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Recovery from acidification of invertebrate fauna at ICP Water sites in Europe and North America

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REPORT

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Abstract

The report presents update results for trends in biology after 2000. The results confirm the results from the ICP Waters report in 2000 and shows continued improvement in biological recovery from regions with new data. The recovery of sensitive fauna corresponds with the chemical recovery of lakes. The improvements is most pronounce in Scandinavia (Norway) and Canada. A beginning recovery is recorded in the Czech Republic, while unstable (not significant) improvements are recorded in Germany. The lagtime in recovery of different species/communities is evaluated to vary from a few to about 20 years, after reaching their critical limit.

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CONVENTION ON LONG-RANGE
TRANSBOUNDARY AIR POLLUTION

INTERNATIONAL COOPERATIVE PROGRAMME ON
ASSESSMENT AND MONITORING OF ACIDIFICATION
OF RIVERS AND LAKES

Recovery from acidification of invertebrate fauna at ICP Water sites in Europe and North America

Prepared by the ICP Waters Programme Subcentre
Laboratory of Freshwater Ecology and Inland Fisheries
University of Bergen, July 2004

Preface

The International Cooperative Programme on Assessment and Monitoring of Rivers and Lakes (ICP Waters) was established under the Executive Body of the Convention on Long-Range Transboundary Air Pollution at its third session in Helsinki in July 1985. The Executive Body accepted Norway's offer to provide facilities for the Programme Centre which has been established at the Norwegian Institute for Water Research, NIVA. A programme subcentre has been established at the Laboratory of Freshwater Ecology and Inland Fisheries at the University of Bergen. The ICP Waters programme has been led by Berit Kvaeven, Norwegian Pollution Control Authority.

Among the aims of the programme are to evaluate the impact of atmospheric pollution, in particular acidification, on surface waters and describe and analyse the effects on aquatic biology. This report summarises the biological results achieved so far from the countries delivering biological data. Focus of the report is on trends and regional changes in surface water chemistry and the effects of these changes on benthic invertebrates. Acidification status based on invertebrates is also given for sites with short biological series. One aim of the report is therefore to give an overview over the existing biological material from the different countries/sites and evaluate these data with respect to acidification.

Bergen, July 2004

Gunnar G. Raddum

Contents

Summary

The ICP Waters 15-year report documented widespread improvements in surface water chemistry in response to emissions controls programs and decreasing acidic deposition. The ultimate goal of emissions control programmes is biological recovery, e.g., the return of acid-sensitive species that have disappeared and the restoration of biological functions that have been impaired during the course of acidification.

In 2000 the ICP Waters programme conducted a trend analysis of available biological data. The report described results from Ireland, UK, Scandinavia and Central Europe including lowland as well as mountain areas. For the UK and most sites in Germany no statistically-significant trends in acidification were found, although some positive signals of improvements in the invertebrate fauna were observed. A clear positive trend was, however, found for the Norwegian sites and for most of the Swedish sites. For the most acidic sites in central Europe, it was concluded that improvement in water quality was low and unstable and had not yet reached levels at which effects on biology can be expected or detected.

This report presents an update of results for trends in biology (invertebrate fauna) since 2000 in lakes and rivers connected to the ICP Waters programme. The report does not give a full literature review of reports of biological recovery from acidification. We have restricted the evaluation to countries that have delivered new data after 2000. The trend analyses are performed by use of different methods including: 1) long-term trends from sediment cores, 2) trend analysis based on acidification indices and number of taxa, and 3) multivariate statistical analyses based on the biological community, water chemistry and time.

The results in this report confirm the results from the report in 2000 and shows that continued improvement in the chemical status of acid-sensitive lakes and streams leads to biological recovery. The trends in biological recovery vary from region to region in accordance with changes in water chemistry.

Responses in different regions:

- In Canada recovery of damaged zooplankton communities has taken place in lakes that chemically have recovered from pH < 6.0 to pH > 6.0. Some recovery of zooplankton species has also occurred in lakes that have not reached the pH > 6. Recovery of benthic invertebrates is also observed 4 – 8 years after the water chemistry has improved to reach the critical limit of the species.
- In Central Europe beginning of zooplankton recovery is recorded in lakes in the Bohemian Forest, Czech Republic. There are also some signs of recovery of sensitive species in several of the German sites, mostly among those situated in the eastern part of the country. However, clear stabile significant recovery is difficult to point out. The general lack of significant trends in the German biological data corresponds with high variation in the water chemistry as well as a general lack of general chemical improvements in the German sites. However, the development in several German sites can quite soon show significant improvements.
- In Scandinavia, especially Norway, all methods used for evaluating biological recovery shows improvements over the last 10 – 14 years. The recovery is most pronounced the

latest years where number of localities with significant recovery has increased considerably. This is in accordance with the development in water chemistry. In some Swedish lakes also significant improvements in benthic communities are recorded over time, corresponding with improvements in water chemistry during the same period.

Historical data, core analyses, from the Alps and the Pyrénées indicate generally stabile pH up-core. The changes among invertebrates seam to be attributed to changes in weathering and climate rather than acidification.

From Finland, Ireland, Polen and UK the programme centre have no new data after 2000 that can be used for updating the situation reporter in 2000. However, other investigations in Finland and UK show that sensitive fauna has recovered in formally acid lakes, due to increased pH and decreased labile aluminium. The data from Latvia demonstrate no acidification problem.

1. Introduction

Over the past 30 years acid atmospheric deposition has received considerable attention as an international environmental problem. The ICP Water monitoring programme is designed to assess, on a regional basis, the degree and geographical extent of acidification of surface waters. Changes in acidic deposition are evaluated with respect to changes in the chemical status of lakes and rivers (dose/responds relationship), the physical environment and biological response. In this report, the focus is on the magnitude of chemical change, evaluation of critical limits of sensitive invertebrates and their responses to the changes.

In 2000 the ICP Waters programme conducted a trend analysis of available biological data for the period 1988-1998. The report described results from Ireland, UK, Scandinavia and Central Europe including lowland as well as mountain areas. For the UK and most sites in Germany no statistically-significant trends in acidification were found, although some positive signals of improvements in the invertebrate fauna were observed. A clear positive trend was found for the Norwegian sites and for some Swedish sites. For the most acidic sites in central Europe, it was concluded that improvements in water quality have not yet reached a level at which stable effects on biology can be detected.

This year's report presents an update of results for trends in invertebrate fauna after 2000. The report does not give a full literature review of other reports of biological recovery from acidification. The evaluation is restricted to countries from which there are new data after 1998. The results are divided into i) long-term trends from sediment cores, ii) trend analysis based on acidification indices and number of taxa, and iii) multivariate statistical analyses in which the biological community, the water chemistry and the time are evaluated together.

1.1 Factors influencing biological recovery

The ICP Waters 15-year report documented widespread improvements in surface water chemistry in response to emissions controls programs and decreasing acidic deposition. The ultimate goal of emissions control programs is, however, biological recovery, e.g., both the return of acid-sensitive species that have disappeared and the restoration of biological functions that have been impaired during the course of acidification.

Chemical recovery has varied mostly in accordance with changes in acid deposition. The greatest recovery in water chemistry has therefore occured in areas with large reductions in sulphur deposition. This has been especially evident in parts of Scandinavia where the S-deposition has been reduced with more than 50%. In these areas we also find the best evidence for biological recovery (Raddum 2003), while recovery of biota are more or less lacking from other areas in Europe. The reason for varying biological recovery can be due to several factors. The rapid response in lakes and rivers in Scandinavia can be caused by soils with poor buffer capacity, and the water quality will therefore respond quickly both during acidification as well as during a recovery process. In areas with higher buffering capacity, localities are likely to respond more slowly on changes in sulphur deposition due to release of stored sulphur, see Prechtel *et al.* (2001) and Skjelkvåle *et al.* (2003). Improvements in water chemistry due to reduced sulphur deposition are in some cases counteracted by increased nitrate deposition like in some areas of Germany (Bolte et al 2001). These variations in chemical recovery influence directly the recovery of invertebrates.

Other factors can also disturb biological recovery. One such factor is variation in the deposition of seasalts. This seems to have masked trends in chemical recovery during the early part of the 1990s in UK (Evans *et al.* 2001) and in turn impeded biological recovery. Another factor may be decrease in base cation deposition. This may have reduced the chemical recovery or in some cases even contributed to further acidification (Stoddard *et al.* 1999). A third factor is climate. Veseley *et al.* (submitted) attributed accelerated decrease of aluminium in lakes in the Tatras to increasing temperature (global warming). Such a phenomenon can enhance recovery of sensitive organisms. Overall, the extent of chemical recovery from acidification varies over time, between regions and sites depending on a wide range of parameters. This depends on deposition change, catchment characteristics as well as influences from these ‘confounding factors’ (Skjelkvåle *et al.* 2003). However, reduction of S deposition will over time lead to significant improvements in the chemical status of acidified waters. This will in turn induce recovery of acid sensitive fauna, a process that occur in stages and will be time consuming. The main stages will be to achieve a stabile target water chemistry above the critical limits of species. The next steps will be dispersal of the species from a source population, survival and internal dispersal in the lake/river as well as time needed to reach natural fluctuations (Yan *et al.* 2003). Temporal fluctuations in water chemistry normally occur over time during chemical recovery. Such fluctuations can cause set-backs of recovery of sensitive organisms (Raddum and Fjellheim 2003, Skjelkvåle *et al.* 2003).

2. Data and methods

In this report the detection of recovery of sensitive invertebrates is tested using several different methods. The methods used are:

- Trends in acidification index, which is especially useful in the pH range, 4.7 - 6.0, where most sensitive species have their critical limits.
- Trends in number of sensitive taxa over time.
- Multivariate statistical analyses. These analyses take into account the whole invertebrate community and in contrast to the acidification indexes they are also useful at pH's below 4.7 and above 6.0.
- Inferred pH development by use of analyses of lake sediments.

The acidification index is based on presence/absence of sensitive species and is a very sensitive method. However, presence of a few sensitive species/individuals will give the same index as presence of many. It is therefore important to combine this method with detailed development of sensitive species. This will indicate the power of the index value. Long term trends in the index are also much more reliable than comparisons between years or short term assessments.

The multivariate statistical analyses use the whole community of the fauna where all species/taxa have the same weight independent of acid tolerances.

The inferred pH development from core analyses is also a multivariate approach based on organisms leaving remains that can be identified to species. Sediment sections consist of several years dependent on the productivity of the lake. Core analyses are of low interest for short term analyses (10 - 20 years), but are useful for analyses over centuries.

The acidification index needs qualitative samples and good quality of the identifications of the most important species. Among the mentioned methods the acidification index has the lowest demand to resolution of the fauna. However, the better the taxonomic work the more reliable is the index. The biological material from all countries in the ICP Water database is suitable for determinations of the acidification index, see Table 1.

The multivariate analyses need quantitative or harmonized relative numbers of the species/taxa over time. It is therefore important not to vary the sites, time of sampling and resolution of the taxonomic work during the monitoring. To trace changes in the fauna in relation to water chemistry at least 10 years/or sets of harmonized data are required.

Table 1. *Summary of data available for biological analysis. For some countries the sampling program for biology has varied over the years. Short series of biological material from ICP Water sites sampled during other programs are also included to enlarge the amount of data. However, trend analysis is only possible for sites with regular sampling programs both with respect to time and locality.*

	Biological data	**Type of analysis**	**Source of data**
Austria	Sediment core	Long term trend > 100 year	EU-projects[1]
Canada	Monitoring data submitted to ICP Waters	Trend anaysis	Canadian monitoring programme[2]
Finland	No new data since 1997	No analysis	The Finnish programs HAPRO and REPRO[3]
France	Sediment core data	Long term trend > 100 year	EU-projects[1]
Germany	Monitoring data submitted to ICP Waters	Trend analysis	National monitoring programme
Italy	Sediment core data	Long term trend > 100 year	EU-projects[1]
Ireland	No new data since 1995	No analyis	
Latvia	Monitoring data submitted to ICP Waters	Trend analysis	National monitoring programme
Norway	Monitoring data submittes to ICP Waters	Trend analysis	National monitoring programme[1]
Poland	Status data	No analyis	EU-projects[1]
Spain	Sediment core, status data	Long term trend > 100 year	EU-projects[1]
Sweden	Monitoring data submitted to ICP Waters	Trend analysis	National monitoring programme
UK	No new data since 1998	No analyis	National monitoring programme - UK Acid Waters Monitoring Network (AWMN)

References to projects:
1) the EU-projects AL:PE, MOLAR and EMERGE with ICP Waters sites included (Wathne *et al.* 1997, Wathne and Rosseland 1999, MOLAR 1999, Mosello *et al.* 2000),
2) The Northern Lake Recovery (NLRS) program (Gunn and Sandøy 2003)
3) Hynynen (2004).

3. Previous findings on biological recovery in the ICP Waters Programme

The ICP Waters database has been previously used to evaluate the critical response of invertebrates to acidified surface waters. Critical limits of acid neutralizing capacity (ANC) were suggested for various regions based on the most sensitive and common organisms. However, setting critical limits is a continuous process as new data and knowledge arise. At present the critical limit is set to ANC 20 µeq/L for Ireland, UK and Norway. For Sweden, Germany and the Voges Mountains of France a limit of 50 µeq/L has been suggested, due to presence of organisms with higher sensitivity. Germany, however, has proposed to use ANC of 20 µeq/L like that used for the Atlantic countries. The reason is that the more sensitive species are rather rare and are therefore of minor relevance. In the high Alps and Pyrenees the available data so far indicate a critical limit of about ANC 30 µeq/L. For further information see Raddum and Skjelkvåle (2001), Skjelkvåle *et al.* (2000).

Documentation of biological response to reduced surface water acidification has so far been scattered. No large-scale biological recovery has been reported. Long-term biological monitoring data show, however, recovery of invertebrates in the Scandinavian countries, while at the most acidified sites in central Europe effects on biology can generally not be detected statistically. However, positive signals of improvements in the invertebrate fauna have been observed. There is a need to understand the sequence of steps in the ecological recovery process since all biological communities are dynamic. Ecosystems may not return to an earlier stage, since they always will reflect the present physical, chemical and biological environment.

A workshop on Models for Biological Recovery from Acidification in a Changing Climate (Wright and Lie 2002) held in collaboration with other research programmes examined evidence for biological recovery. Focus was given to recovery time, consisting of different stages. Main stages were identified to arrival time for the species, internal dipersal time and time for obtaining natural fluctuations of organisms. The recovery time will vary depending on the type of organism. For example, some algae need 0-1 year, some sensitive invertebrate species 1-3 years, some zooplankton species 3-7 years and fish 2-20 years. Static models for estimated biological status are well established, while dynamic models for biological recovery need to be developed further through cooperation between biologists, chemists and modellers.

Trends in biological recovery are often assessed by the use of acidification indices. The use of multivariate statistics is a new method developed within ICP Waters. The aim is to show correlation between changes in various variables (pH, Ca, ANC, TOC and time) and biological responses.

4. Biological status at ICP Waters sites with short data series

4.1 Finland

In the database we have biological data from Finland only up to 1988**.** The status of the studied lakes was at that time acidified with absence of the most sensitive organisms. In Finland, acidification has been studied by the HAPRO project in the mid-1980s. A subset of the lakes in this project was resampled during 2001 in the REPRO project (Hynynen 2004). Some slight recolonization of moderately sensitive invertebrates was recorded in the formerly acidic lakes. Increased pH and decreased labile aluminium concentrations were the main factors explaining the recolonisation of sensitive fauna (Hynynen *op. cit*.).

4.2 Ireland

The three ICP-Waters sites in Ireland are small headwater lake systems situated in an acid sensitive region in Counties Donegal (Lough Veagh) and Galway (Lough Maumwee) on the western seaboard, and in County Wicklow (Glendalough Lake) on the east coast. In 1995 there was a shift in monitoring method of invertebrates. From 1995 the acidic status was evaluated in the field based on the species found in the sample at the site. This gives the acidification score, but no information on species assemblages, numbers and relative densities, necessary for statistical analyses. The invertebrate fauna in the littoral zone of the Irish ICP Waters sites consists of species typical for low or non-acidified lakes. Most of the sites contain very sensitive organisms and obtain the acidification score 1. One locality GLE 13 shows damages on sensitive species, reflecting a strongly acidic site. This situation has with few exceptions been unchanged during the period 1987 - 1995. We have no biological information after 1995. In the locality GLE 11 the recorded number of *B. rhodani* has increased simultaneously with improvements in water quality up to 1995. However, the continuation of this improvement in biology can not be evaluated due to lack of data. It is therefore difficult to conclude about biological recovery for the Irish sites.

4.3 Latvia

The sites with biological sampling are from the rivers Barta, Liela Jugla, Burtneiki, Zaki and Tulija. The sampling started in 1997. Both pH and ANC are far above levels expected to be critical for invertebrates; ANC values are between 2000 and 4000 µeq/L. The sites are valuable as references for water with very high ionic strength. Compared with the other lowland regions the mussels and snails are more frequent in the Latvian waters than in any other areas. Also the relative frequency of sensitive organisms are higher in these rivers than in the other not acidic rivers reporting to ICP Waters. However, this is due to a relatively short list of taxa, indicating that the number of taxa is decreasing in water with high pH, ANC etc. One example is the stoneflies which is not recorded in these localities. Stoneflies are normally indicators of unpolluted oligotrophic water. The Latvian rivers might contain too much organic material for stoneflies. High content of organic material is, however, indicated by the high density of oligochaets. Typical for eutrophic waters are also the high abundance of snails and mussels, which also give raise to a large amount of leeches as recorded in the Latvian sites. The invertebrate composition in the localities indicates that the sites are suitable for monitoring eutrophication and organic pollution. The data constitute a very good basis for the typology of such waters and are suitable for monitoring purposes connected to the Water Framework Directive.

4.4 Poland

One of the Polish sites in the ICP Waters, Dlugi Staw, has been investigated with respect to biology in the AL:PE I, II and Molar projects (Wathne et al. 1997, Wathne and Rosseland 1999, MOLAR 1999 and Mosello et al. 2000). The water of the lake is regarded as ultraoligotrophic with low diversity of species and low density of invertebrates. There are no fish in Dlugi Staw whereas brook trout exists in other lakes in the area. In 1993 and 1994 some moderate sensitive invertebrates were recorded, like the flatworm *Crenobia alpina*, small mussels (*Pisidium sp.*) and two moderate sensitive chironomids. The fauna indicate a clearly acidified lake. No trend analyses are possible.

4.5 UK

The programme centre has not received new biological material from UK after the 12 year report (Skjelkvåle *et al.* 2000). There is, however, mounting evidence of a widespread biological response to declining water acidity in acidified lakes and streams in the UK. On the UK Acid Waters Monitoring Network (AWMN) the species composition of epilithic diatoms (single celled algae which grow attached to rock surfaces) and aquatic macroinvertebrates (largely representing insect larvae and beetles) has changed significantly at approximately half of all sites. In most cases these trends are confined to those sites which show improvements in Acid Neutralising Capacity (ANC). Where epilithic diatom communities have changed, acid sensitive species have usually increased proportionally relative to acid tolerant species. There is also evidence for an increase in the proportion of some known acid sensitive macroinvertebrate taxa, while there has been a notable increase in the representation of predatory animals, most notably Caddis species, which is consistent with an expansion of the aquatic chain. Acid sensitive mosses and higher plants have been found for the first time in the last five years at several sites, mostly those where ANC has increased. In contrast, positive changes in salmonid density have only been identified at one site, which has shown the largest change in ANC on the Network. Here, the density of brown trout has expanded significantly as pH and alkalinity has increased and labile aluminium concentration has declined.

5. Trends in biological changes documented from sediment cores

5.1 Austria

In Austria sampling of invertebrates was carried out from running water during the period 1989-1991. A minor damage on the fauna was observed in spring 1989 at one site in Stingelbach. All the other invertebrate data from the sampling period indicated low or no acidification damage.

From one lake, Schwarzsee ob Solden (AU03), chemical data is reported regularly to ICP Waters. The site is situated 2799 m.a.s.l. Biological data exist from the EU-projects AL:PE I – II and MOLAR (Wathne *et al.* 1997 and Wathne and Rosseland (eds.) 1999). The fauna of the lake is very poor due to the high elevation. In the nineties two moderate sensitive species, the flatworm *Crenobia alpina* and the chironomid *Micropsectra radialis,* were recorded in relative high numbers, indicating a moderate acidic site. The biological data from the lake is few and trend analysis based on contemporary fauna is not possible.

During the EU-project analyses of head capsules of chironomids as well as analyses of diatoms for reconstruction of pH was carried out from sediment cores. The core represents a time span of > 200 years. The inferred pH for this period was quite stabile between 5.7 - 6.0. Most of the lower pH's (5.7 - 5.8) were, however, estimated in the upper part of the core, but no clear trend in acidification was seen (Wathne *et al.* 1997).

The relative occurrence of the dominant chironomid species changed up-core (Figure 1). The up-core increase of *M. radialis*, a moderate sensitive species, demonstrate that acidification did not restrict the development of the species. The shift in the relative occurrence of *Pseudodiamesa branickii* and *M.radialis* is probably due to changed trophic status during the last 200 years. *P. branickii* prefer ultra-oligotrophic water, while *M. radialis* is common in oligotrophic environment (Wathne *et al.* 1997). The reason for the possible change on the nutrition level can be climate change, increased deposition of nutrients or increased weathering due to air pollution. The dominance of *M. radialis* in the top of the core as well as in the contemporary fauna, indicates that acidification is moderate and unchanged during the last decades.

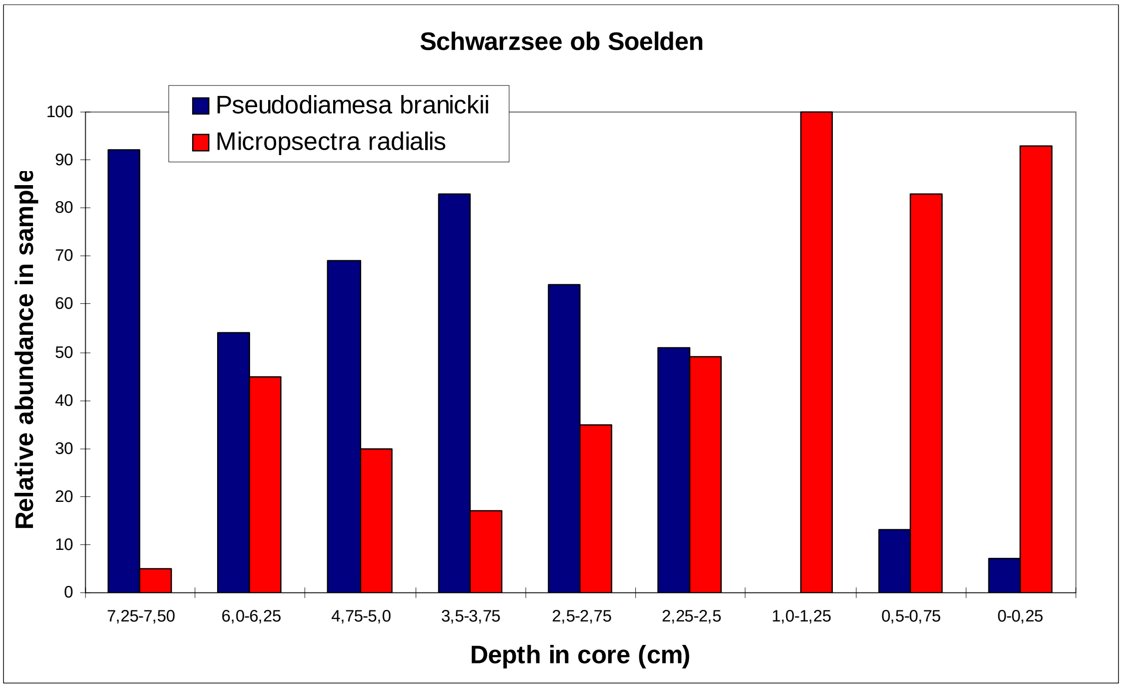


Figure 1. *Relative occurrence of Pseudodiamesa branicki (ultraoligotrophic species) and Micropsectra radialis (oligotrophic).*

5.2 France

Core analyses of diatoms from Lake Aubé (FR01) located in the Pyrénées mountains in southern France showed a very stabile pH over the last centuries. Inferred pH at core bottom and core top were 6.0 and 5.9, respectively. The lake is situated 2091 m above sea level, and with granite in the bedrock. Biological samples were taken during the EU-projects AL:PE and MOLAR (Wathne *et al.* 1997, Wathne and Rosseland 1999). The fauna was dominated by chironomids, indicating low acidification. Core analyses of diatoms showed a very stabile pH over the last centuries. Inferred pH at core bottom and core top was 6.0 and 5.9, respectively.

In the Voges mountains in the North-Eastern part of France, we have information about the extent of acidification (Guerold *et al.* 1999). Damages to the invertebrate fauna, similar to the results obtained for Germany, have been recorded. The most sensitive species were analysed with respect to their critical pH and ANC (Raddum and Skjelkvåle 2001). This study indicated that the most sensitive species could be damaged if ANC fell below 50 µeq/L. This is higher than the critical level of 20 µeq/L set for Norway (Raddum and Skjelkvåle 2001) and later for Germany (Bolte *et al.* 2001).

5.3 Italy

The Italian ICP Waters sites are all located in the Lake Maggiore watershed, in North-Western Italy. Two of the lakes included in the ICP Waters, Lakes Paione Superiore (IT03) and Inferiore (IT04) are high mountain lakes and invertebrate data are available through the EU-rojects AL:PE I and II and MOLAR (MOLAR 1999, Fjellheim *et al.* 2000 and Mosello *et al.* 2000). The lakes are located above the timberline, in areas not affected by local or direct anthropogenic sources of pollutants. They are regarded as very sensitive to acidification (Mosello *op.cit*.).

Core analyses of diatoms in Pione Superiore (IT03) in Italy, showed an inferred pH at the base of the core of 5.8-5.9 (300 years ago) and 5.6 at the top. This is very close to the present pH in the water. The change in the species composition registered over the 300-year period is probably not caused by acidification since pH is more or less unchanged over the last hundred years. Moderate increase in acid deposition may have increased weathering rates, and in combination with increased temperature this may cause increased production in the lake.

Head capsules of chironomids show a shift in the dominance of *Micropsectra radialis* and *Tanytarsus sp. B,* the two most frequent species in the core (Figure 2). The gradual replacements by *Tanytarsus sp*. indicate that the lake has become less oligotrophic during the recent decades, the same pattern as seen for Schwarzsee ob Soelden (AU03) in Austria. Schwarzsee ob Soelden seems to be similar to Pione Superiore 300 years ago (Wathne *et al.* 1997). The changes in the species composition in the two alpine lakes are probably not caused by acidification since pH is more or less unchanged. However, increased acid deposition may increase weathering which can enhance productivity. Wathne *et al.* (1997) suggested that this in combination with increased temperature, has elevated the production of the lake.

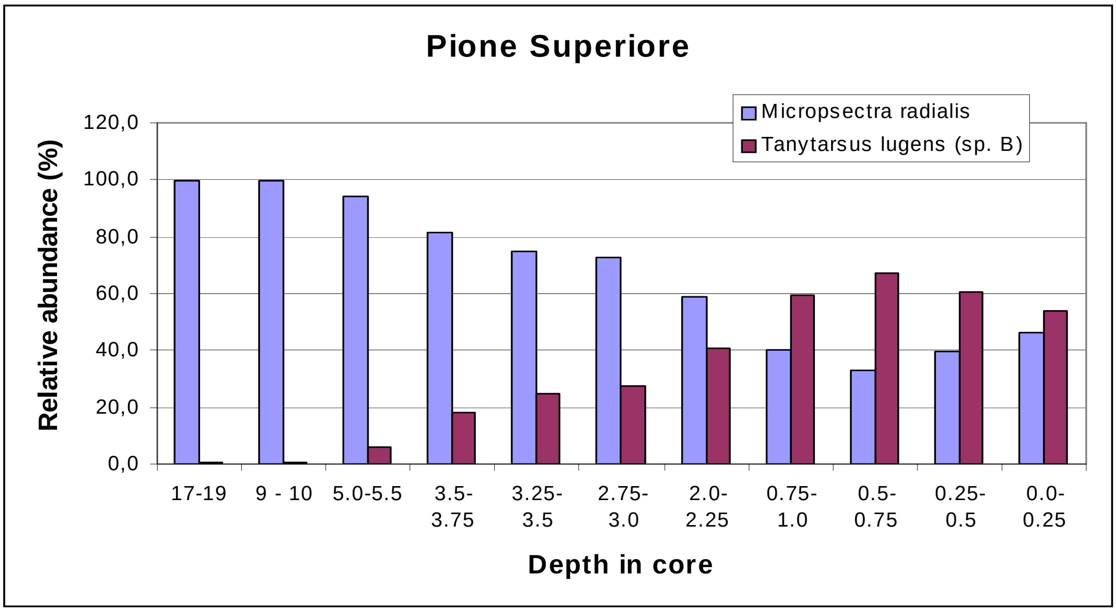


Figure 2. *Relative abundance of Micropsectra radialis and Tanytarsus sp. B in Pione superiore, Italy (from Wathne et al. 1995).*

The contemporary fauna sampled during the 1990s in Pione Superiore and Pione Inferiore contained 3 and 7 sensitive species, respectively. Pione Inferiore contained the most sensitive species indicating no acidification problem, while such species only occasionally were observed in Pione Superiore. The reason for this can be the slightly more acidic water in Pione Superiore, but the higher altitude of this lake will probably be of significance for the invertebrate assemblage. However, the published data are insufficient to evaluate trends attributed to changes in acidification.

5.4 Spain

Core analyses from Lake Redo (ES01) in Pyrénées mountains in Spain, show pH 6.8 at the base (before the industrial revolution), and pH 6.3 at the top of the core. The pH reduction has mainly taken place after 1940. In this period also the concentrations of heavy metals increased, indicating industrial pollution. However, the lake has not reached levels where damages on invertebrates are detected or expected.

Lake Redo has been investigated during AL:PE I, II, Molar and Emerge. The lake has a rich invertebrate fauna consisting of 72 identified species. Eight of the species recorded are sensitive or highly sensitive to acid water, like *Radix peregra* and *Baetis alpinus*. The lake is regarded as not acidified and no damage on the fauna could be detected (Wathne *et al.* 1997).

6. Trends in biological recovery

6.1 Canada

In Canada biological monitoring has been conducted in different lakes. During the first years of the monitoring (1987, 1990 and 1991) invertebrate investigations were carried out in the Experimental Lake Area lakes (ELA-lakes). Small damages were indicated on the invertebrate community in spring 1987 in two of the lakes, while all later samplings indicated no damages on the benthic community.

In the late 1990s, lakes in the Killarney Park, Canada, were investigated with respect to biology. These lakes have been seriously acidified especially from the smelters in Sudbury. Many of the lakes have been monitored for nearly 30 years. Some of them were among the first to show damaging effects of acidification in North America (Gunn and Sandøy 2003). Also large-scale damage to both fish and invertebrates was described in the lakes in the 1970s (Beamish and Harvay 1972, Beamish 1974, Roff and Kwiatkowski 1977). However, the sulphur emission at Sudbury has been strongly reduced during the last decades starting in the 1970s. In the 1990s a total sulphur reduction of > 90% was achieved relative to the peak levels of emissions in the 1960s (Keller *et al.* 1999).

The recovery of water chemistry in the monitored lakes has varied after the strong reduction in sulphur deposition. The lowest increase in pH is measured in the most acidified lakes, while the highest increases are found in the less acidified lakes. Biological recovery is therefore studied in lakes with different rates of chemical recovery and with different degrees of acid damage. The composition of zooplankton species shows recovery in lakes that chemically have recovered from pH < 6.0 to a pH > 6.0. Lakes with pH < 6 had a damaged species composition, while lakes that have increased pH to > 6 show no damage, or a neutral composition of zooplankton species. Also some zooplankton recovery has occurred in lakes that have not reached the status pH > 6. This recovery was, however, not typical for neutral lakes, see Holt and Yan (2003).

Recovery of an amphipod species and of mayflies has also been observed in Killarney lakes (Snucins 2003). A synoptic investigation of 119 lakes for the amphipod *Hyalella azteca* and 77 lakes for the mayflies *Stenacron interpunktatum* and *Stenonema femoratum* made it possible to state pH thresholds for these organisms. The most tolerant species, *S. interpunctatum*, had a pH threshold of 5.3, while the two other species had a threshold of 5.6. Intensive study of two acidified lakes and two reference lakes showed that the species occurred 4 - 8 years after reaching the pH threshold. After observation of the first recolonization in a small lake it took about three years before all suitable habitats were colonized within the lake. In large lakes it was suggested that the internal lake dispersal would take much more time. Snucins (2003) estimated that the time lag from pH-threshold recovery to reestablishment and occupation of all suitable habitats would be 11 to 22 or more years depending on the size of the lake. Snucins (*op cit.*) points out that the biological recovery observed in the Killarney lakes does not represent an endpoint recovery, but it shows some stages in biological reestablishment after sulphur reduction and improvements in water quality. It also gives information about the time lag of biological recovery after chemical recovery above critical limit of a species. The biological recovery described in Killarney is much in line with observations of recovery in Norway either due to reduced sulphur deposition or liming (Raddum 2003, Raddum and Fjellheim 2003).

6.2 Czech Republic

Eight small glacial lakes are situated on forested slopes of the Bohemian Forest (local names: Böhmerwald and Bayerischer Wald, or Šumava) along the border between Bohemia (Czech Republic) and Bavaria (Germany) at altitudes of about 1000 m a.s.l. Like the whole Central Europe the region was exposed to heavy atmospheric pollution. Regional deposition of S and N compounds reached up to ≈ 280 mmol m^{-2} $year^{-1}$ each in the 1980s and then declined by ≈ 80% (sulphur) and ≈ 35% (nitrogen) during the 1990s. Due to small and geologically sensitive catchments, both positive and negative trends in acid deposition were reflected by changes of lakewater chemistry and, to some extent, by biota. While the impoverishment of fauna, which took place from 1960s to 1980s, was obvious, the expected biological recovery coming along with the hydrochemical reversal has been slow (Vrba *et al.* 2003).

The first signs of zooplankton recovery took place recently in some of the Bohemian Forest lakes. Data on changes in hydrochemistry of these lakes are in Kopá ek *et al.* (2002).

Ceriodaphnia quadrangula was reported from the lake erné (Schwarzer See) at the end of the 19th century (Fri and Vávra 1896), under name *Ceriodaphnia pulchella*. In 1935-1936 it was, together with *Cyclops abyssorum* (now extinct in the lake), the main component of zooplankton both in the open water and along the rocky shores (Šrámek-Hušek 1942). Between 1947 and 1969 the species was still present in the lake, but in 1979 after great sampling effort only two specimens were found. None were found in the period 1980-1992, but in 1994 the species appeared in the littoral and in 1997 in the open water (Fott *et al.* 1994, Kohout 2001, Vrba *et al.* 2003, Fott *et al.* in press). At present it occurs regularly along the rocky shores. In the open water it occurs mainly in late summer and in the autumn. (Table 2, Figures 3 and 4).

Zooplankton of the lake Prášilské (Stubenbacher See) was sampled by Fri in June 1871 (Fri 1872). He reported *Ceriodaphnia quadrangula* to be common along the shores. Surprisingly almost no attention was given to zooplankton of the lake in the following period of more than 100 years, until the research oriented to lake acidification began. While the key zooplankton species (*Cyclops abyssorum, Daphnia longispina* s. lat.) survived the period of highest acidification in the 1980s (Fott *et al.* 1994), *Ceriodaphnia quadrangula* was missing in both the littoral and the open water. However, Pražáková (unpublished) found remains of *Ceriodaphnia* in the sediment record, which confirmed the early report of Fri (1872). *Ceriodaphnia* was not found in the lake during the monitoring programme in the years 1997, 1998 and 1999. In October 2002 the species was found in samples from the littoral, and in 2003 it neappeared both in littoral and the open water.

Ceriodaphnia quadrangula is a species rather tolerant to acid stress. Its absence in the lake erné coincided with the extremely high values of ionic (monomeric) Al which reached concentrations of about 700-800 µg l^{-1} in 1988-1989 (Fott *et al.* 1994). It is likely that the population never died out completely in the lake so that it could respond immediately to the hydrochemical reversal in the 1990s. In the lake Prášilské the maximum concentrations of ionic Al were lower (about 250 µg l^{-1} in 1988–1989) which makes the temporary absence of *Ceriodaphnia* less clear.

The beginning of zooplankton recovery in the lake Plešné (Plöckensteiner See) seems to be manifest by the increase in numbers of pelagic rotifers. The increase covered 2-3 orders of magnitude during the decade 1990-1999 (Kohout 2001, Vrba *et al.* 2003, Kohout, unpublished results). The most abundant species are: *Keratella serrulata, Brachionus sericus, Collotheca pelagica, Synchaeta tremula* (Figure 5).

Table 2. *Presence of key crustacean species in the open water of the lakes erné and Prášilské, 1871 – 2003. The grey column indicates presence of remains in the sediment (note that Holopedium and Cyclops do not leave remains).*

REFERENCE	(7), (10)	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)
SAMPLING PERIOD	Sediment	1871	1892	1935	1947	1960	1969	1979	1980	1999
	record		1896	1937		1961			1997	2003
ERNÉ lake										
Ceriodaphnia quadrangula	X	0	X	XX	X	X	X		0	X
Daphnia longispina s.lat.	X	XX	X		0	0	0	0	0	0
Holopedium gibberum		XX	X	0	X	0	0	0	0	0
Cyclops abyssorum		XX	X	XX	X	?	0	0	0	0
PRÁŠILSKÉ lake										
Ceriodaphnia quadrangula	X	XX				0		0	0	X
Daphnia longispina s.lat.	X	X				X		X	X	X
Cyclops abyssorum		X				?		X	X	X

X:present, XX: present in high abundance, : found in 1-2 specimens, 0: not found, ?: species status unclear. (1) Fri 1872, (2) Fri and Vávra 1896, (3) Šrámek-Hušek 1942, (4) Weiser 1947, (5) Procházková and Blažka 1999, (6) Ošmera 1971, (7) Fott and al. 1994, (8) Vrba and al. 2003, (9) Fott and al. in press, (10) Pražáková unpublished.

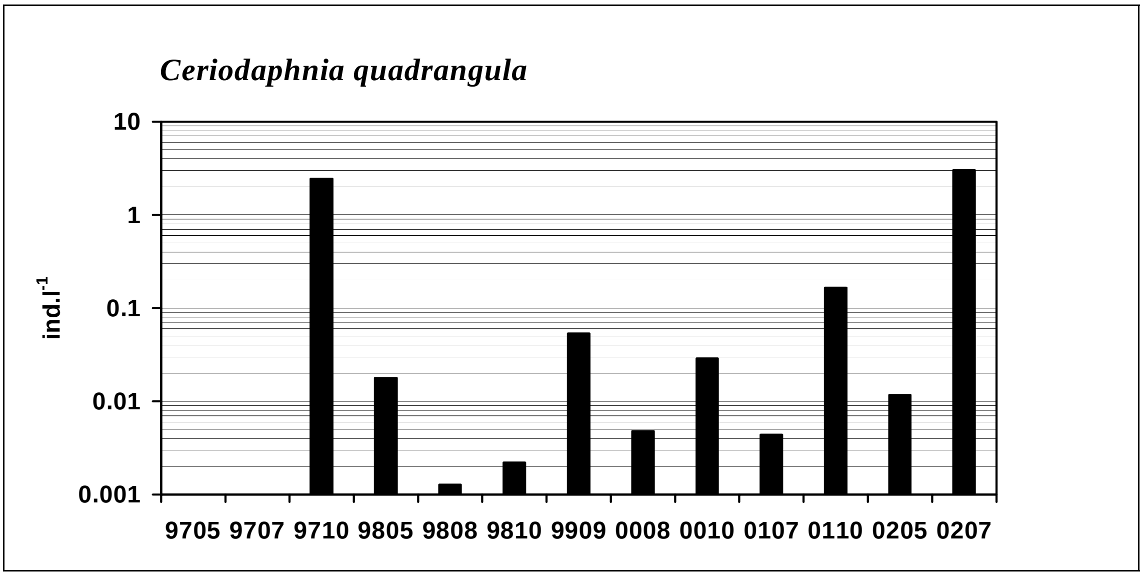


Figure 3. *Abundance of Ceriodaphnia quadrangula in the open water (average column densities) of the lake erné. In the years 1980-1996 no Ceriodaphnia were found, while the species occurred again in fall 1997. Date of sampling in the form YYMM. Units: individuals per litre.*

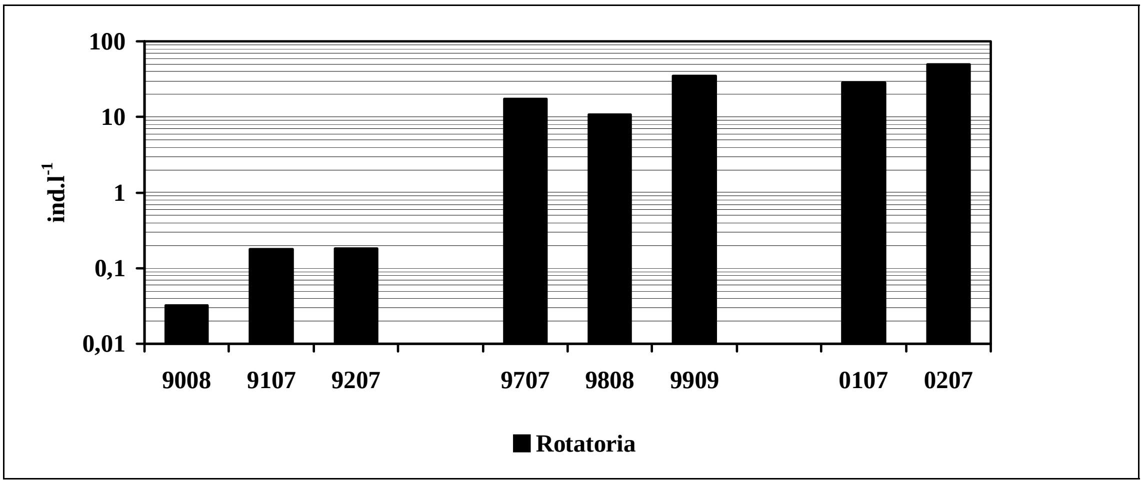


Figure 4. *Abundance of rotifers in summer plankton of the lake Plešné, 1990-2002 (average column densities). Date of sampling in the form YYMM. Units: individuals per litre.*

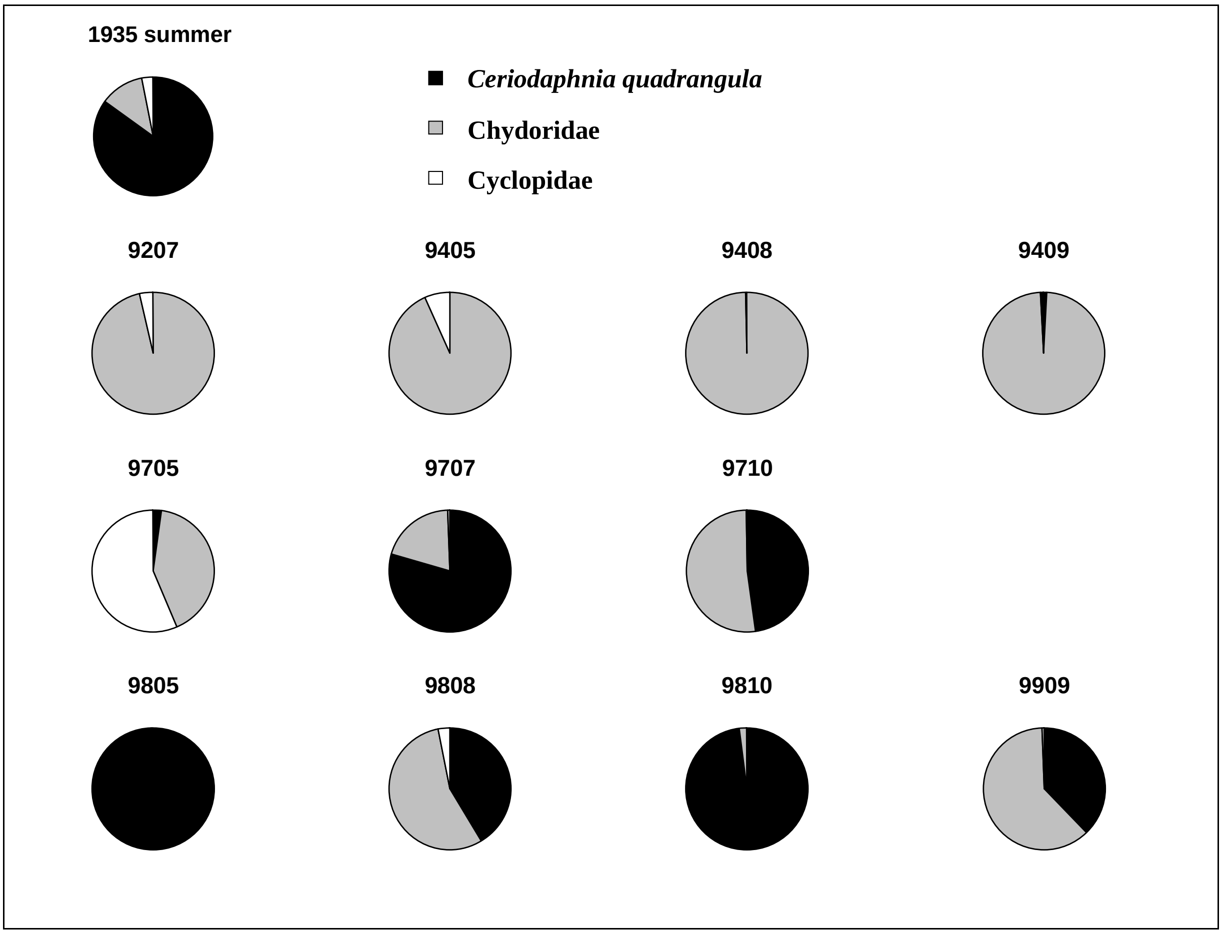


Figure 5. *Relative abundance of Ceriodaphnia quadrangula in the littoral of the lake   erné. In the years 1982-1992 Ceriodaphnia were not found in the littoral. Data from Šrámek-Hušek (1942),   ástková (1995) and Pražáková (unpublished). Sampling time in the form YYMM.*

6.3 Germany

The acidic status of sites in Germany is based on invertebrates. Four different categories are used according to the description given by the Bayerischen Landesamtes für Wasserwirkschaft (1999). The method is named "Säuerezustandsklasse" SZKL. The categories are:

1. Not acidic. This means that pH is mostly > 6.5 and minimum pH is seldom below 6. The most sensitive species are present.
2. Slightly acidic. pH drops can occur, but seldom below 5.5. The most acid sensitive species are absent.
3. Clearly acidic in periods. pH is normally < 6.5, but seldom below 4.3. During low-water periods, for example in summer, pH can increase to neutral levels. Fish populations are damaged. Only acid tolerant species are present.
4. Chronically acidic. pH is usually below 5.5 all year. Minimum pH during snowmelt or heavy rain is < 4.3. Only a few of the most acid tolerant indicator species are present. pH is fatal for sensitive fish.

Table 3 gives an overview of German ICP sites, indicating the acid status at the start of the monitoring and at the last year with biological data. The period with observations vary from a few years to 12 years.

Table 3. *Sites in Germany with time series of biological data and corresponding water chemistry. Changes in pH development for the spring /fall period are indicated. + = slight increase in pH, - = slight decrease in pH, no symbol = no clear trend in pH.*

	SiteName	Nationa Designation	acid category at start of the monitoring		acid category last sampling year		pH development
DE08	Bayerischer Wald, Grosse Ohe	Gro33	1983	2/3	2002	2	
DE10	Bayerischer Wald, Hinterer Schachtenbach	Hin31	1983	3/4	2002	3	
DE23	Bayerischer Wald, Seebach	See30	1983	2/3	2002	3	
DE27	Bayerischer Wald, Vorderer Schachtenbach	Vor32	1983	2	2002	2	
DE25	Elbsandsteingebirge, Taubenbach	Tau5	1992	3	2002	1/2	+
DE07	Erzgebirge, Grosse Pyra	Gro2	1992	4	2002	4	+
DE21	Erzgebirge, Rote Pockau	Rot3	1992	3	2002	2/4	+
DE30	Erzgebirge, Wilde Weisseritz	Wil4	1992	3/4	2002	2/3	+
DE31	Erzgebirge, Wolfsbach	Wol1	1992	1	2002	1	
DE02	Fichtelgebirge, Eger	Ege1	1989	3	2002	1	
DE18	Fichtelgebirge, Röslau	Roe2	1989	4	2002	4	
DE33	Fichtelgebirge, Zinnbach	Zin3	1989	4	2002	4	
DE06	Hunsrück, Gräfenbach	Gra5	1982	4	1999	4	+
DE26	Hunsrück, Traunbach 1	Tra1	1983	4	2001	4	+
DE14	Kaufunger Wald, Nieste 3	Nie3	1987	2	2002	2	-
DE15	Kaufunger Wald, Nieste 5	Nie5	1987	4	2002	4	
DE29	Oberpfälzer Wald, Waldnaab 8	Wal8	1986	4	2002	4	+
DE28	Oberpfälzer Wald. Waldnaab 2	Wal2	1986	2	2002	2	
DE22	Odenwald, Schmerbach 3	Sch3	1987	4	2002	4	
DE03	Rothaargebirge, Elberndorfer Bach	Elb1	1988	2/3	2002	2	
DE32	Rothaargebirge, Zinse	Zin2	1988	3	2002	3	
DE01	Schwarzwald, Dürreychbach	Due6	1987	3/4	2002	3/4	+
DE05	Schwarzwald, Goldersbach	Gol7	1986	1	2002	1	
DE11	Schwarzwald, Kleine Kinzig	Kle1	1985	2	2002	2	
DE04	Sächsische Tieflandsbucht, Ettelsbach	Ett6	1992	3/4	2002	4	+
DE09	Sächsische Tieflandsbucht, Heidelbach	Hei7	1992	4	2002	4	+
DE19	Taunus, Rombach 2	Rom2	1987	4	2002	3	

The spring period is defined as March, April and May. Observations from June are also included as spring values if no other data are available. The fall period consists of September, October and November. Most of the monitored sites show a clearly or strongly acidic status, category 3 and 4.

The composition of ions in the water determines the toxicity for biota. Most important is the concentration of toxic aluminium, which often is correlated with pH. pH can be used as an indicator of the toxicity of the water. Changes in pH signals that alternations in the aquatic communities can be expected after some delay.

Examples of mean pH development for spring and fall for eight of the monitored sites are shown in Figure 6. In Duerreychbach (DUE6, DE01) in the Black Forest a slight increase in pH is indicated, starting in the range of 5 - 5.5 and ending about 0.5 units higher in 2000. During the two following years pH has dropped. In spite of this the increase was pronounced and most clear for the spring period. The biological response to this improvement was still small, and the stream is still acid category 3/4.

Elberndorfer Bach (ELB1, DE03) in Hunsrueck, showed a more or less unchanged pH over > 15 years. Spring pH was usually been around 6, while fall pH was mostly in the range 6.5 - 7. The site had acid category 2/3 in the start of the monitoring, and ends in category 2 in 2002. More individuals representing this category occurred during the last years, indicating stabilisation and strengthening of this slightly acid affected community.

In Graefenbach (GRA5, DE06), Hunsrueck (not shown), pH increased in second half of the 1990s and up to 2002. However, the improvement is still within a strongly acidified range of the pH scale. No recovery of sensitive species is expected before the improvements stabilise above critical limits of sensitive species. The unchanged category 4 is therefore so far in accordance with the development in the water chemistry.

High variation in pH was recorded in Heidelbach (HEI7, DE09), Dahlener Heide. In 1992 both spring and fall pH was < 4. Later, the fall pH varied from 5 to 7, while the spring values, with one exception, stayed around 4. The improvements during fall indicate positive development, but because of the low spring levels as well as high variability in pH, sensitive species have not been able to recover during such conditions. A stabile acid category 4 was therefore observed.

Hinterer Schachtenbach (HIN31, DE10), in East Bavaria, has also relatively unchanged pH since 1983 for spring and fall. The spring pH was mostly in the range 5.5 to 6, but the values have been reduced after 1999 and were below 5 in 2002. The fall values were higher and in most cases > 6. The acid category was about 3/4 in 1983 and 3 in 2002.

Another site with relatively good pH conditions during the 1990s is Grosse Ohe (GRO33, DE08). However, the acidity increased again after 1999. The pH varied from 5 to 6.5 during spring, while the fall values have been from 5.5 to 6.8. Also in this locality the invertebrate community has been quite stabile with regard to sensitive species, indicating an acid status between category 3 and 2 in the start, but ending at 2 in 2002.

Vorderer Schachtenbach (VOR32, DE27), (not shown) had a similar pH development as Grosse Ohe and Hinterer Schachtenbach with most of the measurements between 6 and 7. The spring values indicated increased acidity after 1999 with a mean pH of 5 in 2002. The acidification status based on invertebrates varied around 2 during the first part of the monitoring. During the last part of the 1990s and after 2000 the acidification is more stabile around category 2.

The remaining examples are Schmerbach (SCH3, DE22) in Odenwald, Waldnaab (WAL8, DE29) in East Bavaria and Wolfsbach (WOL1, DE31) in Erzegebirge. In Schmerbach the mean pH was < 3.5 during spring and not above 4.5 either for spring or fall. The acid category 4 has been recorded through the whole period. The fauna is seriously damaged. In Waldnaab pH varied between 4 and 6.5 during fall and between 4 and 7 during spring. Acid sensitive species will have problems in survival when exposed to such variations. The acid category 4 has been the main status for this locality during the monitoring, but occasionally more sensitive species have occurred around 2000, resulting in a temporal increase in the acid category. However, in 2002 the sensitive species were not recorded and the site was in category 4.

The last example is Wolfsbach, a not acidic site. This locality contains the most sensitive species giving the acid category 1. The pH during fall was generally > 7, while the spring values usually have been between 6.5 and 7. The lowest measurements were observed at the start of the monitoring, 1992-1994, and have not changed. Several of the most sensitive species are present in this locality.

There are some signs of recovery of sensitive species in several of the German sites, but no clear stabile significant recovery. The small positive changes that have been detected can after some time turn out to be significant. The highest potential for significant improvements seems to be in the eastern parts of the country. The general lack of clear trends so far in the German biological data, corresponds mostly with the results from the chemical analyses, see Skjelkvåle *et al.* (2003). The sites in central and western Germany show no change in mean pH.

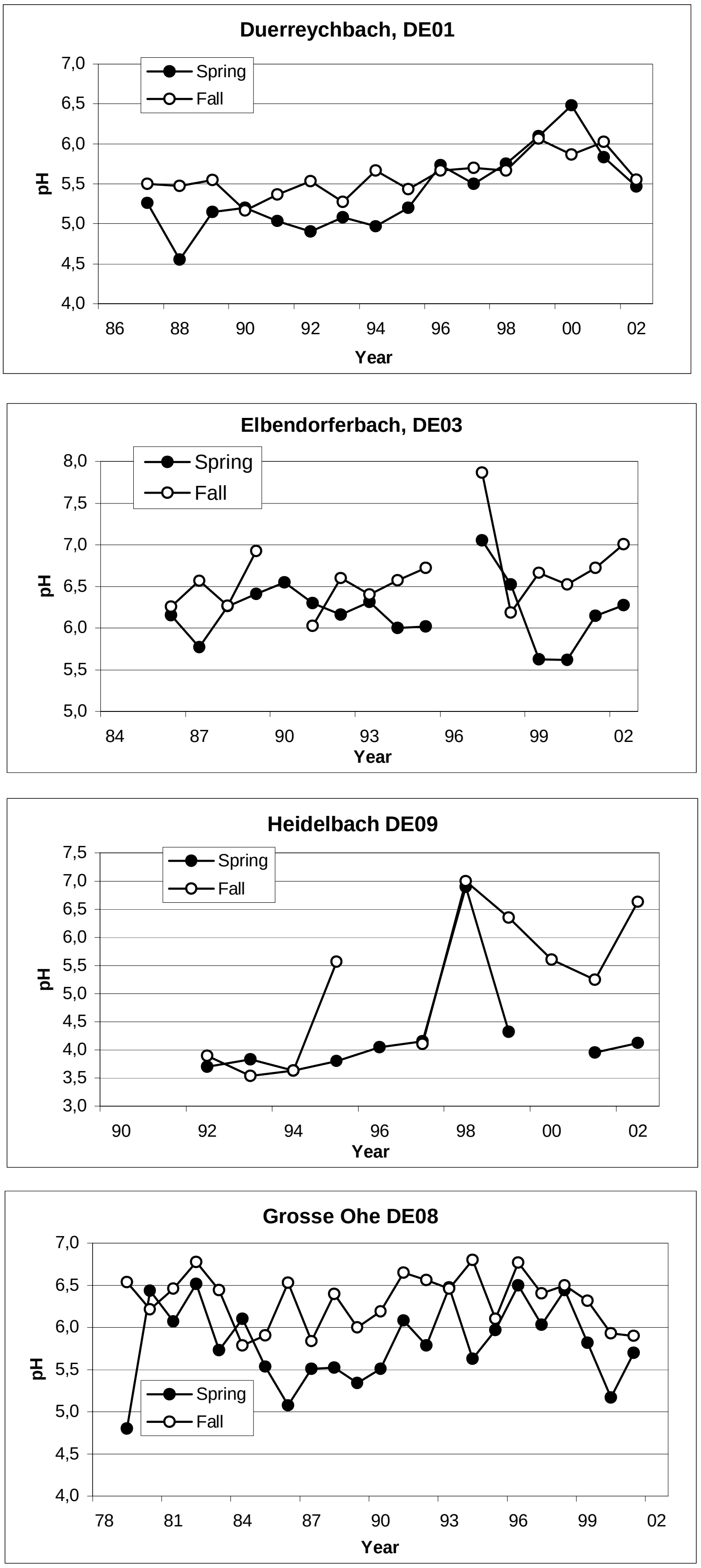
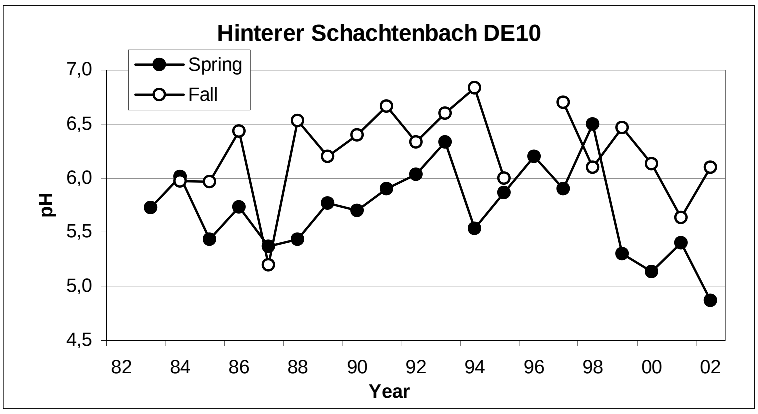
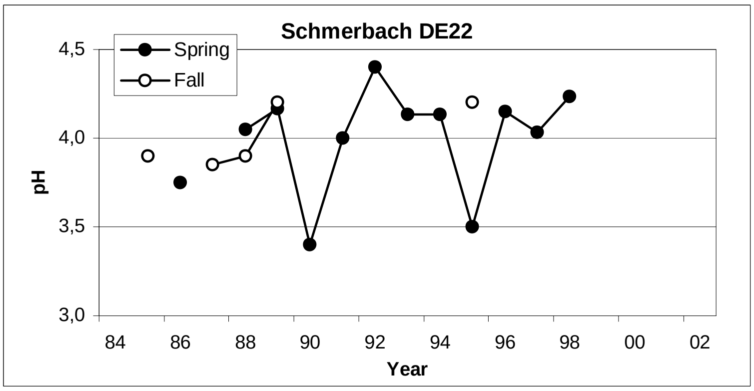
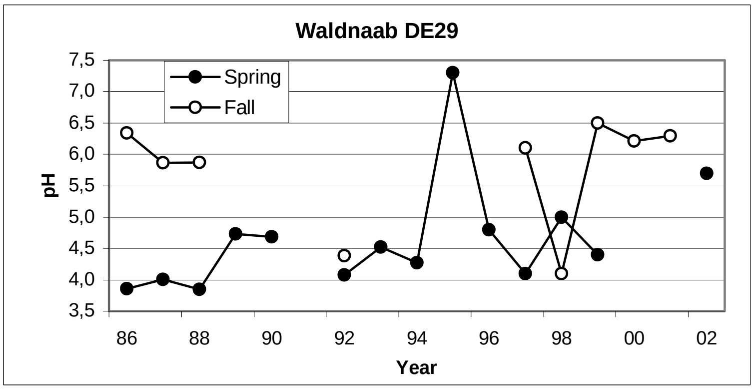


Figure 6. *Median spring and autumn values in German ICP Waters sites.*







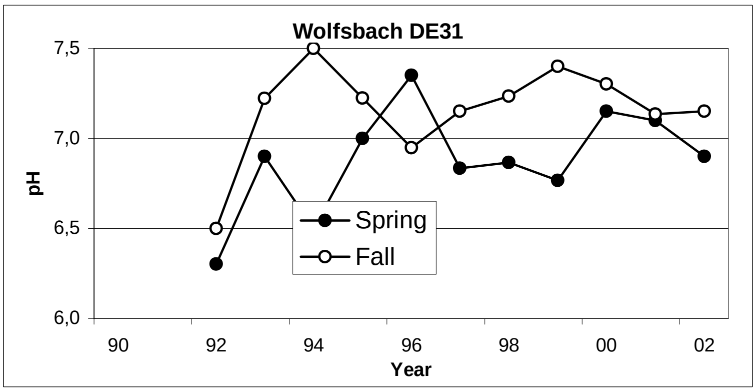


Figure 6. cont.

6.4 Norway

The Norwegian monitoring programme includes 100 lakes in pristine areas sampled every fourth year for biology, and five rivers with sampling every spring and fall. The river monitoring began in the early 1980s and the data are reported to ICP Waters. Chemical and biological recovery in several of these sites has been significant during the last decade. Here we will illustrate the recovery of sensitive invertebrates through different methods including:

- the mean acidification index, which is especially useful in the pH range 4.7 - 6.0 where most sensitive species have their critical limits, see Raddum (1999)
- numerical development of selected sensitive species
- multivariate statistical analyses. These analyses are also useful at pH below 4.7 and above 6.0, sections of the scale where the acidification index looses power.

6.4.1 Trends in recovery based on acidification index

The mean acidification index for the watersheds Farsund, Vikedal, Gaular and Nausta is presented in Figure 7. In Farsund the index indicated a strongly acidic watershed during the 1980s. The index started to increase in the beginning of the 1990s and was highest in 2002. The trend in acidification index for fall and spring after 1991 had a r^2 of 0.84 and 0.70, respectively, and corresponds with the general improvements in water quality of the region of Farsund (Skjelkvåle *et al.* 2003). In Vikedal a similar improvement was recorded. Compared with Farsund, Vikedal was less acidified during the 1980s, but high variation in the index occurred during that period. However, after 1990 the index has shown a more or less stabile increase both for spring and fall and the trends have r^2 of 0.73 and 0.84, respectively.

Further north the acidification has not been as strong. Gaular and Nausta had a relatively high index during the 1980s (Figures 13 and 14), but these were clear damages to the invertebrate fauna. In Gaular improvements of the index also started in the early 1990s. From 1990 the positive trends have r^2 of 0.77 and 0.58 for spring and fall, respectively.

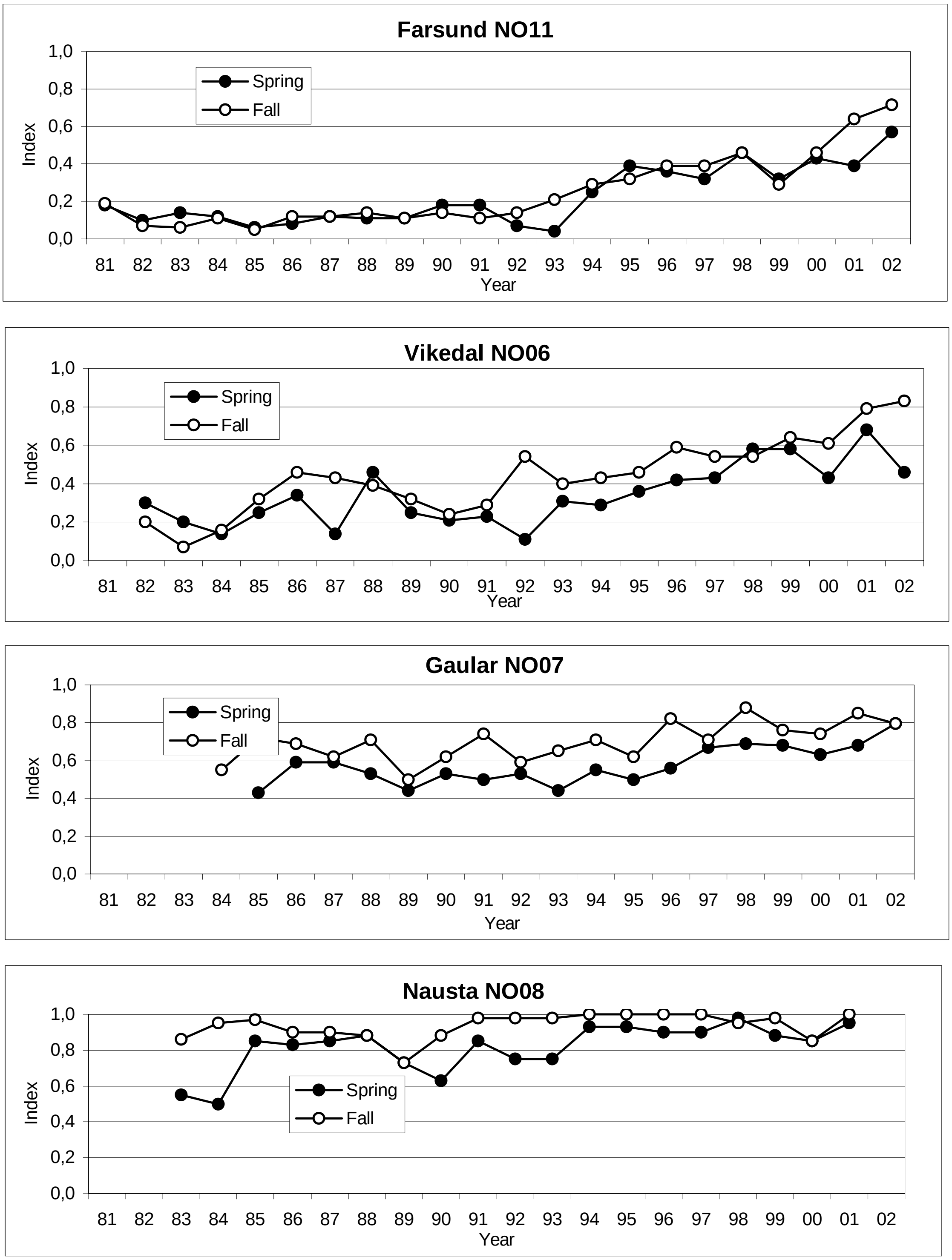


Figure 7. *The invertebrate acidification index in Farsund, Vikedal, Gaular and Nausta during 1981 - 2002.*

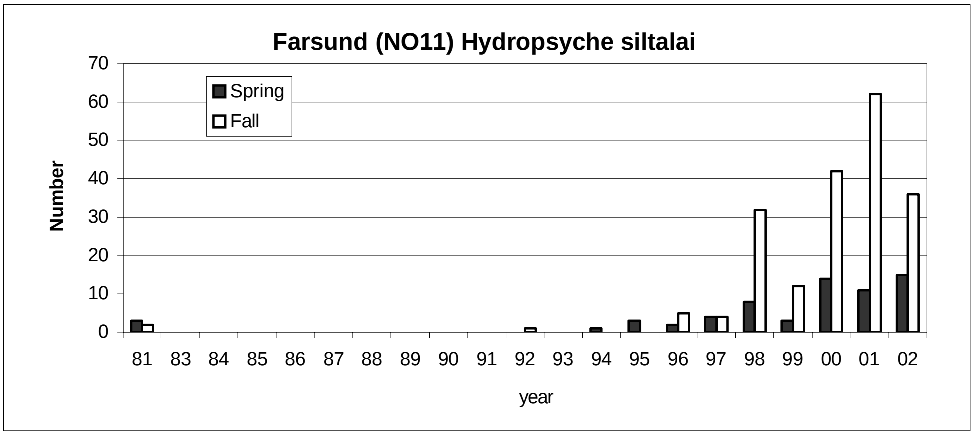
In Nausta the index indicated severe damage at the start of the monitoring followed by an improvement. The index dropped again, both for spring and fall, during the period 1988-

1990. One of the reasons for this was heavy rain and increased seasalt deposition during that period.

During the 1990s the index was around 1 in fall, while the spring values were lower, increasing from 0.65 in 1990 to 0.95 in 2002. The drop in 2000 was connected to a new, but shorter period with heavy rain and increased deposition of seasalts. Reduced deposition of sulphur will reduce the damaging effects of seasalts. It is possible that the seasalt episodes during 1988-1990 caused a larger damage than the episodes in 2000.

6.4.2 Trends in recovery at the species level

Increase in abundance of sensitive species during the 1990s was observed in the monitored rivers. Examples are shown in Figures 8 - 10 for Farsund, Vikedal and Nausta. Both the water chemistry (Skjelkvåle *et al.* 2003) and the invertebrate assemblages showed significant improvements. With respect to chemistry, the south Nordic region has the highest reduction of sulphur over the last decade (Skjelkvåle op.cit.) which corresponds with the improvements in biology. The Nordic region is among those with the thinnest soil cover which seems to be a key factor both for rapid chemical and biological recovery.



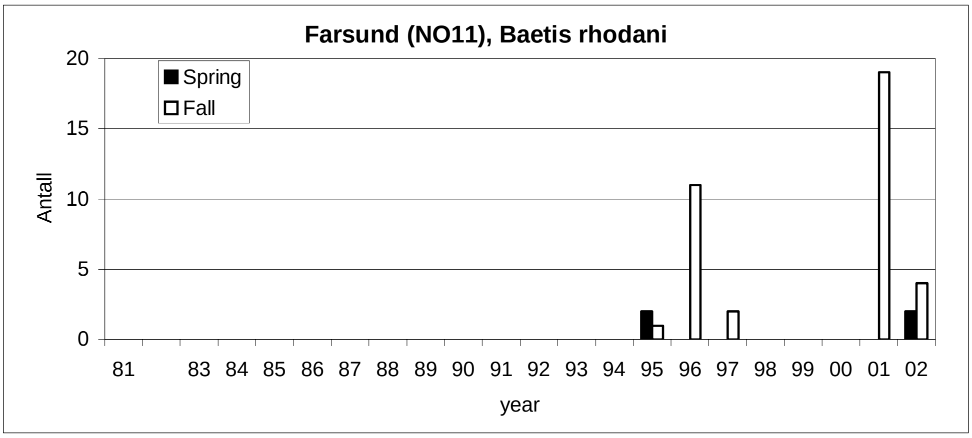


Figure 8. *Development of H. siltalai and B. rhodani in Farsund during the period 1981 - 2002.*

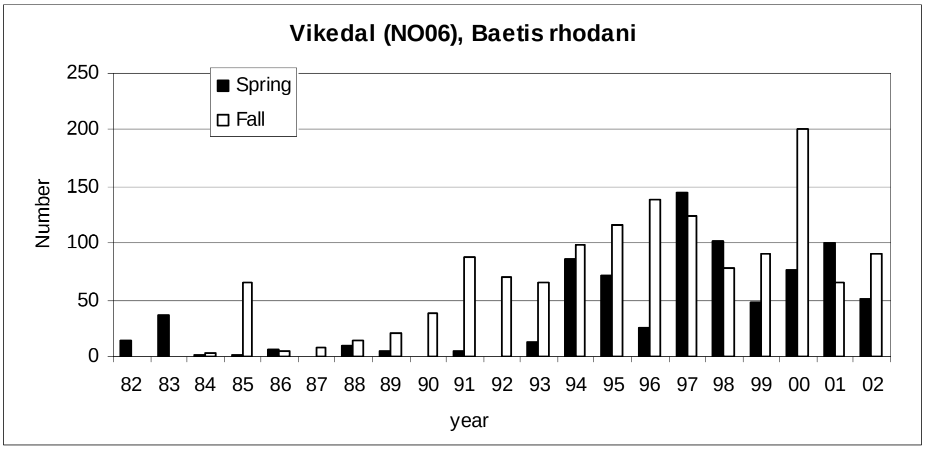
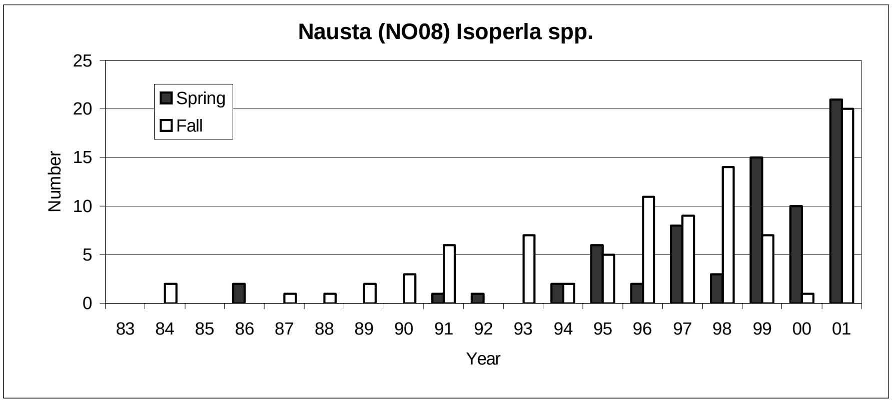


Figure 9. *Development of B. rhodani in Vikedal during the period 1982 - 2002.*



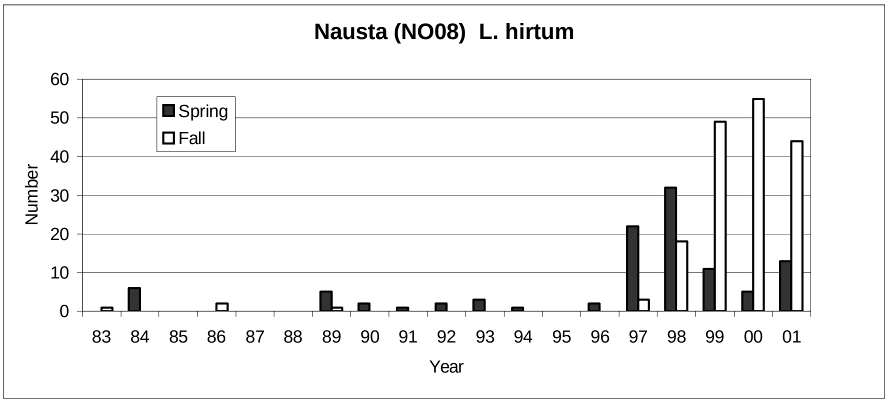


Figure 10. *Development of Isoperla spp. and L. hirtum in Nausta during the period 1983 - 2001.*

6.5 Sweden

The mean acidification index for the Swedish sites has been estimated for the period 1985-1989, 1990-1995, 1997, 1998, 1999 and 2000 based on annual samples collected generally during the autumn (Figure 11). The index for Lake Härsvatn has been 0 through the whole period. In Lake Brunnsjøn and Fiolen the index has been 0.5 and 1 since 1997, respectively, while in lake Fräcksjøn and Stensjøn a stable index 1 has occurred since 1995. The situation in Lake Storasjø has varied with an index between 0 and 0.5, while in Lake Tväringen the index has been stable at 1 over the whole period.

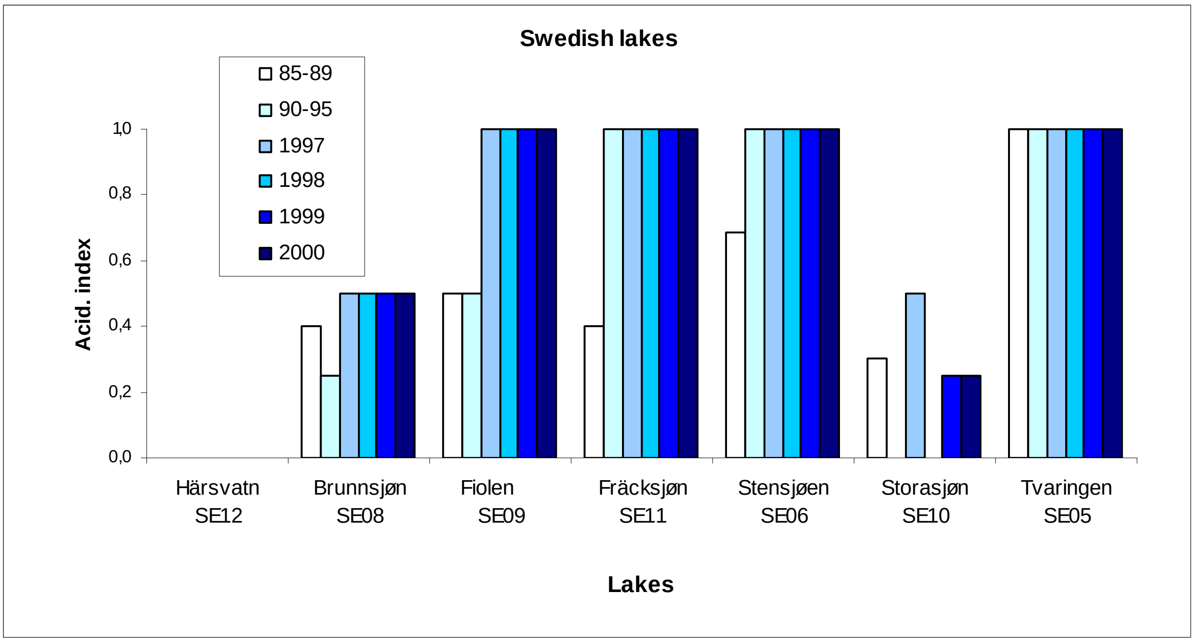


Figure 11. *Acidification indexes for Swedish lakes during the period 1985 - 2000.*

7. Multivariate statistical trend analyses

The method used is described in Skjelkvåle *et al.* (2000), and Halvorsen *et al.* (2002, 2003). We use a partial redundancy analysis (RDA) to find the amount of variation in the biological data that can be explained by linear trends in water chemistry, and a Spearman rank correlation test to explore the connections between the various water chemistry variables and linear time.

These analyses require synchronous biological and chemical data, and the biological data must be enumerated, i.e. absolute or relative density data for the different species. The longer the dataseries, the stronger is the statistical power of the analysis. Three countries have suitable data for this analysis in the ICP Waters datasets, i.e. Norway, Sweden and the United Kingdom. These three datasets were analysed in Halvorsen *et al.* (2002) for the period from 1989 to 1998. Here we include the years 1999, 2000, 2001 and 2002 to see whether additional localities show signals of recovery in the total benthic community, and also if the results from this period are consistent with the results from the first 10 years. We have no biological data from the UK for the three years after 1998 in the ICP Waters biological database, so only localities in Norway and Sweden are analysed here.

7.1 Norway

Three watersheds in western Norway, Nausta, Gaular and Vikedal, were analysed in Halvorsen *et al.* (2003). The Nausta watershed is the northernmost of these three, and the one least affected by acidification. This watershed was also the one where most localities indicated a recovery in the the total community. The watershed and the localities are shown in Figure 12.

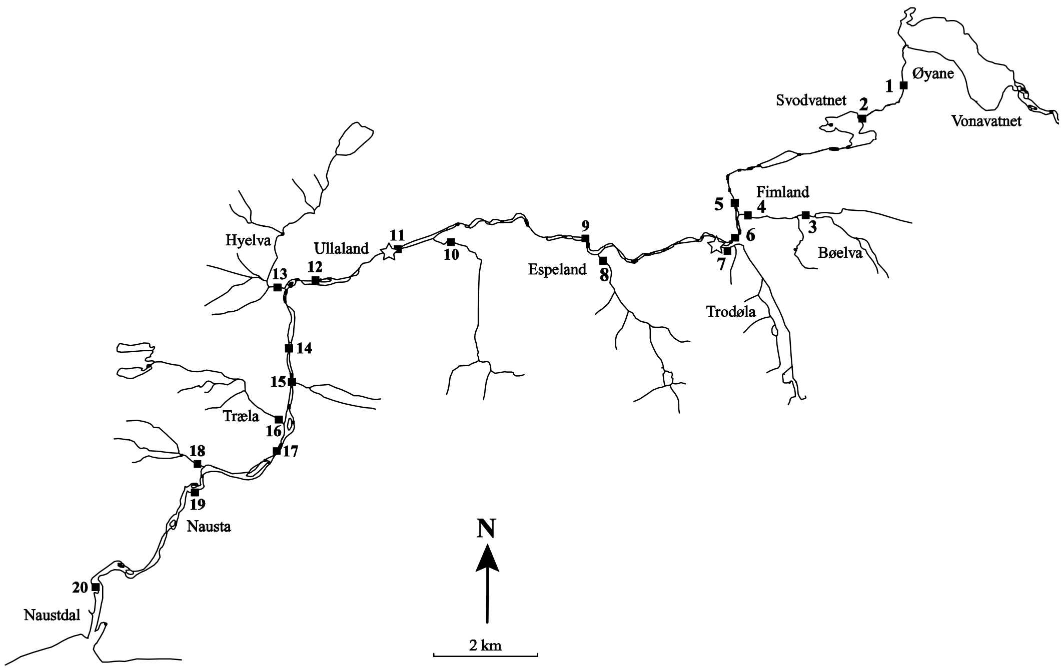


Figure 12. *The Nausta watershed. Benthic localities are shown as numbered, filled squares. Open stars indicate the sampling points for the water chemistry.*

The biological data are relative abundances from kick samples taken during spring and autumn. From 2002 benthic monitoring of this watershed changed from annually to every second year, so no data exist from 2002. The chemistry data are the means for the months March, April and May for the spring samples, and September, October and November for the atumn samples. All procedures are the same used by Halvorsen *et al.* (2002, 2003). The Nausta watershed has two localities where water chemistry is collected, one in the main river (close to locality 11, Figure 12) and one in the tributary Trodøla (locality 7, Figure 12).

The results from the analyses when the water chemistry from the main river is used are shown in Figure 13. All of the localities in the tributaries and most of the localities in the main river showed significant linear trends in the water chemistry for the 13-years period, while only the uppermost locality in the main river showed a significant trend when the period from 1989 to 1998 was analysed.

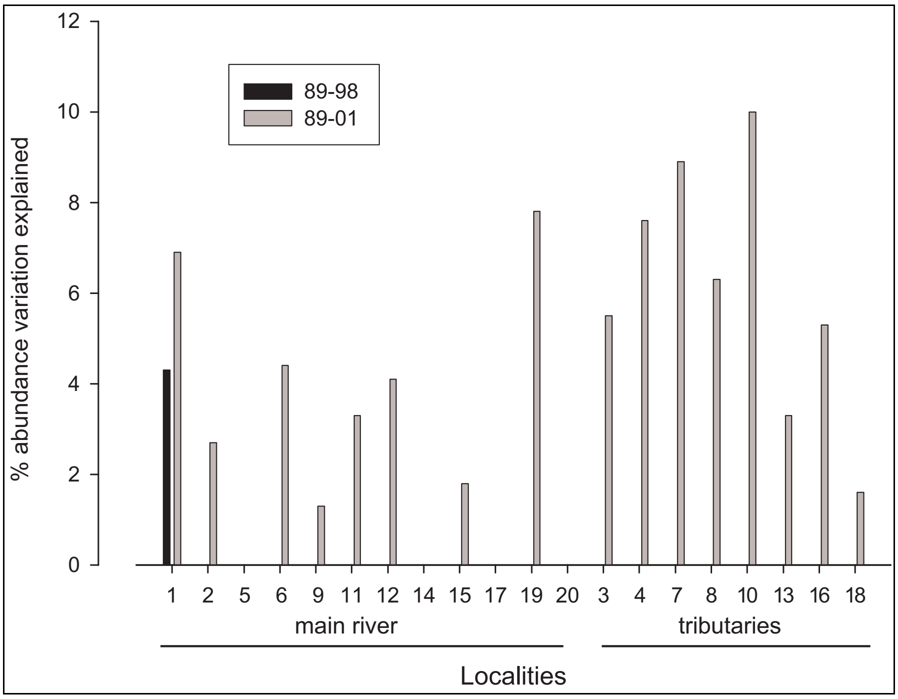


Figure 13. *Localities with indications of recovery in the total benthic community in the Nausta watershed explained by water chemistry data from the main river. The Y-axis gives the amount of variation in the abundance data that can be explained by linear trends in water chemistry.*

The correlation tests (Table 4) show a strong and significant correlation between linear time and an increase in pH and ANC, and a decrease in the concentration of labile aluminum. These trends also were identified in the 10-year period (Halvorsen *et al*. 2002, Table 4).

Table 4. *Spearman Rank Correlation matrix between the environmental variables from the main river in the Nausta watershed from 1989 - 2001 used in the RDA's. (Two tailed tests. ** significant at the 0.01 level. * significant at the 0.05 level).*

	Ca	ANC	TOC	LAl	Time
pH	0.427*	0.793**	0.465*	-0.723**	0.778**
Ca		0.450*	0.085	-0.248	0.320
ANC			0.569**	-0.700**	0.614**
TOC				-0.290	0.059
LAl					-0.703**

The results based on the water chemistry from the tributary Trodøla are shown in Figure 14. Again, all of the localities in the tributaries showed significant trends. The number of localities increased from the first 10-year period. Also the number of localities in the main river with significant trends increased. Locality 2 was the only site that showed a significant trend in the first analysis, but not when the three most recent years were included. The correlation tests between linear time and the water chemistry from Trodøla (Table 5) again indicates that the linear trends in the benthic community can be interpreted as a response to a chemical recovery in the watershed.

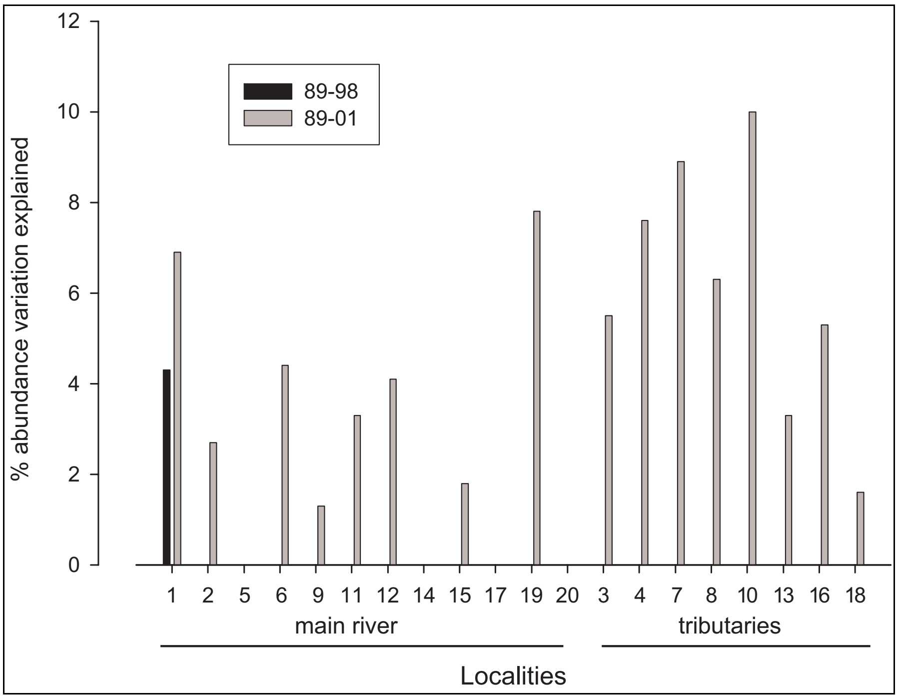


Figure 14. *Localities with indications of recovery in the total benthic community in the Nausta watershed with water chemistry data from the tributary Trodøla. The Y-axis gives the amount of variation in the abundance data that can be explained by linear trends in water chemistry.*

Table 5. *Spearman Rank Correlation matrix between the environmental variables from the tributary Trodøla in the Nausta watershed from 1989 - 2001 used in the RDA's. (Two tailed tests. ** significant at the 0.01 level. * significant at the 0.05 level).*

	Ca	**ANC**	**TOC**	**LAl**	**Time**
pH	0.631**	0.947**	0.415*	-0.925**	0.843**
Ca		0.697**	-0.001	-0.522**	0.643**
ANC			0.473*	-0.848**	0.784**
TOC				-0.337	0.258
LAl					-0.841**

The difference between the two water chemistry datasets is that calcium also shows a significant correlation with linear time in the Trodøla watershed, but not in the main river. The calcium content increased at both sampling stations and even more in the main river than in Trodøla (Figure 15b). The variation between spring and autumn samples was, however, larger in the main river and this is probably the reason why the correlation test is not significant.

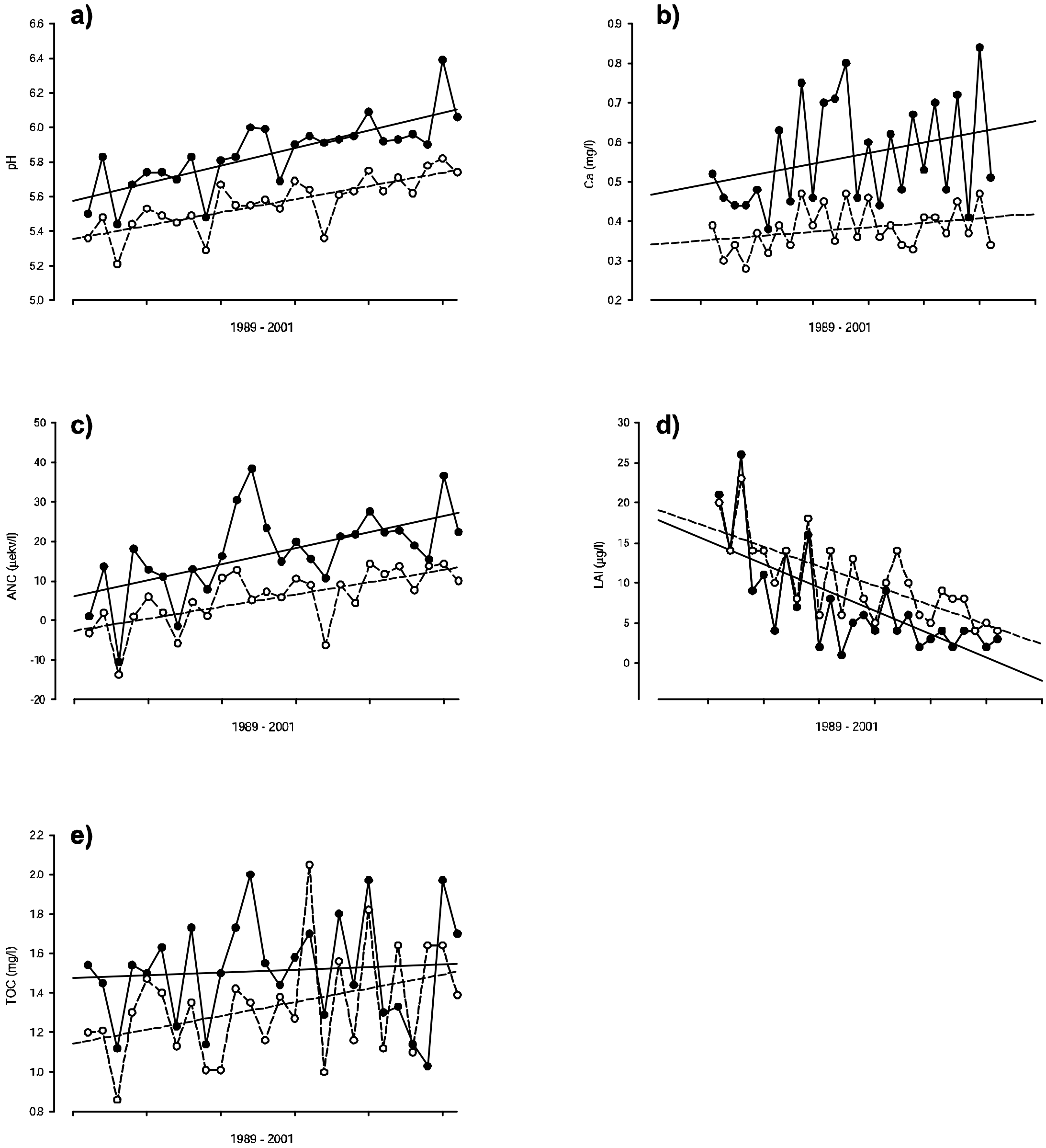


Figure 15. *Line plot and simple regressions of the averages of the water chemistry variables from the Nausta Watershed measured in spring (March, April, May) and autumn (September, October, November) and used in the RDA's. The sampling period is from the spring 1989 to the autumn 2001.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= Labile aluminum (LAl),* ***e)*** *= total organic carbon (TOC). Filled circles and solid lines represent the data from the main river (locality 11), open circles and hatches lines represent the tributary Trodøla (lcality 7)*

Figure 16 shows the Gaular watershed. The results are given in Figure 17.

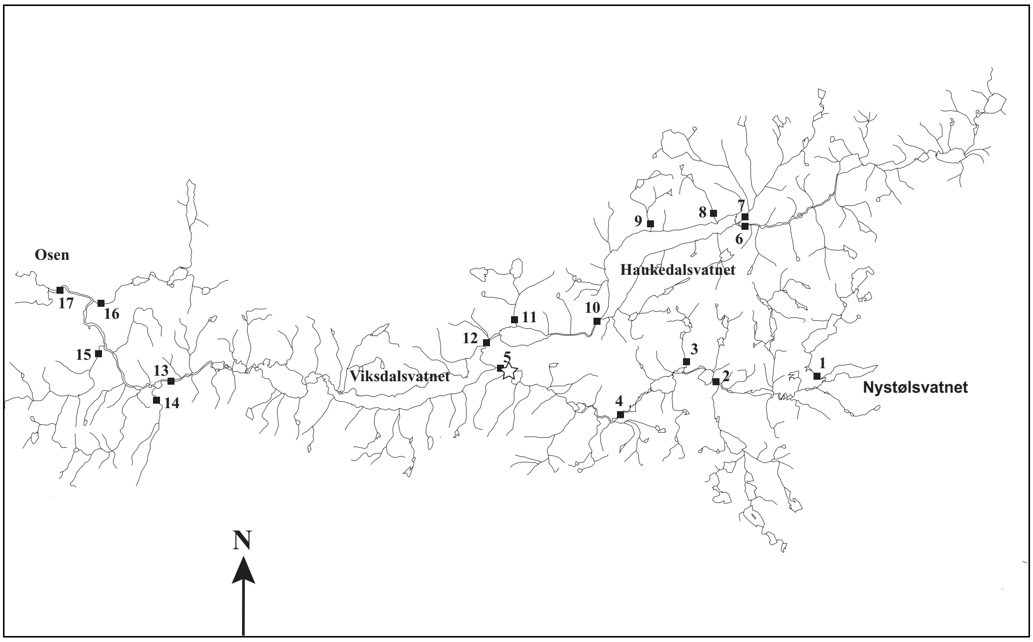


***Figure 16**. The Gaular watershed. Benthic localities are shown as numbered, filled squares. Open star indicates the sampling point for the water chemistry.*

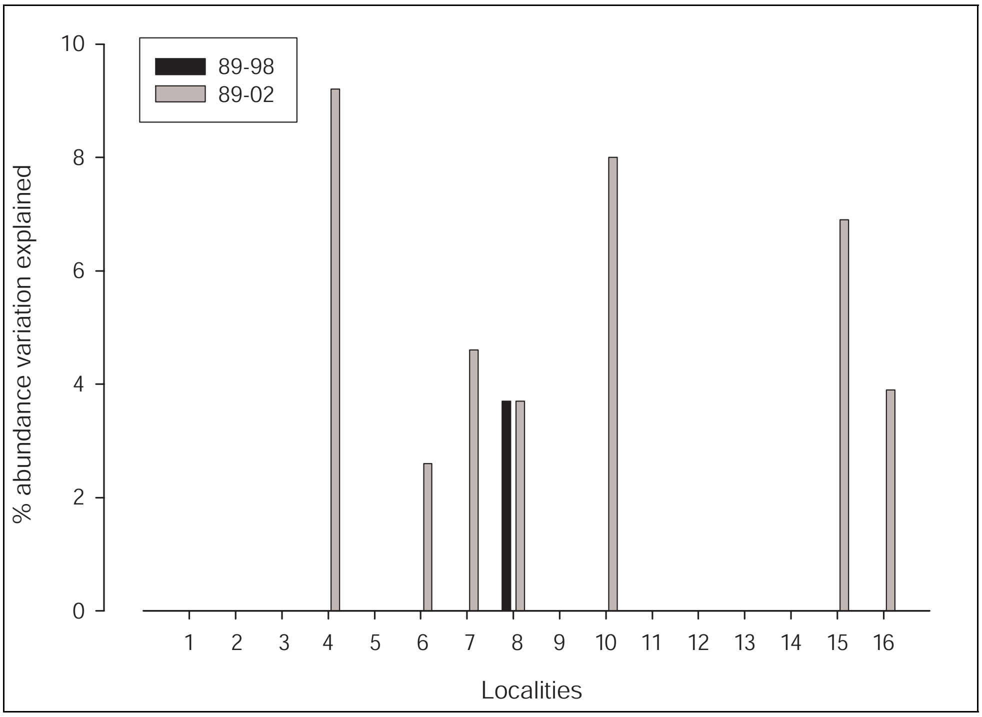


***Figure 17**. Localities with indications of recovery in the total benthic community in the Gaular watershed. The Y-axis gives the amount of variation in the abundance data that can be explained by linear trends in water chemistry.*

Only one locality, the tributary Neselva (loc. 8), showed signs of recovery in the benthic fauna from 1989 to 1998. With the four subsequent years included, recovery was found in three additional localities in the main branch of the river (localities 6, 7 and 10), one locality in the Eldalen branch (loc. 4) about 6 km upstream from the water chemistry site, and two tributaries (loc. 15 and 16) in the lower end of the watershed. Localities 6 and 10 are in the main river, while 7 and 8 are smaller tributaries. The Eldalen branch of the river is the most acidified part of the watershed (Lien *et al.* 1986). Figure 18 shows the development of the water chemistry variables from 1989 - 2002. pH and ANC are positively correlated with linear time, while labile aluminum is decreasing (Table 6).

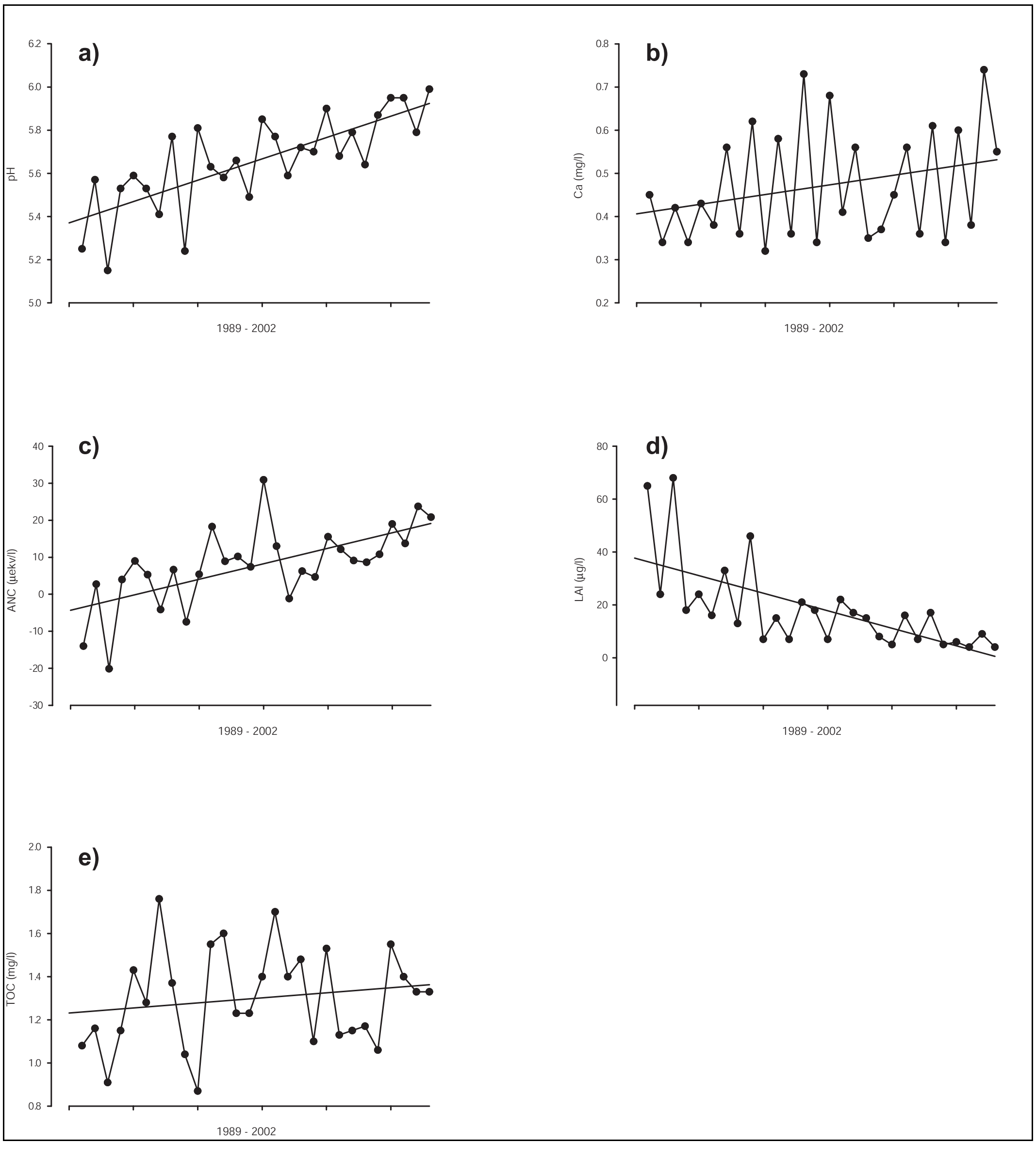


Figure 18. *Line plot and simple regressions of the averages of the water chemistry variables from the Eldalen branch of the Gaular Watershed measured in spring (March, April, May) and autumn (September, October, November) and used in the RDA's. The sampling period are from the spring 1989 to the atumn 2002.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= Labile aluminum (LAl),* ***e)*** *= total organic carbon (TOC).*

Table 6. *Spearman Rank Correlation matrix between the environmental variables from the Gaular watershed from 1989 - 2002 used in the RDA's. (Two tailed tests. ** significant at the 0.01 level. * significant at the 0.05 level).*

	Ca	**ANC**	**TOC**	**LAl**	**Time**
pH	0.036	0.785**	0.220	-0.866**	0.777**
Ca		0.333	0.243	0.119	0.245
ANC			0.431*	-0.678**	0.701**
TOC				-0.183	0.168
LAl					-0.730**

The unlimed part of the Vikedal watershed (Figure 19) shows similar results (Figure 20). Only one locality in a tributary showed recovery during the first 10-year period. With the four subsequent years included, recovery was recorded in four localities including the two in the main river closest to the water chemistry station.

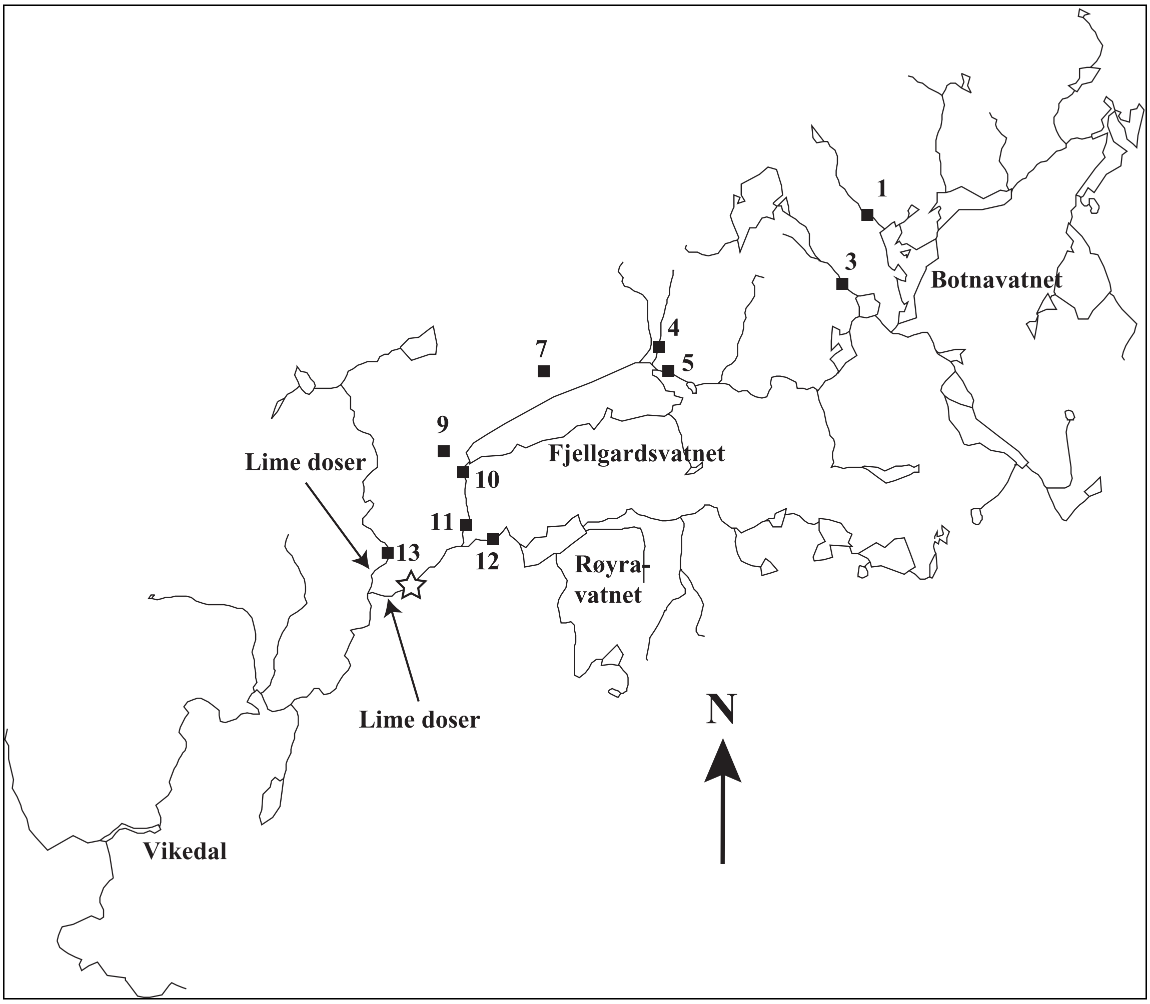


Figure 19. *The Vikedal watershed. Benthic localities are shown as numbered, filled squares. Open star indicates the sampling point for the water chemistry.*

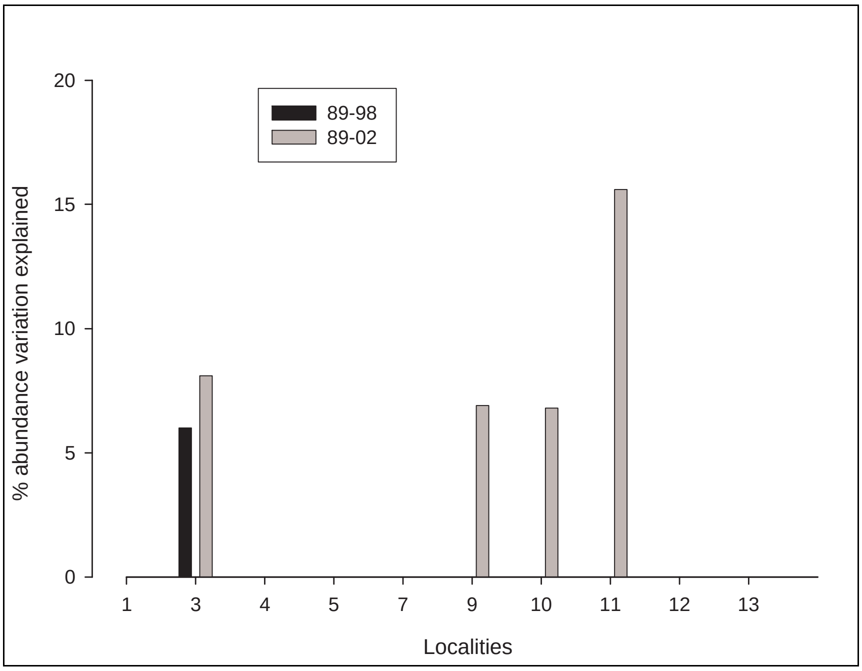


Figure 20*. Localities with indications of recovery in the total benthic community in the unlimed part of the Vikedal watershed. The Y-axis gives the amount of variation in the abundance data that can be explained by linear trends in water chemistry.*

The correlation tests (Table 7) show that pH, ANC and the calcium concentration are positively correlated with linear time, while labile aluminum is negatively correlated. The water chemistry variables for the unlimed part of the Vikedal Watershed are shown in Figure 21.

Table 7*. Spearman Rank Correlation matrix between the environmental variables from the Vikedal watershed from 1989 - 2002 used in the RDA's. (Two tailed tests. ** significant at the 0.01 level. * significant at the 0.05 level).*

	Ca	**ANC**	**TOC**	**LAl**	**Time**
pH	0.472*	0.786**	0.405*	-0.937**	0.871**
Ca		0.488**	-0.209	-0.337	0.407*
ANC			0.260	-0.789**	0.836**
TOC				-0.421*	0.268
LAl					-0.860**

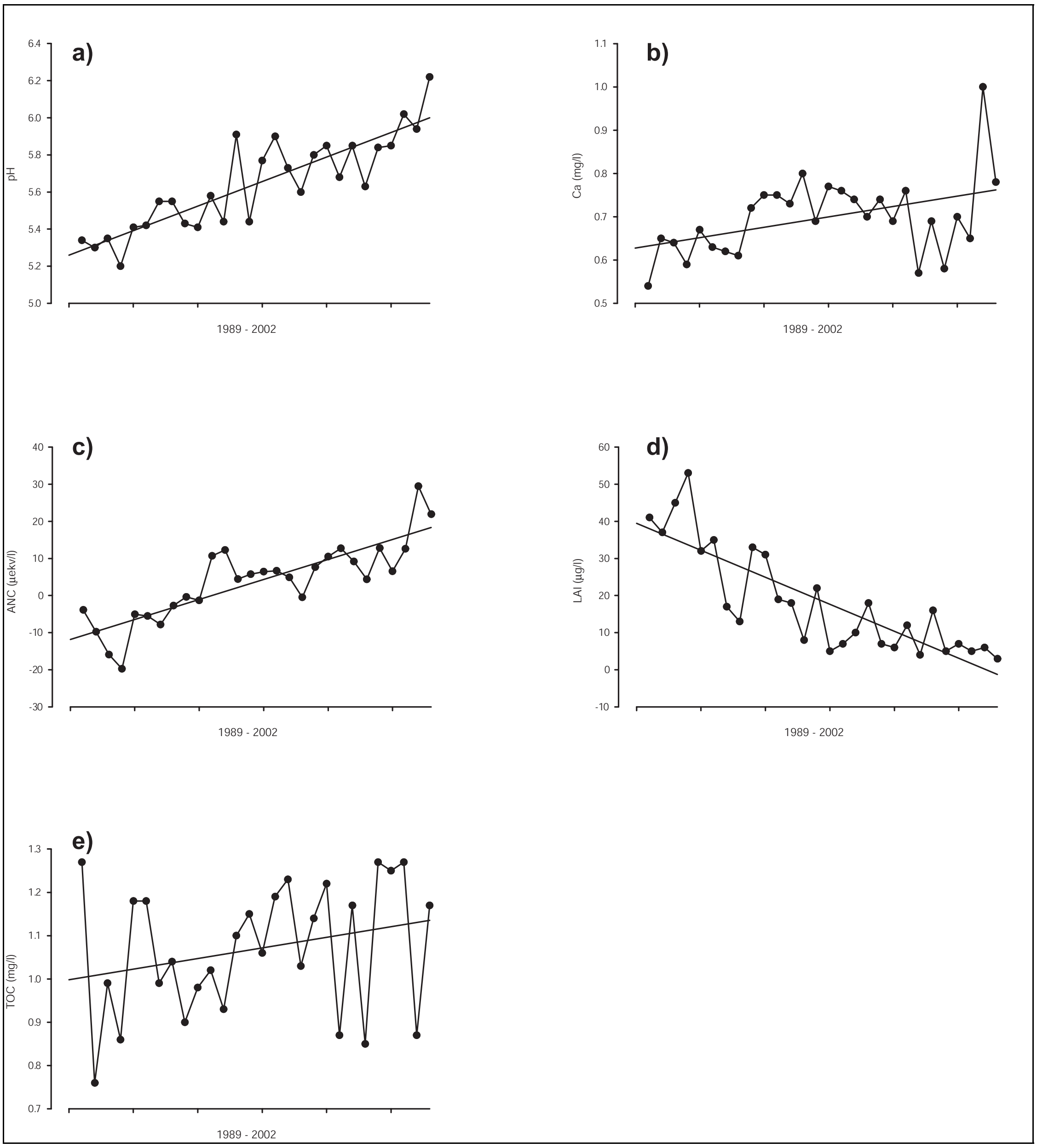


Figure 21. *Line plot and simple regressions of the averages of the water chemistry variables from the unlimed part of the Vikedal Watershed measured in spring (March, April, May) and autumn (September, October, November) and used in the RDA's. The sampling period are from the spring 1989 to the atumn 2002.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= Labile aluminum (LAl),* ***e)*** *= total organic carbon (TOC).*

All of the three Norwegian watersheds show increasing numbers of localities with linear trends of change in the total benthic community, and these trends can be explained as a response to a recovery in chemistry. This is consistent with the results based on the acidity indices, which has also been increasing in all three watersheds. The amount of variation explained by these linear changes in water chemistry is larger in almost all localities when the last three to four years of monitoring are included. Whether this is an artefact of the analysis or a demonstration of an increasing biological recovery following decreasing deposition of suphur, is for the moment uncertain. However, it will be important to monitor these watersheds both chemically and biologically for the future in order to see whether this increase in the importance of linear trends in water chemistry in ordering the benthic community will continue, until some level in water chemistry is reached when other factors will totally dominate the ordering of the benthic community.

7.2 Sweden

The Swedish material consists of quantitative Ekman grabs from the sublittoral and the profundal zone of 7 lakes, taken annually in October. The water chemistry data analysed here are the averages from monthly samples in September and October, or only the October sample when the sample from September is missing. The time series started between 1989 and 1991, and data including 2002 have been analysed.

The results are presented in Table 8. The results from the first 8 - 10 year period (Halvorsen *et al.* 2002) are also included. All of the lakes with sublittoral samples, except Lake Fiolen, showed significant linear trends in the sublittoral samples when the new data from 1999 to 2002 were included. In the profundal zone only Lake Storasjö had a significant linear trend that can be explained as a response to linear trends in acidity chemistry. However, this lake is shallow, and the sampling depth is between 4 and 6 meters, about the same depth as the sublittoral samples in the deeper lakes.

Table 8. *Results from the redundancy analyses of the Swedish ICP Waters lakes.*

	Sublittoral		**Profundal**	
	1991-1998	1991-2002	1990-1998	1990-2002
Brunnsjön	n.s.	19 %	n.s.	n.s.
	1989-1998	1989-2002	1989-1998	1989-2002
Fiolen	n.s.	n.s.	n.s.	n.s.
	1990-1998	1990-2002	1990-1998	1990-2002
Fräcksjön	n.s.	17.8 %	n.s.	n.s.
	1990-1998	1990-2002	1989-1998	1989-2002
Härsvatten	17.2 %	10.2 %	n.s.	n.s.
	1990-1998	1990-2002	1990-1998	1990-2002
Stensjön	n.s.	7.4 %	n.s.	n.s.
			1990-1998	1990-2002
Storasjö			n.s.	33.9 %
			1990-1998	1990-2002
Tväringen			n.s.	n.s.

The correlation tests indicate that there are two different types of chemistry changes in the Swedish lakes. Fräcksjön and Härsvatten show typical indications of recovery from acidification, with strong positive correlations between pH, ANC, TOC and linear time, and significant negative correlation between linear time and the calcium concentration (Table 9). These are typical trends in water chemistry associated with recovery from acidification. The development in the water chemistry variables used in the analyses are shown in Figure 22 for Lake Fräcksjön and Figure 23 for Lake Härsvatten. pH in Lake Härsvatten has increased from about 4.4 to about 4.8 during these 13 years. It is still acid, but 10.2 % of the abundance changes in the sublittoral benthic community can be explained by the linear components of change in the water chemistry variables. That is, a biological recovery following a recovery in the water chemistry can be traced. This signal of recovery was also found for the 9-year period from 1990 to 1998. The amount of change explainable by linear trends was higher for this period (17.2 %). Whether these changes can be interpreted directly as a measure of the strength of the biological recovery over time is for the moment unknown. The figure for linear change in Lake Härsvatten reported in Halvorsen *et al.* (2002; Table 5) was too high. This was due to an error in the data matrix. The error has been corrected and the analyses rerun.

Lake Fräcksjön has not been strongly affected by acidification. The data in our analysis shows a pH in the beginning of the period of about 6.2 (Figure 22 a), and this increased to about 6.4 in the autumn of 2002. However, a signal of recovery is found in the benthic community that can be explained by the linear changes in the water chemistry.

Tabell 9. *Spearman Rank Correlation matrixes among the environmental variables from 1990 - 2002 in the lakes Fräcksjön and Härsvatten used in the RDA's.*

Fräcksjön	**Ca**	**ANC**	**TOC**	**Time**
pH	-0.870**	0.739**	0.807**	0.861**
Ca		-0.817**	-0.713**	-0.865**
ANC			0.770**	0.901**
TOC				0.854**
Härsvatten	**Ca**	**ANC**	**TOC**	**Time**
pH	-0.942**	0.569*	0.816**	0.919**
Ca		-0.633*	-0.829**	-0.968**
ANC			0.425	0.709**
TOC				0.775**

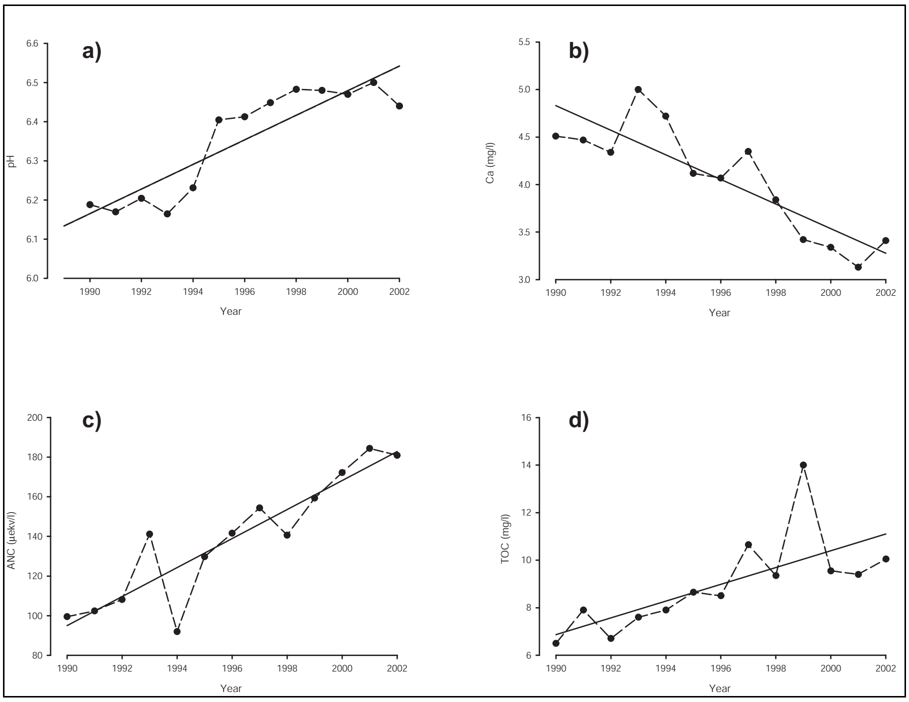


Figure 22. *Line plot and simple regressions of the water chemistry variables from Lake Fräcksjön used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

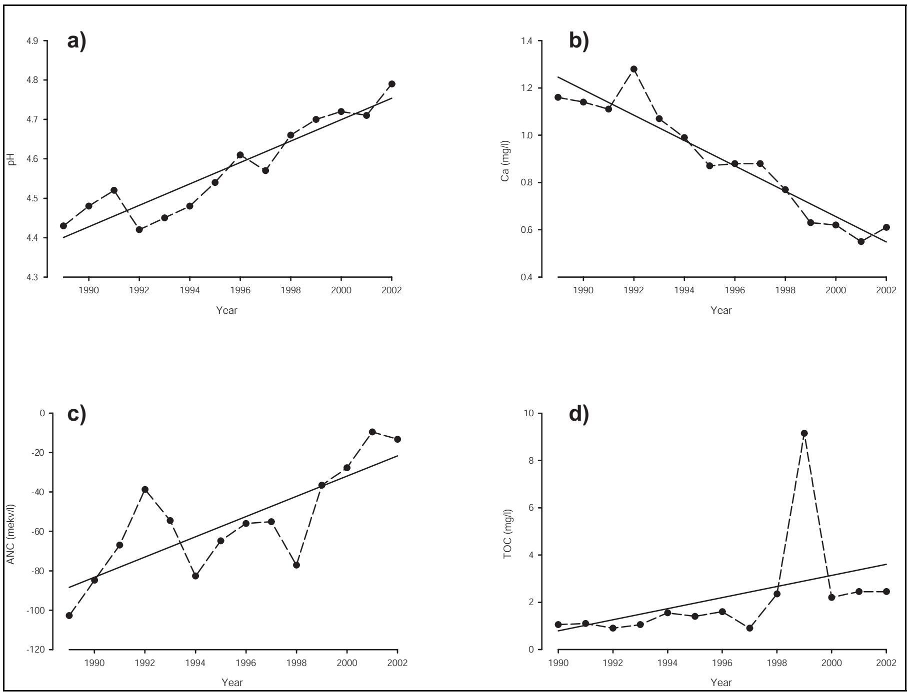


Figure 23. *Line plot and simple regressions of the water chemistry variables from Lake Härsvatten used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

The two remaining lakes with a signal of recovery in the sublittoral are Lake Brunnsjön and Lake Stensjön. Table 10 gives the results from the correlation tests. In this table Lake Storasjö is also included. This lake is shallow and the profundal samples from this lake are from about the same depths as the sublittoral samples in the remaining lakes. The correlation tests for Lake Brunnsjön and Lake Stensjön show that only a decrease in the calcium concentration correlates significantly with linear time. This means that it is only the change in Ca that potentially explains the linear trend in the benthic community. Running a RDA with the forward selection approach (a multivariate analog to multiple regression) on the water chemistry variables also shows this. It is only the Ca variable that contributes significantly to the ordination in these two lakes. Figures 24 and 25 show the water chemistry variables from Lake Brunnsjön and Lake Stensjön, respectively. A decrease in calcium has been described for lakes recovering from acidification from several regions in Europe and North America (Skjelkvåle *et al.* 1998, Stoddard *et al.* 1999, Keller *et al.* 2001). We do not imply that the decrease in the Ca content are the causal factor in ordering the benthic community. It may merely be an indirect factor that falls out as significant in our analyses, and that other factors connected to a chemical recovery from acidification are the ones having a direct effect on the benthic community. However, this decrease may also be caused by other factors like climate change (Keller *et al. op.cit.*). The linear signal in the sublittoral community of Lake Brunnsjön and Lake Stensjön must thus be interpreted with caution. The increases in ANC in both lakes and pH in Lake Brunnsjön may, however, indicate that a chemical recovery due to reduced depositions of sulphur are at least partly responsible for the linear trends seen in the sublittoral benthic community.

Table 10. *Spearman Rank Correlation matrixes among the environmental variables from 1990 - 2002 in the lakes Brunnsjön, Stensjön and Storasjö used in the RDA's.*

Brunnsjön	**Ca**	**ANC**	**TOC**	**Time**
pH	-0.627*	-0.209	-0.481	0.322
Ca		-0.102	-0.322	-0.783**
ANC			0.432	0.364
TOC				0.389
Stensjön	**Ca**	**ANC**	**TOC**	**Time**
pH	0.070	0.195	-0.669*	0.126
Ca		0.093	-0.309	-0.765**
ANC			-0.042	0.446
TOC				0.189
Storasjö	**Ca**	**ANC**	**TOC**	**Time**
pH	-0.473	0.588	0.373	0.391
Ca		-0.656*	-0.509	-0.882**
ANC			0.811**	0.679*
TOC				0.464

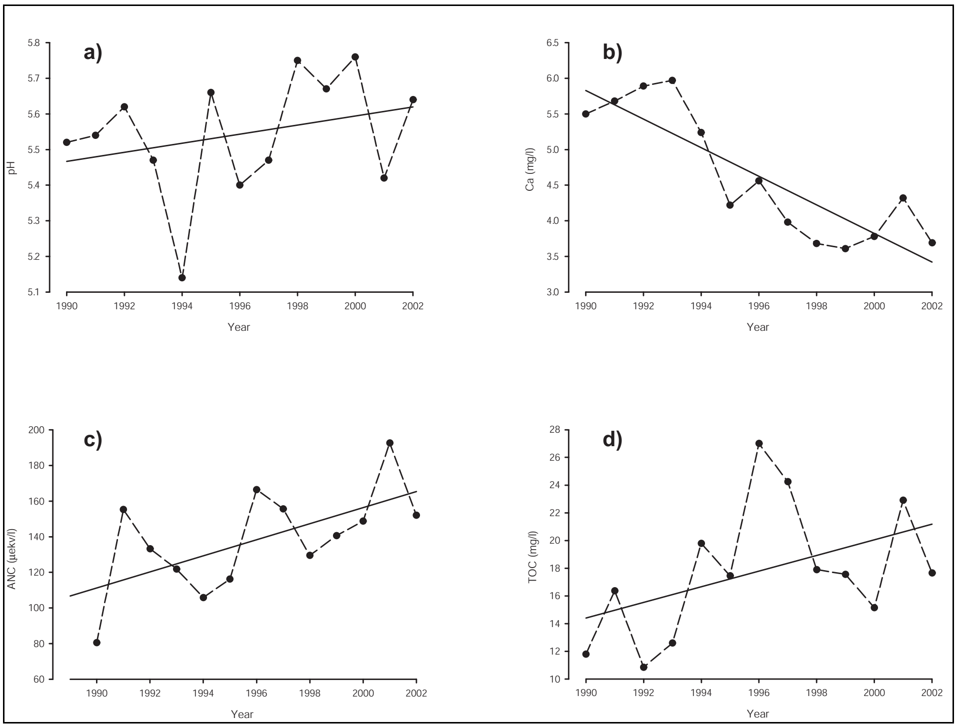


Figure 24. *Line plot and simple regressions of the water chemistry variables from Lake Brunnsjön used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

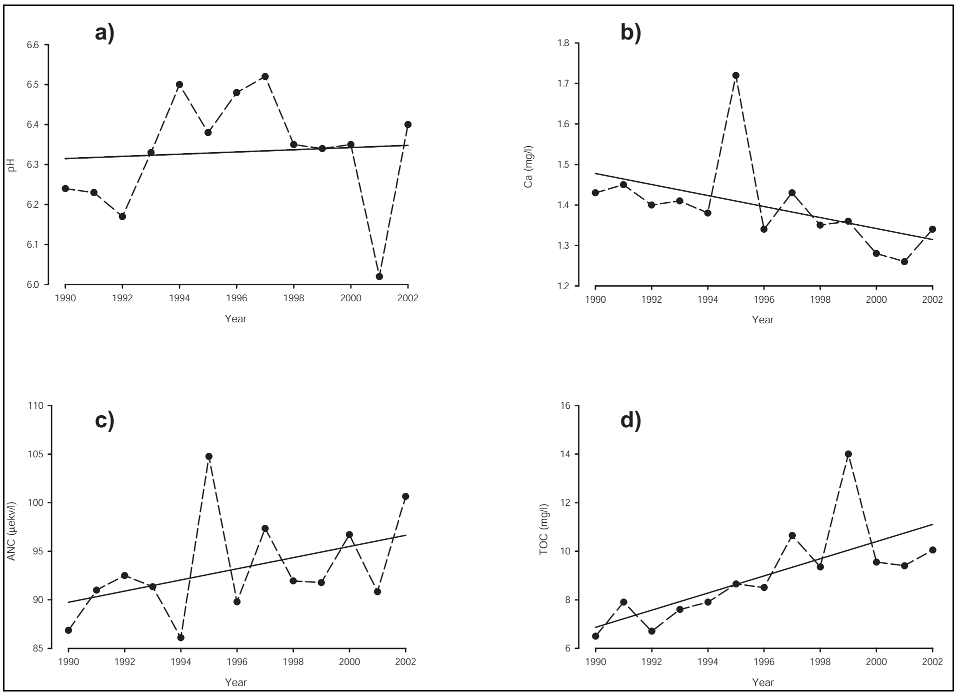


Figure 25. *Line plot and simple regressions of the water chemistry variables from Lake Stensjön used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

The profundal samples of Lake Storasjö also show a signal of biological recovery (Table 8). Linear trends in the water chemistry explain 33.9 % of the abundance variation in the benthic community. A decline in calcium is strongly correlated with linear time, and also an increase in ANC is significantly correlated. The change in water chemistry variables is shown in Figure 26. The increase in ANC and decrease in Ca may indicate that chemical recovery has a role in ordering the benthic community, although pH does not show any trend.

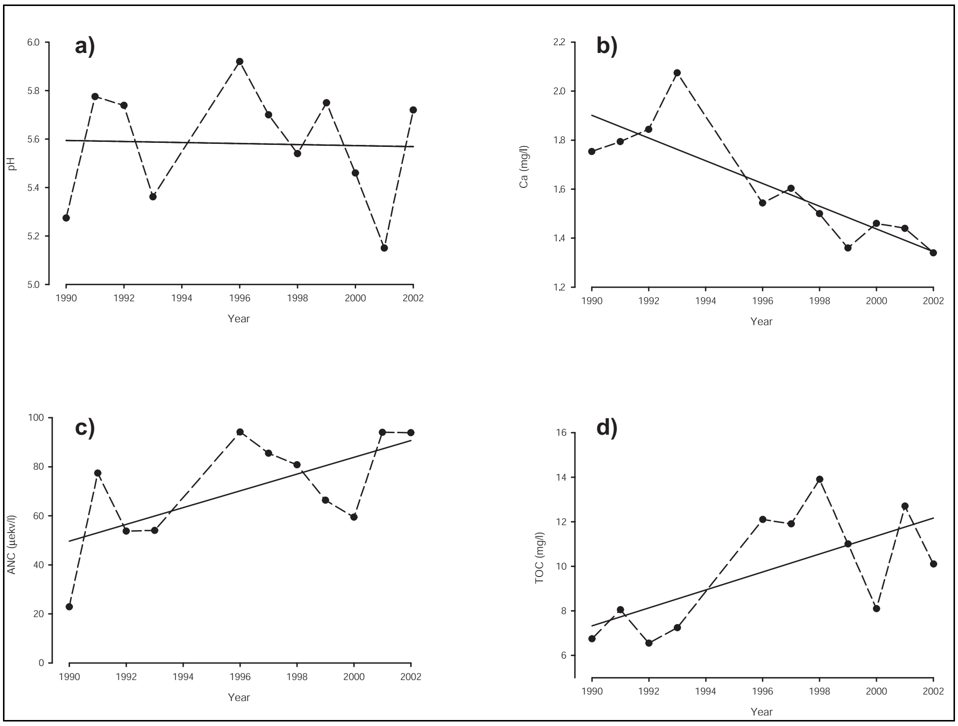


Figure 26. *Line plot and simple regressions of the water chemistry variables from Lake Storasjö used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

The lack of signals of recovery in the lakes with a "real" profundal zone may be a caused by several factors. One important is the confounding effect the oxygen content may have on the profundal community. Oxygen data from the lakes examined here can be found on the internet on the webpage of the Department of Environmental Assesment, Agricultural University of Sweden, Uppsala (http://www.ma.slu.se/). These data show that all of the lakes, except Storasjö, experience oxygen depletion during the summer season in the deep waters, at least in some of the years analysed here. The data are not complete for all of the lakes, but oxygen depletion during one summer season may strongly alter the profundal community, and thus confound any trend of biological recovery. These results are also in line with results in Johnson (1998). In a comparison of several indicator metrics from a set of lakes in Sweden, he found that samples from the profundal showed larger variability, both among samples and among years, than samples from the sublittoral. The conclusion was that samples from the sublittoral would be better than profundal samples when monitoring acidification effects. Confounding effects of the oxygen content in the profundal most likely contributed strongly to these results.

Lake Storasjö is the only lake which shows a linear trend in the profundal community. This lake is shallow with a maximum depth of about 6 meter. This is so shallow that, depending on basin morphology and exposure, a breakdown of the thermal stratification during the summer season is likely and consequently the oxygen saturation in the bottom water may not drop below a critical level for longer periods. The oxygen data for this lake show oxygen depletion for a few years during the examined time period, in addition to winter depletion in some years. However, these incidents has apparently not been so severe that the benthic community has been affected.

Lake Fiolen exhibits no linear trends in the benthic community, neither in the sublittoral nor in the profundal zone for none of the two periods analyzed. The report of a biological recovery in the sublittoral community for the period from 1989 - 1998 in Halvorsen *et al.* (2002) was erroneous, and caused by a lapse when interpreting the results from the analyses. The development in the water chemistry variables used in this study is shown in Figure 27. pH and ANC increases, calcium decreases, while TOC shows no trend. The RDA with only time included as environmental variable gives a significant result, i.e. there is a linear trend both in the sublittoral and profundal communities. However, running the ordination analyses with the water chemistry variables gave no significant result.

In Lake Tväringen only profundal samples exist for the periods examined. There is no signal of recovery in the profundal community in this lake. The water chemistry development in the lake is shown in Figure 28. No evident trend is present in the water chemistry variables, except for a decrease in the calcium concentration. The oxygen data from the profundal also indicates that oxygen depletion may be a problem in some years.

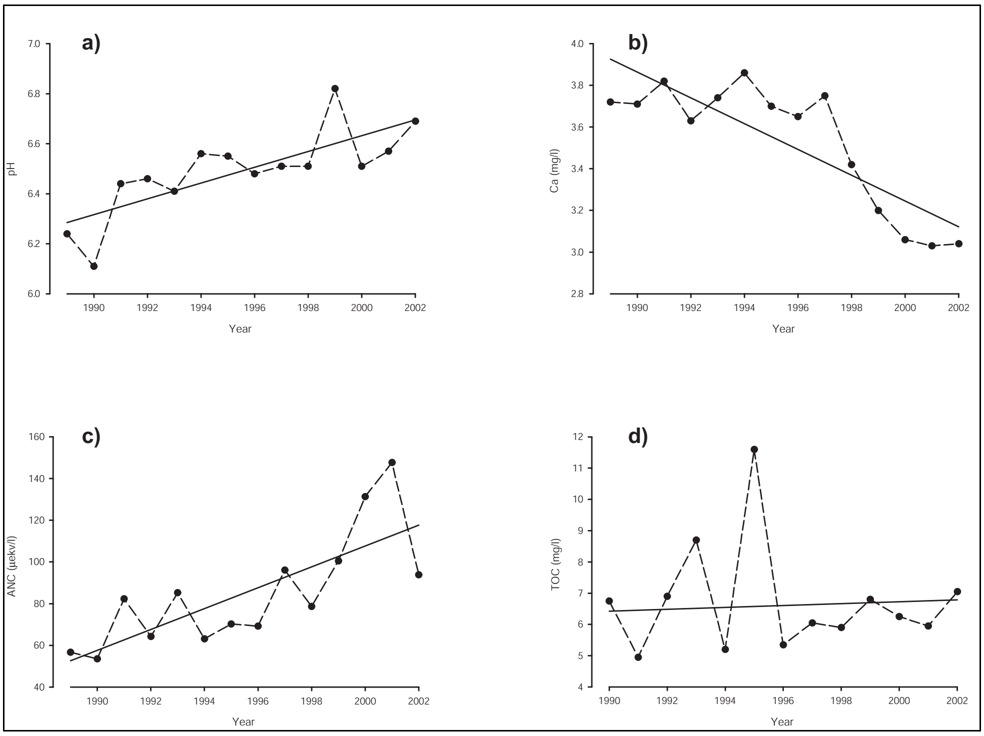


Figure 27. *Line plot and simple regressions of the water chemistry variables from Lake Fiolen used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

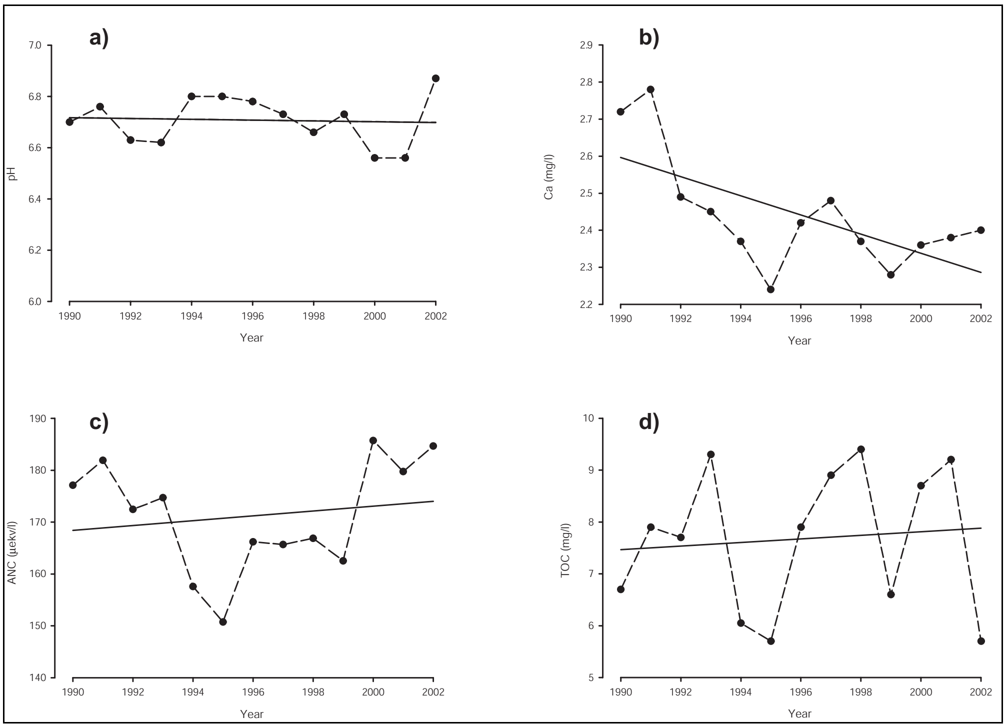


Figure 28. *Line plot and simple regressions of the water chemistry variables from Lake Tväringen used in the analyses.* ***a)*** *= pH,* ***b)*** *= calcium (Ca),* ***c)*** *= acid neutralizing capacity (ANC),* ***d)*** *= total organic carbon (TOC).*

The results from the Swedish lakes show that the two lakes Fräcksjön and Härsvatten, situated close to the western coast in southern Sweden, show a trend of biological recovery in the sublittoral zone during the period from 1990 to 2002. This trend can be interpreted as a result of recovery in the water chemistry. This trend was also apparent in Lake Härsvatten, the most acid of the two lakes, in the shorter period from 1990 to 1998. The lakes Brunnsjön and Stensjön also show significant linear trends in the sublittoral community. However, these changes are linked to a decrease in the calcium concentrations in our analyses, and cannot solely be explained as responses to reductions in acidity. A lagtime between earlier water chemistry recovery and recovery of biota may also be an explanation. Processes connected to climate change may cause these changes, although increases in ANC in both lakes indicate that chemical recovery also plays a part in the biological changes in these lakes. The profundal / sublittoral community of Lake Storasjö also show significant linear trends. Again a decrease in the calcium concentration appears as the strongest explanator, but also an increase in ANC plays a role in the analyses of this lake. For all of the five lakes the TOC also increases. This may be caused by a wetter climate.

The profundal community of the lakes does not show any signs of biological recovery. The confounding effect of oxygen depletion in the deep waters of the lakes is probably one important factor explaining this lack of trend.

8. Conclusions

This report presents an update of results for trends in biology (invertebrate fauna) since 2000 in lakes and rivers connected to the ICP Waters programme. The results in this report confirm the results from the report in 2000 and shows that continued improvement in the chemical status of acid-sensitive lakes and streams leads to biological recovery. This has been evaluated by use of different methods like acidification index, numerical development of sensitive species and multivariate statistical analyses. The trends in biological recovery vary from region to region in accordance with changes in water chemistry.

Responses in different regions:

- In Canada recovery of damaged zooplankton communities has taken place in lakes that chemically have recovered from pH < 6.0 to pH > 6.0. Some recovery of zooplankton species has also occurred in lakes that have not reached the pH > 6. Recovery of benthic invertebrates is also observed 4 – 8 years after the water chemistry has improved to reach the critical limit of the species.
- In Central Europe beginning of zooplankton recovery is recorded in lakes in the Bohemian Forest, Czech Republic. There are also some signs of recovery of sensitive species in several of the German sites, mostly among those situated in the eastern part of the country. However, clear stabile significant recovery is difficult to point out. The general lack of significant trends in the German biological data corresponds with high variation in the water chemistry as well as a general lack of general chemical improvements in the German sites. However, the development in several German sites can quite soon show significant improvements.
- In Scandinavia, especially Norway, all methods used for evaluating biological recovery shows improvements over the last 10 – 14 years. The recovery is most pronounced the latest years where number of localities with significant recovery has increased considerably. This is in accordance with the development in water chemistry. In some Swedish lakes also significant improvements in benthic communities are recorded over time, corresponding with improvements in water chemistry during the same period.

Historical data, core analyses, from the Alps and the Pyrénées indicate generally stabile pH up-core. The changes among invertebrates seam to be attributed to changes in weathering and climate rather than acidification.

From Finland, Ireland, Polen and UK the programme centre have no new data after 2000 that can be used for updating the situation reporter in 2000. However, other investigations in Finland and UK show that sensitive fauna has recovered in formally acid lakes, due to increased pH and decreased labile aluminium. The data from Latvia demonstrate no acidification problem.

9. References

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Appendix A. Reports and publications from ICP Waters

All reports from the ICP Waters programme from 1997 up to present are listed below. All reports are available from the Programme Centre.

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