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EU reference conditions in Swedish lakes identified with diatoms as palaeoindicators — a review

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This study aims to assess the reference conditions and reference diatom communities for Swedish lakes by summarizing 36 performed palaeolimnological studies. The following results can serve as a tool in the discussion about reference conditions according to the European Water Framework Directive. (1) Finding one general fixed time of prevailing reference conditions for all Swedish lakes is impossible. (2) 1850 AD, as an often applied fixed reference time, is not an appropriate reference time for all Swedish lakes. (3) Reference nutrient and pH conditions for many southern Swedish lakes were prevailing until about 2300 BP. (4) Reference nutrient and pH conditions for many northern Swedish lakes prevailed longer and may even prevail today. (5) Different reference diatom communities can be found: either benthic (both acidophilous and alkaliphilous) or planktonic communities.

Introduction

Surface waters are of great importance for humans as they supply drinking water, fishery, transport and recreation (Wetzel 2001, Smol 2008). Severe deterioration of water quality in densely-populated areas during the 1950s and 1960s resulted in increasing limitations in the use of surface freshwaters (Schindler 2006, Smol 2008).

Regulations had to be implemented to mitigate the deteriorating effects of e.g. intensification of agriculture and increasing industrialization on aquatic ecosystems. The primary regulation is the European Union Water Framework Directive (EU WFD, EU 2000). It aims to prevent further deterioration, protect and improve the status of aquatic ecosystems and to promote the sustainable use of water. All member states are obliged to implement all necessary measures to achieve at least 'good' water status for relevant waterbodies by 2015. 'Good' status is defined as conditions deviating only slightly from undisturbed conditions, with low levels of anthropogenic distortion. Ecological status class (high, good, moderate, poor or bad) is assigned, based on the deviation of present-day conditions of different quality elements from reference conditions. The EU WFD focuses on the ecological rather than the hydrochemical water quality (Bennion and Battarbee 2007) in contrast to former water quality regulations. Biological properties (comprising phytoplankton, phytobenthos and macrophytes, invertebrate fauna and fish fauna) are the most important quality elements requiring evaluation, followed by hydromorphology, chemistry and physico-chemistry of the water body as elements supporting the biology (EU 2000).

According to the EU WFD, reference conditions are defined as conditions with no, or only slight, human-induced changes and distortion. Many scientists interpreted the term ‘reference conditions’ according to the EU WFD as ‘conditions without significant human impact’ or ‘background conditions with no, or minimal, anthropogenic stress’ (Bennion *et al.* 2004, Räsänen *et al.* 2006, Taylor *et al.* 2006). This interpretation is also used in this review.

The definition of the reference state is often considered a major practical challenge in implementing the EU WFD (Søndergaard *et al.* 2005, Bennion and Battarbee 2007), as most European surface waters are influenced by humans and undisturbed sites are difficult to find (Bennion and Battarbee 2007).

Type-specific reference conditions need to be established by all member states in order to develop sustainable water management and conservation strategies, and to provide realistic targets for restoration (EU 2000). The first step to determine type-specific reference conditions is the classification of all surface waters. The Swedish Environmental Protection Agency (SEPA) suggested the following lake classification according to which, Swedish lakes need to be assigned to one of the seven ecoregions:

1. mountains above tree-limit,
2. Norrlands inland, below tree-limit, above highest coastlines,
3. Norrlands coast, below highest coastline,
4. Southeastern Sweden, south of Norrlands border, below 200 m a.s.l.,
5. South Sweden, Skåne, Blekinges coast and parts of Öland,
6. Southwestern Sweden, south of Norrlands border, below 200 m a.s.l.,
7. South-Swedish highlands, south of Norrlands border, above 200 m a.s.l.

Furthermore, lakes are differentiated by mean or maximal depth, surface area, humus concentrations and alkalinity (Naturvårdsverket 2007).

Type-specific reference conditions have to be determined based on modelling (predictive modelling or hindcasting methods) or spatially based (by establishing a reference network of sites with high status for every surface water body type) or based on a combination of both approaches. If these methods cannot be applied expert judgement can be used for determination of reference conditions (EU 2000).

Palaeolimnology is a hindcasting method applied for modelling past environmental conditions. It is a valuable tool for gaining insight into the history of surface waters, especially the history of lakes (Smol 2008). Most recent palaeolimnological studies are multi-proxy studies (Birks and Birks 2006); sediment cores are analysed for diatoms, chironomids, cladocerans, pollen, sediment chemistry, chrysophytes, pigments, carbonaceous particles and other proxies, to get a detailed picture about historic conditions and factors causing these.

The analysis of fossil diatoms is a powerful method in palaeolimnology (Battarbee *et al.* 2001). Diatoms, as part of the phytobenthos and phytoplankton, live attached to moist or submerged surfaces or in the open water. Diatoms are one of the most dominant microalgae groups in freshwaters and are responsible for a great deal of the total primary production of aquatic ecosystems. They can be used as indicators for detrimental effects as acidification (Smol *et al.* 1986 and references within, Battarbee *et al.* 2001, 2010) and eutrophication (Battarbee *et al.* 2001, Hall and Smol 2010), among others.

Diatoms are regarded as ideal indicator organisms for several reasons. There is a high number of ecologically sensitive species with well known ecology. They preserve well because their cell wall is impregnated with silica ($SiO_2 \times nH_2O$), they can be easily identified by size, shape and sculpture of cell walls, and they are highly abundant in almost all environments with at least occasional presence of water (Round *et al.* 1999).

Structural and functional characteristics of pre-pollution aquatic ecosystems can be reconstructed by analysing a sediment core for fossil, sedimented diatoms as part of the lake flora (Battarbee *et al.* 2001). Inference of past and reference pH and nutrient concentrations of lakes based on historic diatom community composition have become a standardised and often applied tool (Renberg *et al.* 1993a, 1993b, Bradshaw and Anderson 2001, Renberg *et al.* 2001, Bennion *et al*. 2004, Miettinen *et al*. 2005, Leira *et al*. 2006, Hübener *et al*. 2009, Kirilova *et al*. 2009). Past environmental conditions can be derived by transfer functions, mostly based on weighted-averaging regression and calibration (WA) techniques (ter Braak and van Dam 1989, Birks *et al*. 1990) and the extended weighted-averaging partial least square (WA-PLS) techniques (ter Braak and Juggins 1993). These transfer functions are derived from a training set of modern surface-sediment or littoral diatom assemblages and contemporary environmental data, to quantitatively estimate past environmental conditions. For diatom-based inference of TP values in Sweden, there is a training set containing 45 southern Swedish lakes (Bradshaw and Anderson 2001). Some studies also applied a calibration set containing lakes from Sweden, Danmark, UK and Ireland (Rydberg *et al*. 2006a, 2006b, Bigler *et al*. 2008). For diatom-based inference of pH there are also Swedish training sets, containing 118 northern Swedish lakes (Korsman and Birks 1996) or 100 northern Swedish lakes (Bigler and Hall 2002), while pH inference in southern Swedish lakes is often based on the Surface Waters Acidification Project Palaeolimnology (SWAP) diatom calibration data-set containing lakes in the UK, Norway and Sweden (Stevenson *et al*. 1991).

In Sweden, the anthropogenic contribution to acidification of lakes is assessed with the hydrogeochemical dynamic model MAGIC (Cosby 1985, Moldan 2004). The anthropogenic contribution is the deviation from reference conditions in 1860 AD, caused by deposition of sulfur and nitrogen and modern forestry practices. Prediction of the pre-industrial pH of 55 Swedish lakes was carried out with the two different approaches, both predictive modelling and palaeolimnology in two independent investigations. These gave similar reconstructed pre-industrial pH values and Erlandsson *et al*. (2008) outlined the consistency between these two different applications. The anthropogenic contribution to eutrophication of Swedish lakes is classified by modelling, applying a static hydrogeochemical model (Wilander 2004).

Despite the fact that quite a few palaeolimnological studies have been performed in Swedish lakes, uncertainty about reference conditions remains. One of the paramount questions is how far back in time did these reference conditions, according to the EU WFD, prevail in Swedish lakes? Many of these palaeolimnological studies were performed in Swedish lakes before implementation of the EU WFD but results can still help to answer questions connected to the EU WFD. Furthermore these studies could indicate reference values for pH, nutrient concentrations or reference diatom communities.

Another problem is the definition of type-specific reference conditions. Reference or natural conditions according to EU WFD are conditions without significant human influence, but it is unclear when anthropogenic impact became significant. As hindcasting methods, like palaeolimnology, enable the reconstruction of human history in the vicinity of a lake they can help with decisions of regarding when human influence became significant, and where to set the time point equaling reference conditions (Bennion and Battarbee 2007).

The MAGIC model, applied for calculation of acidity reference conditions in Swedish lakes, adopts 1860 AD as a time point of reference conditions. Generally, many researchers apply the time around 1850 AD (Bennion *et al*. 2004, Leira *et al*. 2006, Bjerring *et al*. 2008), as the time of prevailing reference conditions immediately before the onset of industrialization and intensive agriculture. However, the question remains as to whether there were any significant human-induced changes before this industrialization period. Did industrialization begin shortly after 1850 AD in all regions? Lakes are dynamic ecosystems, constantly changing over time (days, years, centuries or even millennia) even without significant human influence. Which of the different states in lake development would then be the natural state? Therefore, it is questionable whether the time around 1850 AD really represents reference conditions in all Swedish lakes.

This review evaluates the results of palaeolimnological studies with diatoms as palaeoindicators in Swedish lakes (Fig. 1) to define a point in time equaling natural or reference conditions, and will gather information about water quality status and diatom succession during lake development. Only the knowledge of reference

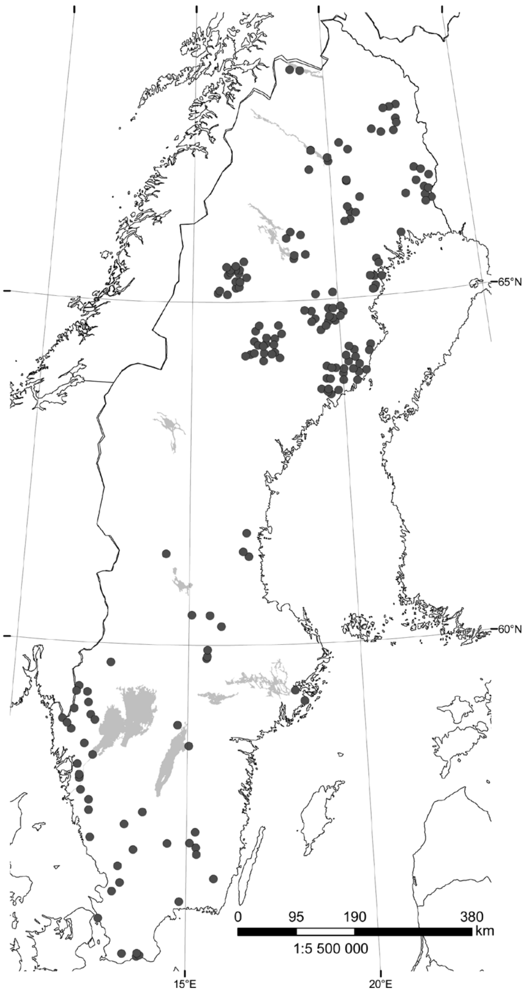

Fig. 1. Map of Sweden indicating the lakes included in the reviewed palaeolimnological studies.

water chemistry and reference diatom communities allows the development of new indices to calculate any significant deviation from natural conditions. This review is the first summary of the results of palaeolimnological studies in Swedish lakes.

Reference conditions

History of Swedish lakes reconstructed based on diatoms

In order to answer the question of when reference conditions prevailed in Swedish lakes, the natural development of lakes and the onset and severeness of anthropogenic influence on the ecology and water quality of lakes need to be studied.

Palaeolimnological studies have shown that all lakes undergo changes over time, a process called aging (Renberg *et al*. 1993b, Engstrom *et al*. 2000). Development of lakes in Sweden started with the last deglaciation beginning about 12 500 BP. Individual lakes developed differently due to natural variability in catchment geology, pedology, hydrology, vegetation and anthropogenic influences. Soil type in the lake catchment area is regarded as the most important factor affecting lake succession (Renberg *et al*. 1993b).

Natural acidification and oligotrophication period (12 500 BP–ca. 2300 BP)

Swedish lakes having thick soils and large base-cation storage in their catchments showed a quite constant pH during the whole postglacial period (Renberg *et al*. 1993b).

However, many lakes in Sweden have catchments with weathering-resistant granitic bedrock covered by thin, poor soils. They have been undergoing a natural, long-term oligotrophication and acidification, typical for lakes with such catchments (Engstrom *et al*. 2000). Originally, soils were rich in exchangeable base-cations and nutrients, but accumulation of humus and the uptake of base cations and nutrients during vegetation growth over time resulted in a decreasing transport of base cations and nutrients and increasing transport of organic acids to the waters. Diatom-inferred pH (dipH) of clear-water lakes dropped from about 6.5–7.5 to 5.2–5.6 in southern Swedish lakes (Renberg *et al*. 1993b); pH in northern Swedish lakes dropped in a similar range: from 6.9 to 5.6 in Makkasjön (Korsman and Segerström 1998), from 7.2 to 6.7 in Vuoskkujávri (Bigler *et al*. 2002) and from 7.2 to 6.5 in Lake Njulla (Bigler *et al*. 2003). Despite this general trend, considerable short-term variations in dipH (up to 0.5 pH units in one century) could be observed by Renberg (1993b), when increasing sediment sample frequency. In a sediment core of Sämbosjön, southwestern Sweden, diatom assemblages indicative of relative high nutrient concentrations were found since lake development about 11 000 BP, followed by diatoms indicating decreasing nutrient concentrations and pH from about 8000 BP–6000 BP (Digerfeldt and Håkansson 1993).

Anthropogenic alkalization and eutrophication period (ca. 2300 BP–1900 AD)

In many southern Swedish acid-sensitive lakes, the period of natural acidification and eutrophication was followed by the anthropogenic alkalization and eutrophication phase. The dipH increased due to agricultural expansion including grazing of free-ranging livestock, burning of forest to improve grazing and other culture-related practices (Renberg *et al*. 1993a, 1993b). These practices released base cations and nutrients from the forest biomass and upper soils and enhanced the transport of base cations and nutrients to the surface waters. This transport resulted in increases in dipH of about one unit from 5.2–5.6 up to 6.0–6.8 and a slight nutrient increase. Pre-industrial sulfur-deposition arising from early mining and metal industries also contributed to the alkalization and eutrophication from 1000 AD onwards (Bindler *et al*. 2002). The authors suggest several mechanisms. First, the excess sulfur increased base-cation transport to the lakes by increasing cation exchange in catchment soils. Second, the sulfur deposition altered the iron-phosphorus cycle in the lakes by formation of iron sulfide, which subsequently resulted in a release of phosphorus. Last, the emission of base cations during ore processing is suggested to contribute to the alkalization.

Additionally, Renberg and Hultberg (1992) and Renberg *et al*. (1985) proposed a short 'transitory eutrophication period' between 1700 AD and 1900 AD. Suggested causes for the eutrophication are increased fall-out of nitrogen and phosphorus from the atmosphere and inflow of nutrients and base cations from the catchment soils during the initial phase of increased acidity of precipitation.

Signs of elevated nutrient concentrations could be observed in several southern Swedish lakes. Diatom assemblages, indicative of high nutrient concentrations, were found throughout a sediment core of Havgårdssjön, covering 700 years BP (Dearing *et al*. 1987), from about 2500 BP onwards in Bussjösjön (Håkansson and Regnéll 1993) and from about 4000 BP onwards in Sämbosjön (Digerfeldt and Håkansson 1993). The indicated high nutrient concentrations in all three lakes are ascribed to human impact in the catchment.

No signs of alkalization could be observed in northern Swedish Vuoskkujávri (Bigler *et al*. 2002). Other northern Swedish lakes, with sparse human activity, underwent a natural alkalization period from 3000 BP–1000 BP. Diatom-inferred pH of Makkasjön increased up to 6.0 during that period (Korsman and Segerström 1998). The natural alkalization is explained by frequent lightning-induced wildfires and reduced mire expansion due to drier climate, resulting in decreasing transport of humic acids to the lakes. A fire-induced rise in pH was also observed in Finish lakes by Korhola *et al*. (1996). The mechanisms are not clear; the authors suggested mixing of the water column and increasing inputs of erosional material and ashes.

Modern acidification and eutrophication period (1900 AD–present)

The period of modern acidification and eutrophication is characterized by rapid, severe acidification and eutrophication of surface waters.

Severe acidification is due to high deposition of acidifying substances, resulting in decreasing pH down a pH to 4.5–5 in many Swedish lakes. Southern Swedish lakes with granitic bedrock covered by thin, poor soils in their catchments were worst affected by this acidification process. In Lilla Öresjön, slow deterioration of water quality with decreasing pH had already begun by about 1900 AD, while acute modern acidification could be observed around 1960 AD (Renberg 1990, Renberg *et al*. 1990). Acute acidification peaked around 1960 AD in many acid-sensitive lakes in southern Sweden (S Sweden), such as Lake Lysevatten (Renberg and Hultberg 1992), Lake Örvattnet (Ek *et al*. 1995), lakes Härsvatten, Stora Tresticklan, Hjärtsjön, Övre Skärsjön, Storasjön and Harasjön (Ek and Korsman 2001).

Southern Swedish lakes with easily weathered bedrock and thick soils in their catchments were less affected by modern acidification. Rydberg *et al*. (2006a) studied the pH history of six lakes with different bedrock and soils in the Dalslands Kanals watersystem. They found that only lakes with weathering-resistant bedrock and thin soils showed slight signs of acidification, while lakes within the same watersystem, but on weathering bedrock with thick soils showed no sign of decreasing pH despite high sulfur emission and deposition levels.

Due to the gradient in sulfur deposition, decreasing from south to north (Bernes 1991), severe acidification is a problem mostly affecting lakes in S Sweden and a study conducted on 118 northern Swedish lakes did not show modern, large-scale acidification in the north (Korsman 1999).

Besides sulfur deposition patterns, bedrock and soils, the content of total organic carbon (TOC) influences the severeness of acidification. Organic acids, as one fraction of TOC, do not only have the ability to acidify surface waters but also to buffer against acidification caused by mineral acids (Hruška *et al*. 1999). Ek and Korsman (2001) found smaller decreases in dipH in lakes with high TOC concentrations (> 9 mg l^{-1}) as compared with those in lakes with low TOC concentrations (< 7 mg l^{-1}) after being exposed to similar loads of acid deposition.

There are also lakes which underwent a different development, for example lakes along the Gulf of Bothnia coast. They are about 200–300 years old, formed by isostatic land lift of about 1 cm per year. They are located just a few meters above sea-level, and their catchment is dominated by fine-grained marine sediments, with very high sulphur concentrations. For example, Blåmissusjön experienced severe acidification in the 1940s, with dipH decreasing from 6.0–7.0 to 3.0 (Renberg 1986). This decrease could not be connected to acid deposition, but rather to oxidation of the sulfur-rich sediments leading to acidic runoff. This process was caused by lowering of the water table or ditching in the catchments in the course of intensive agriculture.

Anthropogenic-induced eutrophication of surface waters during the 20th century was mainly due to nutrient loading from untreated human sewage, changing land use and urbanization (Schindler 2006). The eutrophication process started earlier in many Swedish lakes, but this period is characterized by severe nutrient loading with peaking nutrient concentrations. In a study of sediment cores from the two largest Swedish lakes, Vättern and Vänern, diatom-inferred total phosphorus concentrations (diTP) was shown to peak around 1960/1970 AD in both lakes (Renberg *et al*. 2003). The sediment record of the nutrient-rich Nedre Milsbosjön shows increasing diTP from 1960 AD and peaking diTP about 1990 AD, due to intensive livestock breeding (Rydberg *et al*. 2006b). Diatom-inferred total phosphorus concentrations of Florsjön and Östersjön were highest before 1900 AD, due to human impact and are now decreasing (Bigler *et al*. 2008). Bradshaw and Anderson (2001) found rapid eutrophication in the Ekoln basin of Lake Mälaren in the 1960s, with diTP raising from 50–60 μg TP l^{-1} prior to 1900 AD up to 99 μg TP l^{-1} in 1966 AD and a slow recovery afterwards. They claim that mesotrophic conditions were found long before 1900 AD, as indicated by the diatom flora. Renberg *et al*. (2001) also suggested that eutrophication and pollution of Lake Mälaren started with the formation of the lake in medieval times due to extensive agriculture and mining. They observed two major shifts in diatom community composition in the Södra Björkfjärden basin of Lake Mälaren. The first shift is assigned to increased silica concentrations, while the second major species shift suggests increasing phosphorus concentrations. The diTP concentration was quite stable at about 40 μg TP l^{-1} until it started to increase in 1850 AD and phosphorus concentrations peaked around 1950 AD (70 μg TP l^{-1}). A slight decrease in diTP of about 10–20 μg TP l^{-1} could be observed in recent decades. Comparison between diatom-inferred and measured phosphorus concentrations suggest an overestimation of about 20–30 μg TP l^{-1} in the TP model, however, the trend of eutrophication is still obvious, as is the timing of changes. The authors suggest a background level of 10–20 μg TP l^{-1}. If this suggestion is correct, the predictions of pre-industrial TP concentrations (about 6 μg TP l^{-1}; Persson *et al*. 1990) by catchment-lake modelling underestimate the real situation.

Liming period

Renberg *et al*. (1993b) proposed one further period for limed lakes as indicated by changes of the fossil diatom flora: the liming period (starting in the 1970s/1980s). Recovery from acidification after reduction of sulfur emission and deposition is very slow in acid-sensitive areas with low weathering rates. When studying the diatom community of ten acidified, unlimed lakes in S Sweden, minor signs of recovery could be observed in diatom community assemblage of only one lake, Lake Örvattnet (Ek and Korsman 2001). The diatom communities of the other nine lakes did not show any recovery, indicated by shifts in abundance of species or even a shift in species composition.

Liming of surface waters was suggested to be the only method to counteract severe acidification (Skeffington and Brown 1992). Extensive liming of Swedish surface waters started in 1977 AD and is still practiced.

For northern Swedish lakes Korsman and Segerström (1998) proposed a liming/fertilization period. It is also characterized by increased pH values due to anthropogenic impacts like liming and fertilization in the course of fishery management.

Problems in determining reference conditions

Determination of reference conditions

Many palaeolimnological studies were conducted in Europe, trying to find time points in history with prevailing reference conditions. Generally the definition of reference conditions according to the EU WFD (maximal slight human induced changes, EU 2000) is difficult.

1850 AD was adopted by many researchers as a fixed time of reference conditions to simplify the evaluation of ecological status of lakes (Bennion *et al*. 2004, Leira *et al*. 2006). It represents the time immediately before the onset of the industrialization and intensive agriculture. While a fixed reference time in the past would be useful, it does not take the different history of human influence in different countries, regions or even at different lakes into consideration.

In regard to human history in Sweden, it is clear that 1850 AD is not an appropriate reference time for all Swedish lakes. Many southern Swedish lakes were already significantly impacted by human settlement, agriculture and mining in pre-industrial times about 2300 BP–1000 BP (Renberg *et al*. 1993a, 1993b, Bradshaw and Anderson 2001, Renberg *et al*. 2001). In this period, pH values in many southern Swedish lakes were raised by about one unit (Table 1) and for some lakes it could be shown that they were already eutrophied 1000 BP, due to human impact (Table 2). In both cases, diatom assemblages were different from the ones dominating during the previous period of natural acidification and oligotrophication (Tables 3–5).

Most European studies show the same pattern: conditions prevailing in 1850 AD were not at all anthropogenically uninfluenced conditions. Bjerring *et al*. (2008) found that the majority of 21 Danish lakes studied, were already eutrophied by 1850 AD, and the same results were shown for the Danish Lake Søbygård (Bennion *et al*. 1996). The start of nutrient enrichment of the Danish Lake Dallund Sø was shown to be from approximately 3700 BP, due to settlement activities such as land clearance (Bradshaw *et al*. 2005). Significant, continuous eutrophication in six lakes in northern Germany started between 1400 AD and 1700 AD, (Hübener *et al*. 2009), and Leira *et al*. (2006) studying 34 Irish lakes added evidence that a set reference point is not appropriate for all lakes in Ireland.

Despite the fact that 1850 AD is not a 'true' reference condition for all lakes, it is generally not possible to set one general fixed time of prevailing reference conditions for all Swedish lakes. Anthropogenic influences in northern Sweden (N Sweden) were not as severe as in S Sweden, for example acidifying sulfur deposition is lower in the north as compared with those in the south (Bernes 1991).

Following the definition of reference conditions according to the EU WFD, conditions prevailing during the period of natural acidification and oligotrophication would correspond to natural conditions in most southern Swedish, acid-sensitive, clear-water lakes. This phase is characterized by constant changes in pH and

Table 1. Diatom-inferred pH development in Swedish lakes as described by different authors. Empty spaces indicate unavailable information.

Lake	Natural acidification	Alkalization	Modern acidification	Liming	Author
Aborreträsket	yes		no	yes	Korsman 1999
Blåmissusjön	yes		yes		Renberg 1986
Bodanesjön		no			Renberg *et al*. 1993a, 1993b
Bösjön			yes	yes	Guhrén *et al*. 2007, Guhrén *et al*. 2003, Norberg *et al*. 2008
Brunnsjön		anthropogenic			Renberg *et al*. 1993a, 1993b
Brunnsjön			no		Ek & Korsman 2001
Ejgdesjön	yes		no	yes	Guhrén *et al*. 2007, Guhrén *et al*. 2004, Norberg *et al*. 2008
Gaffeln	yes	anthropogenic	yes		Renberg *et al*. 1993a, 1993b
Gårdsjön	yes		yes		Renberg *et al*. 1985, Renberg & Hellberg 1982, Renberg *et al*. 1993b
Gåstjärnen	yes		no		Korsman 1999
Grysjön		anthropogenic			Renberg *et al*. 1993a, 1993b
Gyltigesjön		anthropogenic	yes	yes	Guhrén *et al*. 2007, Guhrén *et al*. 2003, Norberg *et al*. 2008
Gysslättasjön	yes	no	no	yes	Guhrén *et al*. 2007, Ek *et al*. 2001, Norberg *et al*. 2008
Harasjön			yes		Ek & Korsman 2001
Härsvatten	yes	anthropogenic	yes		Renberg *et al*. 1993a, 1993b, Renberg & Hellberg 1982, Ek & Korsman 2001
Hjärtsjön			yes		Ek & Korsman 2001
Iglasjön		anthropogenic			Renberg *et al*. 1993a, 1993b
Källsjön	yes	anthropogenic	no	yes	Guhrén *et al*. 2007, Korsman *et al*. 2000, Norberg *et al*. 2008
Knottorpasjön		anthropogenic			Renberg *et al*. 1993a, 1993b
Korptjärnen	yes		no	yes	Korsman 1999
Långsjön	yes	anthropogenic	no	yes	Guhrén *et al*. 2007, Guhrén *et al*. 2003, Norberg *et al*. 2008
Lien	yes	anthropogenic	no	yes	Guhrén *et al*. 2007, Guhrén *et al*. 2004, Norberg *et al*. 2008
Lilla Holmevatten	yes	anthropogenic			Renberg *et al*. 1993a, 1993b
Lilla Öresjön	yes	anthropogenic	yes		Renberg *et al*. 1993a, 1993b, Ek & Korsman 2001, Renberg 1990
Lundsjön		natural		yes	Rosén & Hammarlund 2007
Lysevatten	yes	anthropogenic	yes	yes	Renberg & Hellberg 1982, Renberg *et al.* 1993a
Lysevatten			yes	yes	Renberg & Hultberg 1992
Makkasjön	yes	natural	no	yes	Korsman 1999, Korsman & Segerström 1998, Rosén & Hammarlund 2007
Njulla	yes				Bigler *et al*. 2003
Örvattnet			yes		Ek & Korsman 2001, Ek *et al*. 1995
Övre Skärsjön			yes		Ek & Korsman 2001
Rotehogstjärnen			no		Ek & Korsman 2001
Rudegyl		anthropogenic			Renberg *et al*. 1993a, 1993b
Skidsjön	yes		no		Korsman 1999
Sotaure				no	Rosén & Hammarlund 2007
Stengårdshultasjön	yes	anthropogenic	yes	yes	Guhrén *et al*. 2007, Gählman *et al*. 2001, Norberg *et al*. 2008
Stensjön	no	no	yes	yes	Guhrén *et al*. 2007, Ek *et al*. 2001, Norberg *et al*. 2008
Stora Härsjön			yes	yes	Guhrén *et al*. 2007, Guhrén *et al*. 2004, Norberg *et al*. 2008

continued

Table 1. Continued.

Lake	Natural acidification	Alkalization	Modern acidification	Liming	Author
Stora Lången		anthropogenic			Renberg *et al.* 1993a, 1993b
Stora Skärsjön		anthropogenic			Renberg *et al.* 1993a, 1993b
Stora Skarsjön			yes		Almer *et al.* 1974
Stora Tresticklan			yes		Ek & Korsman 2001
Storasjö			yes		Ek & Korsman 2001
Tryssjön	yes	anthropogenic	no	yes	Guhrén *et al.* 2007, Guhrén *et al.* 2004, Norberg *et al.* 2008
Tussjön		no			Renberg *et al.* 1993a, 1993b
Västra Skälsjön			no	yes	Guhrén *et al.* 2007, Guhrén *et al.* 2004, Norberg *et al.* 2008
Vuoskkujávri	yes				Bigler *et al.* 2002

nutrient concentrations, so the conditions at the end of this period and shortly before the onset of anthropogenic alkalization and eutrophication could be set as reference or natural. A general range of pH values for each period is given for southern Swedish acid-sensitive lakes by Renberg *et al*. (1993a, 1993b). They infer pH of 5.2–5.6, which would equal the reference pH value prevailing before the onset of human agriculture and thus significant human influence.

In some northern Swedish lakes, natural acidification was followed by a natural alkalization phase. In these cases, reference conditions were prevailing at the end of this period of natural alkalization.

Distinguish human-induced changes from natural changes

However, the onset of ‘significant’ anthropogenic influence is not the only important factor in setting reference conditions. It is also inevitable to try to differentiate natural changes of

Table 2. Diatom-inferred trophy development in Swedish lakes as described by different authors. Empty spaces indicate unavailable information.

Lake	Natural oligotrophication	Anthropogenic eutrophication	Modern anthropogenic eutrophication	Author
Bussjösjön		yes		Håkansson & Régnell 1993
Florsjön		yes	no	Bigler *et al.* 2008
Foxen			no	Rydberg *et al.* 2006a
Kassjön		yes	no	Anderson *et al.* 1995
Krageholmssjön		yes	yes	Håkansson 1989
Laxsjön			yes	Rydberg *et al.* 2006a
Lelången			no	Rydberg *et al.* 2006a
Mälaren (Ekoln)		yes	yes	Bradshaw & Anderson 2001
Mälaren (Södra Björkfjärden)		yes	yes	Renberg *et al.* 2001
Nedre Milsbosjön		yes	yes	Rydberg *et al.* 2006b
Östersjön		yes	no	Bigler *et al.* 2008
Råvarpen			yes	Rydberg *et al.* 2006a
Sämbosjön	yes	yes		Digerfeldt & Håkansson 1993
Stora Le			no	Rydberg *et al.* 2006a
Vänern			yes	Renberg *et al.* 2003
Västra Silen			no	Rydberg *et al.* 2006a
Vättern			yes	Renberg *et al.* 2003

Table 3. Dominant diatom taxa in different Swedish lakes after deglaciation until present. Empty spaces indicate unavailable information.

Lake	After lake development	Natural acidification and oligotrophication period	Anthropogenic alkalization and eutrophication period	Modern anthropogenic acidification and eutrophication period	After liming
Aborreträsket	*Pinnularia* spp.	*Peronia fibula*, *Eunotia* spp., *Frustulia* spp., *Brachysira* spp.			*Achnanthes minutissima*, *Nitzschia gracilis* appear
Blåmissusjön	small, epiphytic *Fragilaria* spp.	*Eunotia exigua* abundances increase, followed by dominance of small, epiphytic *Fragilaria* spp.		almost 100% *Eunotia exigua*	
Brunnsjön, Gaffeln, Grysjön, Härsvatten, Iglasjön, Knottorpasjön, Lilla Holmevatten, Rudegyl, Stora Lången, Stora Skärsjön	*Fragilaria* spp., *Cyclotella* spp.	increase of *Aulacoseira* spp., *Brachysira* spp., *Eunotia* spp., *Frustulia* spp.	*Cyclotella* spp., *Asterionella formosa*, *Aulacoseira ambigua*	*Brachysira* spp., *Eunotia* spp., *Navicula subtilissima*, *Tabellaria binalis*	
Brunnsjön (Ek & Korsman 2001), Rotehogstjärnen			*Aulacoseira distans* var. *tenella* and *Tabellaria flocculosa* dominant	*Cyclotella stelligera* and *C. comta* dominant	
Bussjösjön		*Stephanodiscus* ssp. and/or *Cyclostephanos* ssp.	epiphytic/epipsammic *Fragilaria* spp. and *Cyclostephanos dubius*, followed by exclusive dominance of *Stephanodiscus* spp.		
Bösjön		*Cyclotella kuetzingiana*, *C. meneghiniana*, *Aulacoseira italica* var. *valida*, *A. distans* var. *alpigena*		increasing *Peronia fibula*, *Eunotia incisa*, *Actinella punctata*, *Surirella delicatissima*, *Surirella linearis*, *Tabellaria binalis*, decreasing *C. kuetzingiana*	increase in *C. kuetzingiana*, *Aulacoseira distans*/*subarctica*, *Achnanthes minutissima* agg.
Gyltigesjön			*Fragilaria virescens* var. *exigua*, *Achnanthes minutissima* agg.	*Tabellaria flocculosa*, *Frustulia rhomboides* var. *viridula*	*Cyclotella comta*

Gysslättasjön	*Cyclotella stelligera*, *Aulacoseira distans* var. *tenella*, *A. lirata*, *A. perglabra* *A. perglabra* var. *floriniae*, *Fragilaria* spp., also *Peronia fibula*, *Eunotia* spp.	gradual increase of *Frustulia* spp. & *Brachysira* spp.		return of *Cyclotella stelligera*, *C. comta*, *Aulacoseira ambigua*, *Fragilaria virescens* var. *exigua*, but also *Peronia fibula*, *Eunotia incisa*
Gårdsjön	periphytic *Nitzschia* spp. & *Fragilaria* spp.	*Cyclotella comta*, *C. atomus*, *C. pseudostelligera*, *Achnanthes microcephala*, *Cymbella microcephala*, *C. gracilis*, *Fragilaria virescens*, *Navicula radiosa*, *Synedra acus*	*Cyclotella species* replaced by *Cyclotella kuetzingiana* and *Melosira* spp., other species stable	*Brachysira* spp., *Eunotia* ssp., *Navicula subtilissima*, *Tabellaria binalis*
Gåstjärnen	*Navicula* spp., *Pinnularia* spp., *Frustulia* spp.	*Cymbella* spp. flora, followed by a shift to *Eunotia* spp. flora		modern flora including *Neidium*
Harasjön, Hjärtsjön, Övre Skärsjön, Storasjön, Stora Tresticklan			*Cyclotella* spp. and *Aulacoseira* spp. dominant	*Eunotia* spp., *Brachysira* spp., *Tabellaria binalis.*, *Frustulia* spp., *Navicula leptostriata* dominant
Kassjön	*Aulacoseira* spp., *Fragilaria* spp.		*Asterionella* ssp., *Tabellaria* spp.	*Asterionella* spp., *Tabellaria* ssp.

ecosystems from human-induced changes.

For example high nutrient levels are often supposed to be human induced, but Miettinen *et al*. (2005) and Räsänen *et al*. (2006) suggest that naturally nutrient-rich lakes may be common in Finland, with diatom-inferred phosphorus levels between 30–40 μg l^{-1} before the impact of modern agriculture. Also Sacrower See in northeastern Germany was shown to be naturally eutrophic with diatom-inferred phosphorus levels of 50–70 μg l^{-1} short after development of the lake (Kirilova *et al*. 2009).

Climate is another factor influencing lake ecosystems and their catchments in several ways (Fischlin *et al*. 2007 and references within) and the Holocene climate is variable due to natural causes (Mayewski *et al*. 2004). Climate-induced changes of lake levels, water renewal times and insolation resulted in increased nutrient availability in Canadian Lake 239 during the mid-Holocene (Moos *et al*. 2009). The authors observed increasing diTP concentrations concurrent with decreasing diatom-inferred lake-levels due to a shift towards a drier and warmer mid-Holocene climate. Changing lake-levels during the Holocene were also observed in southern Swedish lakes, indicated by changes in the ratio of planktonic to benthic diatom species, e.g. in Bjäresjösjön (Gaillard *et al*. 1991), Bussjösjön (Håkansson and Regnéll 1993), Krageholmssjön (Håkansson 1989) and Havgårdssjön (Dearing *et al*. 1987). Gaillard *et al*. (1991) assumed for the most part regional climatic shifts causing the lake-level changes while stating that climatic and human impacts are difficult to separate. If the lake-level changes resulted in a changed nutrient status can not be judged, since the studies were not designed to answer this question, but would be interesting to find out. The change from the initial eutrophic state of Sacrower See to a prolonged oligo- to mesotrophic period was also ascribed to climate warming, resulting in a decreasing internal phosphorus loading (Kirilova *et al*. 2009). Korsman and Segerström (1998) observed a natural alkalization in some northern Swedish lakes about 3000 BP–1000 BP. They ascribed this alkalization as a consequence of a drier climate.

Kirilova *et al*. (2009) concluded that definition of reference conditions should not just take

Table 4. Dominant diatom taxa in different Swedish lakes after deglaciation until present. Empty spaces indicate unavailable information.

Lake	After lake development	Natural acidification and oligotrophication	Anthropogenic alkalization and eutrophication	Modern anthropogenic acidification and eutrophication	Liming
Korptjärnen	small *Fragilaria* spp. and *Stauroneis* spp.	increase of *Frustulia* spp., *Pinnularia* spp., *Navicula* spp., *Cymbella* spp., *Neidium* spp.			*Achnanthes minutissima*
Källsjön	*Achnanthesminutissima* agg., *Cyclotella kuetzingiana* and *Aulacoseira* spp.	increase of *Eunotia* spp. and *Aulacoseira distans* var. *tenella*			increase of *Asterionella formosa*, *Cyclotella stelligera* and *Achnanthes minutissima*
Lien	*Cyclotella kuetzingiana* and *C. glomerata*	increasing *Peronia fibula*, *Eunotia incisa*, *E. curvata*, *E. naegelii* and *E. vanheurckii* var. *intermedia*, *Aulacoseira distans*/*subarctica*			*Cyclotella comta*
Lilla Öresjön	small *Fragilaria* spp., followed by *Cyclotella kuetzingiana*, *C. comensis*, *C. comta*, *C. stelligera* and *Achnanthes minutissima* agg.	increase of *Aulacoseira distans* var. *tenella*, *Brachysira* ssp., *Eunotia* ssp., *Frustulia* ssp., *Peronia fibula*, *Tabellaria flocculosa* and *Asterionella ralfsii* var. *americana*, *Navicula leptostriata*, *Tabellaria quadriseptata*	*Cyclotella kuetzingiana* returns, *Achnanthes minutissima* agg.	*Navicula leptostriata*, *Eunotia naegelii*, *Tabellaria quadriseptata*, *T. binalis* increasing	
Lundsjön					same as in Makkasjön
Lysevatten			*Cyclotella* spp., *Asterionella formosa*, *Aulacoseira ambigua*, later *Cyclotella kuetzingiana* and *C. glomerata*		after first liming: *Achnanthes microcephala* and *Cymbella microcephala*, after second liming: *Achnanthesminutissima* agg., *Synedra acus* and *Cymbella microcephala*

Lysevatten (Renberg and Hultberg 1992)			*Cyclotella kuetzingiana* and *C. glomerata* dominant	*Brachysira* spp., *Eunotia* spp., *Frustulia* spp., *Tabellaria* spp., *Navicula* spp. and *Peronia fibula* dominant	*Achnanthes minutissima* agg., *Cymbella microcephala* and *Synedra acus* dominant
Makkasjön	*Pinnularia biceps* and *Eunotia* spp., subsequently dominance of *Navicula* spp. and *Cymbella* spp., later again dominance of *Pinnularia biceps*	*Eunotia* spp.	increase of *Pinnularia* spp., *Brachysira* spp. and *Stauroneis* spp., afterwards flora characterised by *Cymbella* spp., *Pinnularia* spp., *Frustulia* spp.	peak of *Navicula hoefleri* and *N. cumbriens*, followed by dominance of *N. seminulum* var. *intermedia* and *Nitzschia gracilis*	sudden change to an assemblage that never occured before, 60% *Navicula seminulum* var. *intermedia*
Mälaren (Ekoln)			1000 BP isolation from Baltic Sea; *Aulacoseira* spp.	increase of *Stephanodiscus* spp.	
Mälaren (Södra Björkfjärden)			1000 BP isolation from Baltic Sea; *Stephanodiscus alpinus*, later *Aulacoseira islandica*	*Stephanodiscus parvus* and *Cyclotella* spp.	
Nedre Milsbosjön				*Aulacoseira* spp., *Fragilaria* spp., *Cyclotella* spp., followed by dominance of *Stephanodiscus parvus*	
Njulla	small *Fragilaria* spp.	*Navicula digitulus* and diverse benthic fauna	*Achnanthes marginulata*, *A. kriegeri*, *Navicula minima*, *Pinnularia biceps*, *P. microstauron* var. *microstauron*, *Aulacoseira distans* var. *nivalis*, later: *Brachysira*spp. and *Suriella linearis* increasing		
Örvattnet			*Aulacoseira* spp. and *Cyclotella* spp. dominant	*Navicula leptostriata*, *Tabellaria binalis* and *Eunotia incisa*	

Table 5. Dominant diatom taxa in different Swedish lakes after deglaciation until present. Empty spaces indicate unavailable information.

Lake	After lake development	Natural acidification and oligotrophication	Anthropogenic alkalization and eutrophication	Modern anthropogenic acidification and eutrophication	Liming
Östersjön	*Aulacoseira* spp., *Tabellaria* spp., *Fragilaria* spp., *Cyclotella* spp.			*Asterionella formosa*, *Cyclostephanos dubius*, *Fragilaria crotonensis*	
Skidsjön	*Navicula* spp., *Frustulia* spp., *Eunotia* ssp.	shift to *Cymbella* spp. and *Brachysira* spp. flora, later shift to *Navicula* spp.		*Eunotia* spp., *Tabellaria* spp.	
Stengårds-hultasjön			*Cyclotella comta*, *Cyclotella comensis* and *Tabellaria flocculosa* increase	increase of *Aulacoseira distans/subarctica*, *Achnanthes minutissima* agg. and *Cyclotella comensis*	
Stensjön	*Cyclotella* ssp. and *Aulacoseira* ssp.		*Cyclotella kuetzingiana* increase	increase of *Brachysira vitrea*, *B. brebissonii*, *Fragilaria virescens* var. *exigua*, *Peronia fibula* and *Eunotia incisa*	*Cyclotella stelligera*, *C. comta*, *Asterionella formosa*, *Synedra acus*, *Achnanthes minutissima* agg., subsequently increase of *Aulacoseira distans/subarctica*
Stora Härsjön	*Cyclotella kuetzingiana* and *C. glomerata*			*Asterionella ralfsii* var. *americana*	
Stora Skarsjön (Almer *et al*. 1974)				increase of *Eunotia* ssp., *Tabellaria binalis* and *Amphicampa hemicyclus*	
Sämbosjön	only 15% planktonic diatoms increase to 70%–80%, mainly *Cyclotella comta*	planktonic forms stay dominant, dominance of single taxa changing			

continued

Table 5. Continued.

Lake	After lake development	Natural acidification and oligotrophication	Anthropogenic alkalization and eutrophication	Modern anthropogenic acidification and eutrophication	Liming
Tryssjön	*Cyclotella kuetzingiana* and *Aulacoseira italica* var. *valida*	*Aulacoseira distans* var. *tenella*, *A. distans/ subarctica*, *A. distans* var. *nivalis*, *Brachysira styriaca* and *Fragilaria constricta*			*Cyclotella kuetzingiana*, *Aulacoseira distans/ subarctica*
Vänern				increasing *Asterionella formosa* and *Stephanodiscus* spp.	
Västra Skälsjön	*Aulacoseira distans* and *A. subarctica*		*Cyclotella glomerata*	*Aulacoseira lirata*, *Frustulia rhomboides* var. *saxonica*, *Eunotia naegelii*, *E. rhomboidea*, *E. tenella* and *Achnanthes marginulata*	
Vättern				*Tabellaria flocculosa; later Asterionella formosa*, *Fragilaria crotonensis*, *Stephanodiscus medius*, *Aulacoseira islandica*; later *Stephanodiscus alpinus*	
Vuoskkujávri	small *Fragilaria* spp., later *Cyclotella comensis*, *Aulacoseira ambigua*, *Fragilaria nanana*, *Achnanthes minutissima*, *Asterionella formosa* and *Stephanodiscus alpinus*	*Aulacoseira ambigua* and *Aulacoseira subarctica*, appearance of *Cyclotella rossii*, *Cyclotella glomerata* quite abundant			

human influence into account, but also natural climatic effects. Depending on the climate scenario considered, different natural nutrient states are possible.

Acidification can also be due to natural or anthropogenic factors. In the 1960s, scientists became aware of the problem of rapid and severe acidification of surface waters in the northern parts of Europe and America. To counteract the effects of surface water acidification, the Swedish Environmental Protection Agency (SEPA) launched the national liming program in 1977 AD. It is still in practice and since then more than 8000 Swedish lakes have been repeatedly limed. There have been critical voices, putting the extensive and expensive liming program in question (Lundquist 2003 and references within). Despite the knowledge that anthropogenic-induced acid deposition, leading to acidification, is rather low in N Sweden, northern Swedish lakes are still limed. Deposition values are about half of the deposition of S Sweden (Bernes 1991). Wilander and Fölster (2007) assessed that none of the limed lakes in the county Norrland, northern Sweden, was acidified.

Bishop *et al.* (2001) argued that many surface waters in N Sweden are naturally acidic due to the crystalline bedrock with limited acid neutralizing capacity (ANC), high concentrations of total organic carbon (TOC) and sulphide-rich soils. Acidification episodes during snow melt, typical for surface waters in N Sweden, are mostly due to dilution of ANC by meltwater and organic acids (natural acidity) and only a small proportion can be attributed to mineral acids from atmospheric deposition (anthropogenic acidification) as shown for Swedish streams by Laudon *et al.* (2000).

In 1990, more than 3500 lakes in the two northernmost counties of Sweden were classified as anthropogenically acidified by the Swedish Environmental Protection Agency (SEPA). Studying the prehistoric diatom community in 118 lakes in these two counties, Korsman (1999) found a long-term natural acidification trend in all lakes, with just minor changes in pH, alkalinity and colour from pre-industrial to present times. No support for modern, large-scale acidification could be found. Significant changes in diatom-inferred pH could only be observed in five out of 118 lakes, all located in the south-eastern coastal part, the region with the highest acid deposition in N Sweden and additionally sulfur-rich sediments. The author suggested sulphur deposition, ditching (causing the oxidation of sulphur in these sulphur-rich, marine sediments) and land use changes were causing this significant decrease in pH. The studies of the diatom sediment record in the northern Swedish lakes Makkasjön (Korsman and Segerström 1998), Vuoskkujávri (Bigler *et al.* 2002) and Njulla (Bigler *et al.* 2003) did not find signs of modern anthropogenic acidification, but indicated a long-term natural acidification due to soil-forming processes and vegetation development.

Problems arising when reference conditions are unclear: Does liming help to re-establish pre-acidification conditions?

The acidification of Swedish surface waters, resulting in the Swedish liming program, is a good example of the necessity of determining 'right' or 'true' reference conditions in order to mitigate negative effects in a sustainable and effective manner. Does restoring a lake back to a certain pH by liming also result in ecological restoration? This is a question still unanswered.

The program for integrated studies of the effects of liming acidified waters (ISELAW) was launched in Sweden in 1989 to monitor the effects of liming in lakes and streams. As a part of this program, diatom assemblages in sediment cores were analysed in a number of lakes, and pH history of the lakes was reconstructed. When analysing the results of this program, Norberg *et al.* (2008) found that just five out of 12 lakes of the Swedish liming program were acidified during recent decades, according to the past diatom assemblages (Table 1). Since palaeolimnological methods do not allow detection of episodic acidification and low sedimentation rates plus bioturbation may mask acidification signals the authors can not exclude that more lakes should be classified as recently acidified. Despite this uncertainty in acidity classification all 12 lakes were and still are limed (Table 1).

Guhrén *et al.* (2007), studying the same 12 lakes, defined the reference or natural conditions as conditions before the onset of anthropogenic alkalization and eutrophication. They inferred pH values around 5.3–6.5 for ten out of the 12 ISELAW lakes during this period. This pH range would then also be the natural or reference pH range for acid-sensitive, clear-water lakes in S Sweden.

Observed ecological effects of liming are ambiguous. Studying the 12 ISELAW lakes, Guhrén *et al*. (2007) found that after liming some diatom communities returned to the pre-acidification or pre-alkalization diatom composition, while others did not change, changed very slightly or changed to a community composition previously not found in the history of the specific lakes. Even though diatom communities after liming differed from those found previously in some of the lakes, the five recently acidified lakes showed clear signs of recovery from acidification in the form of increasing pH (Norberg *et al*. 2008). Renberg and Hultberg (1992) found a similar result in Lake Lysevatten, S Sweden. Liming increased pH of lake water from 4.4–4.9 (during the acute acidification phase in the 1960s) up to pH 7.5, resulting in a diatom flora not found in that lake before. The second and third liming after reacidification to pH 5 could not restore the natural diatom flora of this specific lake. The same observation was made in Makkasjön and Lundsjön. Liming changed diatom assemblages to assemblages not found before in these lakes (Rosén and Hammarlund 2007). Several other studies showed that the planktonic diatom flora of lakes in Sweden and the United Kingdom found prior to recent acidification could not be restored completely by liming and that species not found or found just in low abundances became dominant after liming or vice versa (Flower *et al*. 1990, Anderson *et al*. 1997, Hörnström 2002). Therefore, deciphering if liming could restore a good ecological state is difficult.

Norberg *et al*. (2010) investigated whether liming of 31 Swedish lakes resulted in a diatom composition that has occurred previously in one of the 291 Swedish lakes of a reference data set. Studying, among others, the same lakes as Guhrén *et al*. (2007) the results showed that all diatom assemblages, established after liming, could be found in pre-industrial times, prior to major human impact or even in initial pristine conditions directly after lake formation in one or more lakes of the reference data set. Therefore, even if the diatom composition after liming could sometimes not be observed in the specific lake previously, the established diatom assemblages after liming are not unnatural or untypical for Swedish lakes.

Determination of type-specific reference conditions — the Swedish lake typology

Besides the time of onset of significant human impacts and the differentiation between natural and anthropogenic induced changes of aquatic ecosystems a third factor is important: to classify lakes into different lake types.

The EU WFD requires the establishment of type-specific reference conditions for all lakes by all EU member states. Classification of lakes has to be meaningful; different lake types have to react in a specific way to pressures and have to show specific reference conditions.

In Europe, several palaeolimnological studies have been conducted to test classification of lakes. Hübener *et al*. (2009) discussed the level of differentiation of German lake types, according to Mathes *et al*. (2002), required to set reference conditions. They suggest a further differentiation of the Central Baltic Lake-Type 1 (German Lake-Type 10, defined as stratified, carbonate-rich lowland lakes with large watershed area and retention times of 0.1–10 years) when identifying reference diatom assemblages and reference trophic state. Attempts to differentiate Scottish lochs based on their diatom assemblages resulted in one specific reference assemblages for each of the four different lake types, differentiated by water depth and alkalinity/productivity and the authors were confident in their approach of characterizing lake types (Bennion *et al*. 2004). Miettinen *et al*. (2005) found that water chemistry variables, like total phosphorus concentrations and water colour, should be ignored in the Finnish lake typology according to EU WFD, suggesting catchment geology for the typology instead. Räsänen *et al*. (2006) found that naturally eutrophic lakes in Finland are an appropriate type for WFD-based lake management, although factors like depth and morphology also seem to influence diatom assemblages and may therefore be good criteria for further classification of naturally eutrophic lakes.

Swedish lakes are differentiated by ecoregions (seven regions), mean or maximal depth (deep or shallow), surface area (large or small), humus concentration (clear or humic water) and alkalinity (high or low alkalinity), as suggested by the SEPA. Several palaeolimnological studies have been conducted in Swedish lakes. However, most studies were not connected to the EU WFD and information required for classification of these lakes was too fragmentary to be able to suggest type-specific reference conditions.

Further work is needed to collect and analyse all background data to be able to assign these lakes to a lake type.

Determination of reference diatom communities for Swedish lakes

Information about the acidity and trophy development and diatom succession over time from 61 Swedish lakes was gathered (Tables 1–5) and a few very general remarks about possible reference communities can be provided.

Small, benthic, alkaliphilous *Fragilaria* species dominated the diatom communities in many Swedish lakes during the period directly after deglaciation (Tables 3–5), due to the alkaline conditions (Bigler and Hall 2002) and their ability to quickly adapt to changes in the environment (Lotter *et al*. 2010). Later on, during this period these communities are often replaced by a planktic, alkaliphilous community dominated by *Cyclotella* species (Renberg 1990). Bigler *et al*. (2006) suggested intensification of thermal stratification of the water column as the cause.

During the period of natural acidification and oligotrophication assemblages shift to assemblages dominated by acidophilous, mostly benthic taxa as *Eunotia* spp., *Frustulia* spp., *Brachysira* spp., *Tabellaria* spp., *Peronia* spp. and *Aulacoseira* spp. (Renberg *et al*. 1993b). Planktic diatoms generally show decreasing abundances and biodiversity with decreasing pH, acidification is suggested to shift diatom productivity from the pelagic zone to the littoral and profundal (Renberg *et al*. 1985).

In some northern Swedish lakes the community shifted from a benthic, *Fragilaria*-dominated assemblage to a planktic community, dominated by *Cyclotella* spp. (Bigler *et al*. 2006) or *Cyclotella* spp. and *Aulacoseira* spp. (Bigler *et al*. 2002) without any further shifts. In other lakes in N Sweden, the assemblages shifted from a benthic, *Fragilaria*-dominated assemblage to communities dominated by benthic taxa as *Navicula* spp., *Achnanthes* spp., *Pinnularia* spp. and *Cymbella* spp. (Korsman and Segerström 1998, Bigler *et al*. 2003). This shift to a different benthic community is explained by decreased mineral turbidity due to stabilization of the catchment, resulting in increased light availability and also natural acidification processes (Bigler *et al*. 2003).

Thus, possible reference diatom communities of some Swedish lakes could either be benthic, acidophilous communities (dominated by *Eunotia* spp., *Frustulia* spp., *Brachysira* spp., *Tabellaria* spp., *Peronia* spp. and *Aulacoseira* spp.), planktic communities (dominated by *Cyclotella* spp. and *Aulacoseira* spp.) or benthic communities (dominated by *Navicula* spp., *Achnanthes* spp., *Pinnularia* spp. and *Cymbella* spp.). Without knowledge of lake type, diatom reference communities cannot be determined in general, being possible for single lakes only.

With the beginning of the anthropogenic alkalization and eutrophication period the diatom communities of many Swedish lakes shift back to planktic assemblages, with *Cyclotella* spp., *Aulacoseira ambigua* and *Asterionella formosa*, indicative of more alkaline conditions (Renberg *et al*. 1993b). *Stephanodiscus* spp. and *Cyclostephanos* spp. are becoming dominant in lakes undergoing anthropogenic eutrophication, as most species belonging to these taxa are indicative of relative high nutrient concentrations (Håkansson 1989, Håkansson and Regnéll 1993, Digerfeldt and Håkansson 1993).

Conclusions

Evaluating the results of palaeolimnological studies with focus on diatoms in Swedish lakes shows that it is not possible to set one general fixed time of prevailing reference conditions for all Swedish lakes, as significant human disturbances started much earlier and were more severe in S Sweden as compared with the northern parts of the country.

1850 AD, a date applied by many European researchers as the time before significant human influence, is not an appropriate reference time for all Swedish lakes. In many southern Swedish lakes diatom assemblages were already anthropogenically altered due to increased pH and nutrient levels.

Reference acidity and nutrient conditions in acid-sensitive, clear-water lakes in S Sweden, as defined by the EU WFD, would be conditions prevailing at the end of the natural acidification and oligotrophication period about 2300 years ago. Reference acidity and nutrient conditions in most northern Swedish lakes prevailed longer compared to S Sweden or may even prevail today, since anthropogenic disturbances started later than in southern Swedish lakes.

In order to determine the reference acidity conditions more variables have to be taken into account. Not all lakes in S Sweden are vulnerable to anthropogenic acidification to the same degree, lakes on weathering bedrock with thick soils show no or only slight signs of acidification. Also lakes with high TOC levels are less vulnerable to acid deposition than lakes with low TOC levels. In these lakes acidity reference conditions may even be found today.

The number of palaeolimnological studies of Swedish lake sediments is too small and information published too general to be able to determine reference conditions. An analysis of the complete data set (background data) of all palaeolimnological studies conducted in Swedish lakes will surely help to establish reference conditions (pH, trophy and diatom communities) for at least some of the Swedish lake types. It may also indicate how relevant the suggested Swedish lake typology is.

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