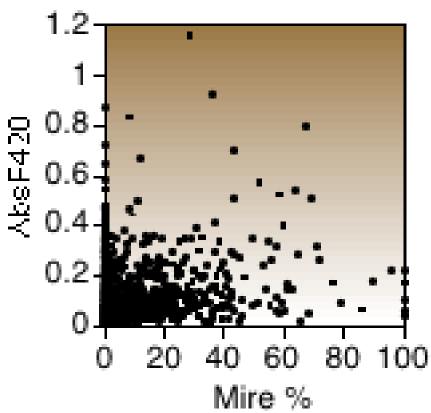
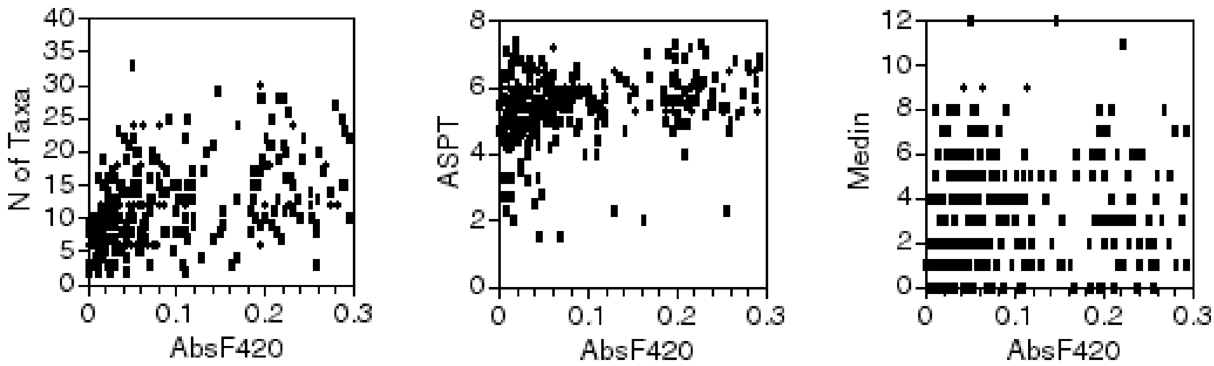


Figure 12. Absorbance at 420 nm on filtered sample, plotted against the relative distribution of mire in the catchment

The ecological impact of NOM is continuous, and there is no evident threshold value for setting a class limit. For BF, there is only a weak relationship between colour and index, with no natural threshold (see Figure 13). In the Finnish suggestion to a typology, three types according to water colour were suggested, < 30 mg Pt/l, 30 - 90 mg Pt/l and > 90 mg Pt/l (0.06 $AbsF_{420/5}$ and 0.18 $AbsF_{420/5}$). BF data may support the limit of 0.06. It is however possible that water colour has a higher influence on other variables, such as phytoplankton and macrophytes. We can thus only give a preliminary suggestion of using two types according to water colour with 0.06 AbsF420 as the limit value.



Figur 13. Variation of BF indices by $AbsF_{420/5}$ in non-polluted lakes within the Swedish national survey 1995.

Calcareous and organic

According to system A water bodies are either dominated by siliceous, calcareous or organic geology in the catchment. By using alkalinity and colour to define the classes we end up with a number of lakes that are influenced by both calcareous and organic geology, i.e. they are both high in colour and alkalinity. If we use the highest limits suggested for class limits, only 1% of the lakes in the national survey are both coloured and hard, (polluted lakes included) (Table 2). We can then define this category as a rare species, excluded from the typology and the intercalibration network. If we on the other hand use the lowest class limits, as many as 24% of the lakes will belong to that class and would therefore probably need to be treated separately within the typology. If we use the limits of alkalinity 0.4 meq/l and AbsF420 0.06, 11% of the lakes are included. We can use the BF data from non-polluted waters to test if the category brown hard waters can be separated as a group by DFA. If only the brown waters were considered, 2/3 of the water bodies within the class with hard brown waters were correctly classified, both for lakes and rivers. The same results were found if only the hard waters were included in the analysis. Consequently, the BF data lend some support to a forth class according to geology, with brown hard waters, and we will use that class in our suggestion of typology.

Table 2. Distribution of Swedish lakes% within the national survey of lakes into classes of colour (absF420) and alkalinity

	Alkalinity				
	<0.2	0.2-0.4	0.4-1	>1	Sum
AbsF420					
<0.06	20	8	4	3	36
0.06-0.18	26	9	5	3	42
>0.18	16	4	2	1	22
Sum	62	21	11	6	100

River catchment area

Our data from the national survey only consists of small rivers with catchments between 10 and 250 km^2. According to this data, a class of rivers with catchments <100 km^2 seems to be justified (Figure 14). For larger catchments, extreme conditions, such as ANC >1 meq/l, Tot-P >20 µg/l and an ASPT index <5, hardly occurs. It is questionable whether there is relevant for as many as four classes according to catchment area. In Sweden only 10 rivers are >10 000 km^2, and it will probably not be possible to find reference stations to these rivers if they are impacted by human activities. If the largest class is merged with the class >1000 km^2, smaller rivers fulfilling the criteria for high ecological status can be used as references. Our suggestion is then 3 area classes: 10 - 100 km^2, 100 - 1000 km^2 and >1000 km^2.

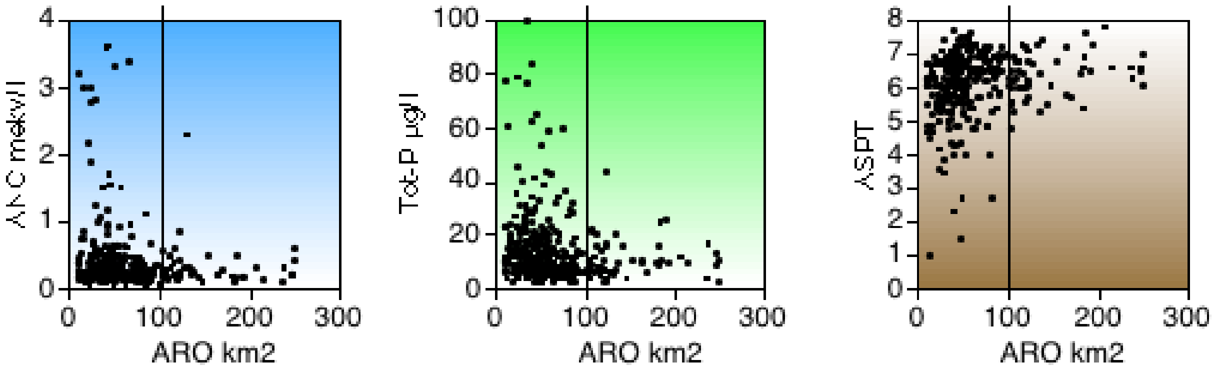


Figure 14. ANC, Tot-P and ASPT index of BF for non-polluted streams in the Swedish national survey 1995.

Lake size

Lake area

According to system A, only lakes >0.5 km^2 are included in the typology, which can be interpreted so that only lakes larger than 0.5 km^2 should be treated as single water bodies. In Sweden however, there are 93 575 lakes out of c. 100 000 between 0.02-0.5 km^2. These small lakes consists a great national resource and should be included in the work by the water district authorities, but we suggest that these lakes are not reported to the EU, and thus not are included in the common Nordic typology and work with reference conditions and intercalibration. For our evaluation of the typology this means that only 347 lakes with WC and 93 lakes with BF from the national survey can be used.

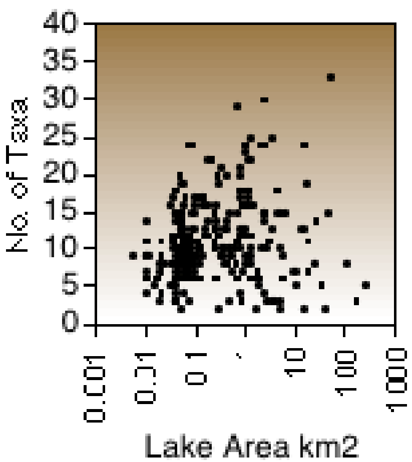


Figure 15 Number of Taxa of BF plotted by lake area in non-polluted Swedish lakes.

Lake area is an important ecological factor by many reasons. First it covariates with other factors such as turnover time and depth. Further, the number of species increases with size. In Figure 15 it is shown by the lake survey data that this relationship is not linear. It is only the probability of finding a high biodiversity that increases by area. Species rare communities are found also in larger lakes. It is also shown that the probability of finding a high species community only increases to a certain area, c. 10 km2. In system A, there are four types according to lake area with the borders 1, 10 and 100 km2. We question whether there is a need for so many area classes. Especially the largest class seems unmotivated based on the BF data, and since there are only 24 lakes >100 km2 in Sweden, it will be difficult to find reference stations and intercalibration sites for many types in the largest class according to system A. The range of the smallest size class also is narrow compared to the other classes. In the Finnish suggestion, three classes were suggested, with the borders 5 and 40 km2. By this classification however 87% of the Swedish lakes >0.5 km2 are in the smallest class, and only 1% in the largest class. Unless there are ecological reasons for setting the borders between the size classes, we suggest a more even distribution of the lakes between the classes than in the Finnish suggestion. Our suggestion is three size classes for lakes >0.5 km2 with the borders 2 and 10 km2. The distribution of Swedish lakes according to the different classification systems are shown in Table 3.

Table 3. Distribution of Swedish lakes according to three suggestions of classification according to lake area.

Lake area km^2	N	Lake area km^2	N	Lake area km^2	N
0.5-1	2 473	0.5-5	5 799	0.5-2	4 444
1-10	3 808	5-40	805	2-10	1 837
10-100	383	>40	84	>10	407
>100	24				

Lake depth

Lake depth is the most important factor controlling the mixing characteristics of lakes. Together with colour and turbidity it also defines whether there is a profoundal zone in the lake. Hence, the factor is important, but at the same time problematic to use for classification of lakes. It is strongly correlated to lake area, so there will be very few objects in types with small area and large depth and vice versa. We therefore suggest that instead of lakes being classified by depth they are classified according to whether they get thermally stratified during summer or not. If no data on temperature profiles are available, this can be modelled by data of lake area, lake depth, lake form, and topography around the lake. Although these data all can be estimated by using GIS applications, it is a large task to perform for all lakes, and so far we assume that lakes deeper than 3m get stratified. We thus suggest that lakes are classified into two depth classes – deeper or shallower than 3 m. It could be justified to have a larger depth class, but there are to few objects in our database to test whether lakes >3m in the same area class can be separated into groups by depth. For example there were only three lakes between 0.5km^2 and 2km^2 deeper than 15m.

One big problem for classifying lakes according to depth is that there are only scarce data on lake depth. Mean depth can be roughly estimated by maximum depth, but still it is only known for about 1/3 of Swedish lakes >0.5 km^2.

Other factors

Lake turnover

Lake turnover time is important for the nutrient status of the lake. With a short turnover time, the concentrations of e.g. Tot-P and AbsF420 will be close to the mean concentration in the inflow to the lake. When the residence time in the lake gets longer, sedimentation decreases the content of nutrients and organic matter. If we plot those variables against turnover time, however, we don't see any such relationship (Figure 16). Other factors seems to have a greater influence of those variables, although we only used data from the Fenno-Scandian shield, to keep the variation low. We perhaps would have seen a relationship with turnover time if we for example only plotted lakes within one size class in one ecoregion, but since lake depth only is known for a limited number of lakes, this could not be done.

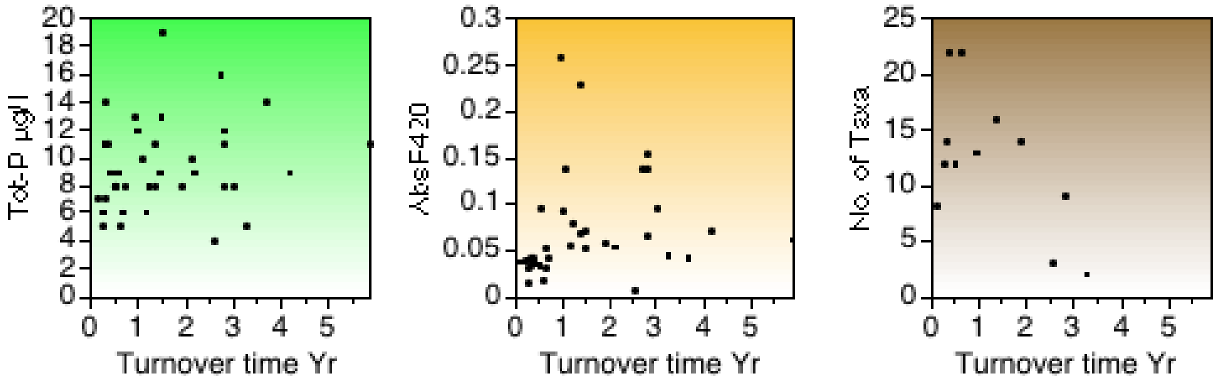


Figure 16. Tot-P, AbsF420 and No. of taxa plotted against lake turnover time in lakes in the Fenno-Scandian shield in Sweden.

Lake area in catchment

For the same reason that a long turnover time in a lake is expected to give a clearer water due to sedimentation, lakes with a high water surface area in the catchment are expected to have low water colour and nutrient content. This is clearly seen in Figure 17, where the range of Tot-P and $AbsF_{420/5}$ declines towards low values as the % water surface in the catchment increases. This could justify a classification by presence of lakes in the catchment, but much of this variation is covered by the classification by water colour so we don't think this would improve the typology.

Another function of lakes within the catchment is to serve as a regional species pool. Hence the biodiversity is expected to increase by increasing presence of lakes in the catchment. This was however not seen in the BF data (Figure 17). On the contrary, the most species rich communities were found in lakes with a low fraction of water surface in the catchment. As a conclusion we don't suggest any classification by lake area within the catchment in a Swedish typology.

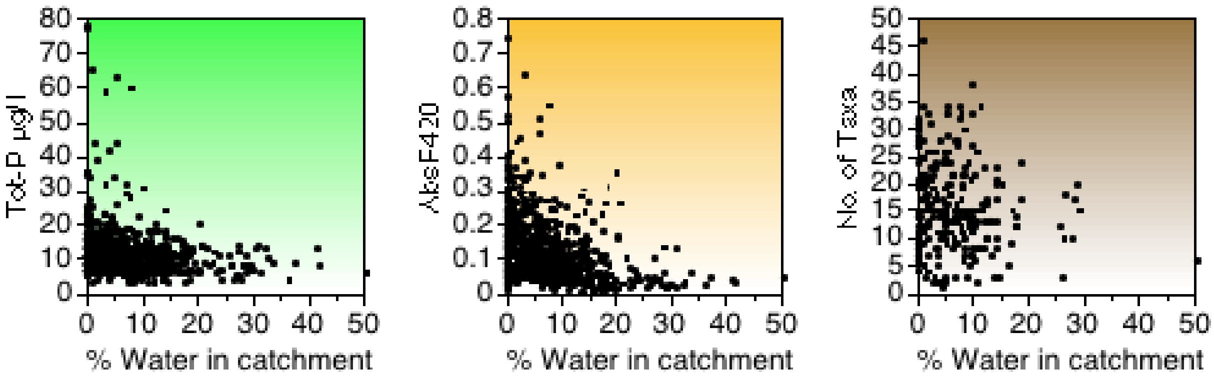


Figure 17. Tot-P, AbsF420 and No. of taxa plotted against % of water surface int the catchment for lakes in the Fenno-Scandian shield in Sweden.

Stream velocity and slope in streams.

Stream velocity is of high importance for the status of a riverine ecosystem. The flow rate determines for example the type of bottom substrate, the nutrient supply for plants and epifyton and the food supply for filter feeders. However, the flow rate varies over time and it is not feasible to measure the flow rate for each river segment. Instead the slope of the river can be used for classification of rivers, since the slope is highly correlated to the flow rate. Slope, however, has to be clearly defined. Is it the slope of the sampled site or the slope of the catchment that is of interest? At present we don't have any data on stream slope, so we can't include that variable in the typology.

A suggestion to a typology for Swedish surface waters.

Lakes

Our primary suggestion to a typology for lakes includes 4 ecoregions, 2 depth classes, 3 lake area classes and 4 geochemical classes (Table 4). At this stage, we only include lakes >0,5 km^2 for practical reasons. The suggestion includes 96 theoretical lake types, which can be compared to 324 theoretical types according to System A. A class of lakes <0.5 km^2 will add another 32 theoretical types if added to the system. If southern Sweden is divided into three regions, one high elevated, one low elevated south-eastern region and one low elevated south western region, the typology will include 144 theoretical types for lakes >0.5 km^2. Our primary suggestion is however that southern Sweden is one region. For many of the types there will be no, or very few objects, e.g. large brown lakes in the Borealic uplands. To sort out types with no or few objects, we applied the typology to all Swedish lakes >0.5 km^2 in the Swedish national register of lakes (SMHI, 1996). The distribution between geological classes within each class defined by ecoregion, depth and size, is applied from the national survey 1995.

Table 4. A suggestion to a Swedish typology for lakes according to the WFD.

Ecoregion/altitude	Lake Mean Depth	Lake Area	Geology
Borealic Upland	< 3 m	<0.5 km^2	Siliceous
Fenno-Scandian Shield > 200 m.a.s.	> 3 m	0.5 - 2 km^2	Organic ($AbsF_{420/5}$ > 0.06)
Fenno-Scandian Shield < 200 m.a.s.		2 - 10 km^2	Calcareous (Ca > 0.5 meq/l)

Central Plain	> 10 km²	Mixed calcareous and organic

If types with <10 objects are removed from the typology, 47 types remains (Table 5). The distribution of lakes in the different types is shown in Table 5. The number of lakes is only a rough estimation where the largest error probably is in the distribution into depth classes. For most lakes, the lake mean depth is not known and depth is estimated from lake area and geographical location (SMHI; 1996). This will probably mean that the number of lakes with unusual relation between area and depth, such as large shallow and small deep lakes, will be underestimated.

Table 5. Number of lakes within each type in a suggestion to a typology for lakes according to WFD.

Ecoregion	Depth	Lake Area	Geology			
			Silceous	Org.	Calc.	Org. + Calc.
Borealic uplands	Shallow	S	17	17		
		M				
		L				
	Deep	S	910	173	63	16
		M	354	52	26	
		L	133			
Fenn-Scand. >200m	Shallow	S	19	42		
		M				
		L				
	Deep	S	373	808	47	31
		M	202	192	67	10
		L	58			
Fenn-Sand. ≤200m	Shallow	S	37	149		
		M		14		
		L				
	Deep	S	293	339	23	23
		M	163	227		
		L	79			
Central Plain	Shallow	S	28	168	84	112
		M			23	79
		L				
	Deep	S	174	349	87	174
		M	94	57	94	57
		L	28		71	

Rivers

For streams we suggest a typology based on the same ecoregions as for lakes, with 4 or 6 ecoregions. Further we distribute the rivers into 3 size classes, and 4 geological classes (Table 6). If we use 4 ecoregions, we get 48 theoretical classes. The national registers of rivers and catchments are at present not in the state that the frequency of of objects for the types can be estimated.

Table 6 A suggestion to a Swedish typology for rivers according to the WFD.

Ecoregion/altitude	Catcment Area km2	Geology
Borealic Upland	10 - 100 km^2	Siliceous
Fenno-Scandian Shield > 200 m.a.s.	100 - 1000 km^2	Organic ($AbsF_{420/5} > 0.05$)
Fenno-Scandian Shield < 200 m.a.s.	> 1000 km^2	Calcareous (Ca > 0.5 meq/l)
Central Plain		Mixed calcareous and organic

In the national surveys, only rivers between 10 and 250 km^2 were included. If we limit the evaluation to the smallest size class: 10 - 100 km^2, and use the water chemistry from RI95 we get the distribution of the types as presented in Table 7. Since the spatial variation in water chemistry decreases by catchment area, it is not possible to generalise the data from the national surveys to larger size classes.

Table 7. Non-polluted rivers in the national survey 1995 distributed into types according to a preliminary suggestion to a typology.

Ecoregion	Catch. Area	Geology	Number of streams
Borealic uplands	10 - 100 km2	Siliceous	59
		Organic	21
		Calcareous	5
Fenn-Scand. >200m	10 - 100 km2	Siliceous	16
		Organic	43
		Calcareous	4
		Organic and Calcareous	4
Fenn-Sand. ≤200m	10 - 100 km2	Siliceous	5
		Organic	56
		Calcareous	1
		Organic and Calcareous	5
Central Plain	10 - 100 km2	Siliceous	3
		Organic	13
		Calcareous	1
		Organic and Calcareous	27

Unusual lake types

Types within the system with few objects

In our suggestion to a typology for lakes, there are 27 theoretical types with no objects and 12 types with less than 10 objects (Table 8). We suggest that it is not justified to set up a network of reference stations for such small groups, and therefore they are excluded from the typology. The limit of <10 lakes could be thought to be to high. If we instead set the limit of exclusion to <6, we include 3 more types, all with lake areas >10km^2. As we mentioned above, lakes with more unusual relationships between area and depth are probably underestimated, so that perhaps some of the unusual classes with large shallow lakes actually contains more objects, and should be included in the typology.

The fact that these lakes types contains few object does not mean that they are not valuable. On the contrary, just because they are so rare, they should get special attention when they are found. However, the number of lakes are just estimations, and we don't know where these lakes are situated, so it will be extremely difficult to find representative reference stations to set up reference conditions, so it is not possible to include them in the typology

Table 8. Rare types of lakes with <10 objects within a suggestion of typology for Swedish lakes within the WFD.

Ecoregion	Depth	Lake Area	Geology			
			Silceous	Organic	Calcaerous	Org. + Calc.
Borealic uplands	Shallow	S				
		M	2			
		L	1			
	Deep	S				
		M				
		L				
Fenn-Scand. >200m	Shallow	S			2	2
		M	5	5	2	
		L				
	Deep	S				
		M				
		L		9	9	5
Fenn-Sand. ≤200m	Shallow	S				
		M				
		L				
	Deep	S				
		M				
		L				
Central Plain	Shallow	S				
		M				
		L			2	9
	Deep	S				
		M				
		L				

Large lakes

The rationale behind the typology is that each type shall combine a number of similar lakes into one group, where a smaller number of lakes not affected by human impact shall define the reference conditions for that type. The largest lakes, as Lake Vänern, Lake Vättern and Lake Mälaren, would then be compared to other lakes >10 km^2, when determining the ecological status. However, those large lakes could also be seen as unique objects, often with a large natural within-lake variation, and it may not be relevant to compare them to smaller reference lakes. The large lakes are already subjects to intensive monitoring programs and research. We thus suggest that they should be treated as single objects, with reference conditions set by expert judgement. An important issue is whether there are more large lakes than the three mentioned, which should be treated in a similar way.

Unusual types not covered by the typology

Besides the unusual types described in the typology system and the large lakes, there might be unusual lake types not defined by the typology. For example lakes with unusual morphogenesis, such as kettle lakes and tectonic lakes, might have characteristics that are not covered by the typology. We suggest that such unusual lake types are paid special attention in the management plans.

Evaluation of the typology for lakes for setting reference conditions.

The demands from the WFD on the typology are high. For each type, the reference conditions should be defined by waterbodies in that type that are not affected by any human activities. This requires that the metrics of the ecological quality elements for reference objects within a type are well defined without a too large variation. The data from the national surveys of lakes then offers a great possibility to test whether the typology produces well defined groups of lakes with a small variation in metrics of benthic fauna and water chemistry. In Figures 18 and 19 we present how Tot-P and the Medins index for benthic fauna varied within the types of our suggested typology for lakes. We have only used data from sites not affected by eutrophication, acidification and liming according to the criteria presented above. To diminish the influence of between year variation, we have used the average from the two surveys in 1995 and 2000 for each lake.

The results show a large variation within the types. For deep, small, organic lakes in the Fennoscandian shield >200m altitude for example, Medins index varies between 2 and 9, and Tot-P varies between 5 and 16 µg/l. This large variation raises questions as to whether it is possible to use type specific reference values as a basis for calculating ecological quality ratios. If the reference value for Tot-P for this specific type is set to 10 µg/l, and the limit between high and moderate status is set to two times the reference value, it means that the lakes with the lowest natural concentration of Tot-P have to be elevated fourfold before the Water Frame Directive prescribes measures for improving the water quality. On the other hand, for a lake with a natural background level of 16 µg/l within this type, it will be difficult to achieve good status. However, one reason for the large within-type variation could be that our criteria for the exclusion of polluted sites was not appropriate, and that some of the lakes really are affected by anthropogenic eutrophication and acidification. While this could be the case for a few lakes in southern Sweden, it is not likely in northern Sweden > 200m. In this region the observed variation has to reflect the natural variation. A part of this variation might be due to a high non-synchronous between-year variation, and that a mean of autumn values from two years was not sufficient for a representative average of the lake quality. To investigate this we used data from the national monitoring program of reference lakes and used 5 year means taken four samples/year. Since the number of reference lakes is limited we only present data for four common types (Table 9). We excluded lakes affected by eutrophication and acidification. Although the within-type variation of Tot-P decreased, the 90 percentile is still twice as high as the 10 percentile.

For Medins index values below 6 are regarded as affected by acidification. Thus, it is somewhat surprising that most values are below that limit for non-acidified lakes. Medins index is however developed for streams in southern Sweden, and the application of the index to litoral fauna in all of Sweden might be doubtful. Still, the large variation in the index indicates that there will be difficulties in using type specific reference values for estimation the ecological quality.

Figure 18. Distribution of Medins bottom fauna index in non-polluted lakes from the national survey of lakes in Sweden. Mean values of the two surveys 1995 and 2000 are presented for lakes separated according to a suggestion to a typology for lakes.

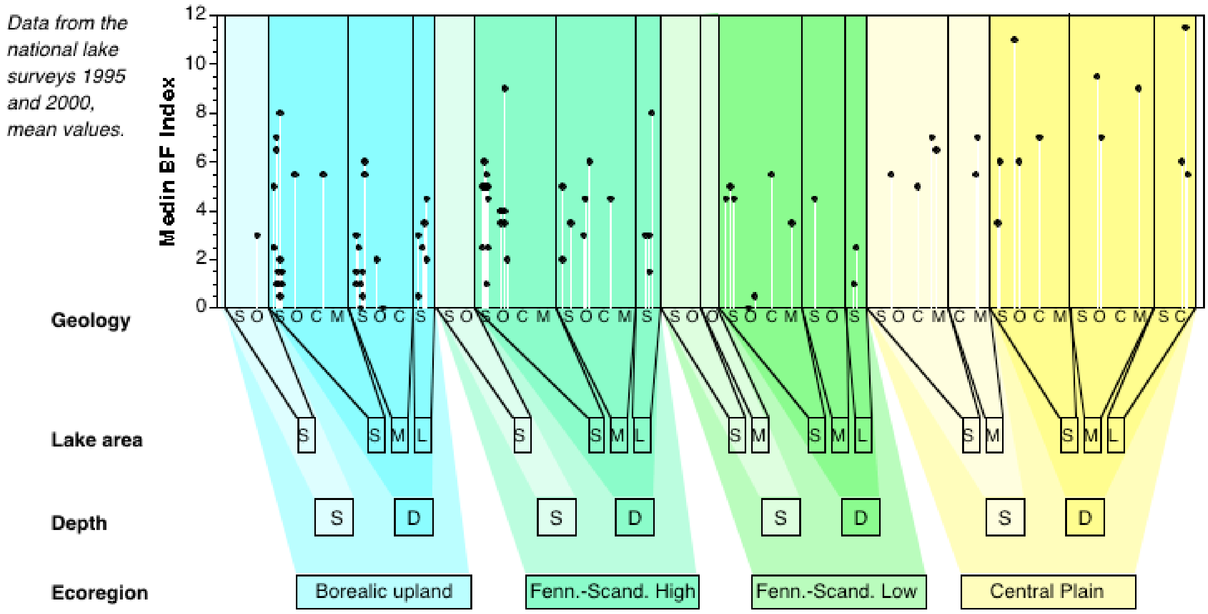


Figure 19. Distribution of Tot-P in non-polluted lakes from the national survey of lakes in Sweden. Mean values of the two surveys 1995 and 2000 are presented for lakes separated according to a suggestion to a typology for lakes.

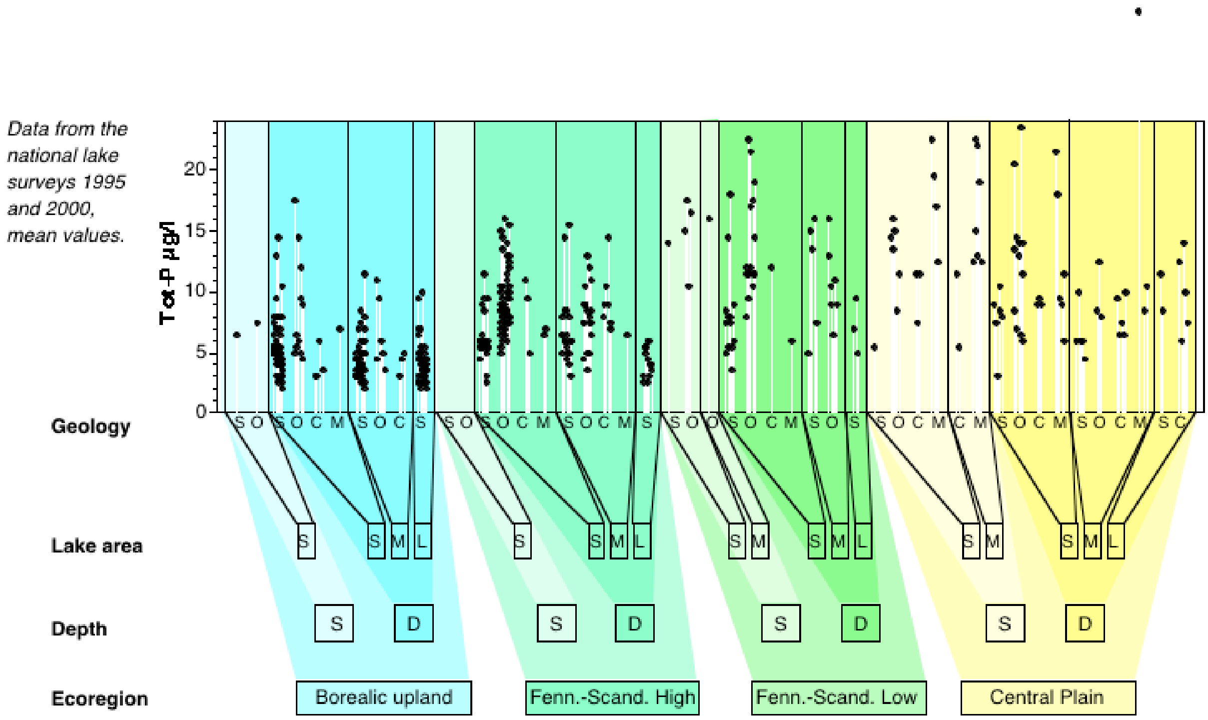


Table 9. Tot-P in unpolluted reference lakes from for types according to a suggestion to a typology according to WFD. 10 and 90 percentiles of Tot-P concentrations are given in the table.

Ecoregion	Depth	Lake Area	Geology	No. of lakes	Tot-P µg/l	
					10 perc.	90 perc.
Fenn.Sc. >200m	>3m	0.5-2 km2	Organic	13	6	11
Fenn.Sc. <200m	>3m	0.5-2 km2	Organic	7	6	12
Centr. Plain.	>3m	0.5-2 km2	Siliceous	6	5	11
Centr. Plain.	>3m	0.5-2 km2	Organic	12	7	16

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