

CONVENTION ON LONG-RANGE TRANSBOUNDARY AIR POLLUTION

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

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The six year report: Acidification of surface water in Europe and North America Dose/response relationships and long-term trends

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Abstract: The International Cooperative Programme on Assessment and Monitoring of Acidification of Rivers and Lakes is designed to: (1) establish degree and extent of acidification of surface waters, (2) evaluate dose/response relationships and (3) define long-term trends and variations in aquatic chemistry and biota attributable to atmospheric pollution. Data from 200 sites in 14 countries of Europe and North America are available in the database at NIVA. Dose/response relationships show that the fauna are adapted to different water qualities in different regions, and that critical limits for the fauna must be calculated according to data for the specific region. Long-term trends of water chemistry show decreases in SO_4^{2-} and Ca^{2+} at many sites. Nitrate shows no consistent trends.
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Main conclusions

The International Cooperative Programme (ICP) on Assessment and Monitoring of Acidification in Rivers and Lakes is designed to:

- Establish degree and extent of acidification of surface waters
- Collect information to evaluate dose/response relationships
- Define long-term trends and variations in aquatic chemistry and biota attributable to atmospheric pollution, particularly acidic deposition.

Data from 200 sites in 14 countries in Europe and North America are available in the database.

Approximately half of these have both biological and chemical data, the other have only chemical data.

Sites included in the Programme are monitored by each country individually. The monitoring methods used have been harmonised, and intercalibration studies are completed each year. The Programme Centre and the Programme database are located at the Norwegian Institute for Water Research (NIVA).

Dose/response

- There is a strong relationship between water quality and fauna. The major biological division is between taxa associated with high ANC and conductivity (the sensitive species), and taxa associated with high Al, low pH and ANC (the tolerant species).
- The effect of acidification on invertebrates varies from region-to-region depending on the content of Ca^{2+} , humic substances and ionic strength. In areas with strongly oligotrophic water, the fauna is adapted to freshwater with low conductivity, $\text{pH} > 5.5 - 6.5$ and $\text{ANC} > 10 - 30 \mu\text{eq/L}$ to survive. In Central Europe, the fauna is found to be much more sensitive and is adapted to waters with high ionic strength, rich in Ca^{2+} , $\text{pH} 6.0 - 8.0$ and $\text{ANC} 50 - 200 \mu\text{eq/L}$. The critical load of $\text{ANC} 20 \mu\text{eq/L}$ developed in Norway and Scandinavian, should thus be increased to $\text{ANC} 50 \mu\text{eq/L}$ for Central Europe based on aquatic fauna.
- Preliminary work on predicting pH from invertebrate assemblages is promising.

Long-term trends

- Analysis of long-term trends in the ICP data aims to discover regional changes in surface water chemistry, due to changes in deposition.
- A decrease in SO_4^{2-} is a major feature for the sites. The decrease in SO_4^{2-} does not necessarily result in an improvement of water quality in terms of acidification.
- Nitrate showed a range of trends from significantly increasing to significantly decreasing. Several of the sites are characterised by NO_3^- breakthrough. Nitrate is a strong acid, and its presence in higher concentrations may indicate significant acid loading.
- A decrease in Ca^{2+} occurred at many European sites.
- The combined decrease in Ca^{2+} and increase in NO_3^- may account for the lack of recovery from acidification at sites with decreasing SO_4^{2-} .

Intercalibration

- Chemical intercalibration show that the results are compatible between laboratories, and results are improved over the years.
- Biological intercalibration conducted in 1992 show that the methods are reliable and suitable for evaluation of acidification. The tests included, however, only a selection of the species present at the various sites.

Further Work - The ICP on Waters Programme in the next three years

- Priorities according to programme aims:

Continue the work on the understanding of the acidification of aquatic ecosystems by linking changes in water chemistry with changes in biology and continue the work to evaluate long-term trends in water chemistry due to acid inputs.

- Continue data quality control including exercises in chemical and biological intercalibrations.
- Assessment of the importance of acidification by NO_3^- loading.
- The coming three years will further emphasise the importance of international Cooperation on methods and development of the data base. The ICP-waters will take initiatives to have a general discussion of methods on long-term trend analysis and establish close relationships to the other ICP's.

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1. Introduction

1.1 Programme background

The International Cooperative Programme on Assessment and Monitoring of Acidification of Rivers and Lakes was established under the Executive Body for the Convention of Long-Range Transboundary Air Pollution at its third session in Helsinki July 1985 (ECE/EB AIR/7, Annex/V).

At this meeting Canada was appointed as lead country for the first phase. The delegation from the Union of the Soviet Socialist Republics offered to support activities in the leading Programme. The Executive Body also accepted the offer from Norway to provide facilities for the Programme Centre. After the first phase, Norway also was appointed to be lead country for the Programme.

It was agreed that the monitoring programme would be designed to assess, on a regional basis, the degree and geographical extent of acidification of surface waters. The data collected should provide information on dose/response relationships under different conditions and correlate changes in acidic deposition over periods of years with the physical, chemical and biological status of lakes and streams (EB.AIR/W6.1./R21). The objectives has been reformulated later on keeping the intentions still.

The Programme Task Force has the mandate to plan and supervise activities undertaken within this Programme. At its first meeting, which was held in Grafenau (Germany) on 27 April 1986 the Programme Task Force discussed the scope, objectives and general organisation of the programme and formulated a monitoring manual to be recommended to the Working Group on Effects.

As reported to the fourth session of the Executive Body, the International Cooperative Programme are based on existing programmes in participating countries, implemented by voluntary contributions in the starting phase. Following the starting phase in 1986, the operational part of the programme using harmonised method started 1987 and has proceeded for a six year period.

The Programme Task Force, at its second meeting in Oslo, 14 October 1986 adopted the Programme Manual compiled by the Programme Centre and based on a draft outline prepared by Canada. Canada, after finalising the Programme manual, entrusted Norway to take the role as lead country in the subsequent implementation phase of the programme. This was confirmed by the Working Groups on Effects at its sixth session and reported to the Executive Body on its fifth session (ECE/EB.AIR/16 - Annex II) in December 1987.

1.2 Programme Aims and Programme Objectives

The mandate, the intentions of the programme, the goals and objectives are given in the manual.

Programme Aims

- Establish degree and geographic extent of acidification of surface waters
- Collect information to evaluate dose/response relationships
- Define long-term trends and variations in aquatic chemistry and biota attributable to atmospheric pollution, particularly acidic deposition.

Programme objectives

- Establish an international network of surface water monitoring sites
- Promote international harmonisation of monitoring practices by:
 - Specifying a manual on methods and operations
 - Conducting inter-laboratory quality assurance tests
 - Compiling a centralised database with data quality control and assessment capabilities
- Develop and/or recommend chemical and biological methods for monitoring purposes
- Report on:
 - Geographic extent of acidification using terrain sensitivity and current information sources
 - Dose/response relationships by:
 - a) determining changes in systems with similar buffering capacity and hydrology but different acid inputs
 - b) comparing chemical and biological responses of systems with different buffering capacities but similar deposition levels
 - Long-term trends by using sites with data over periods of years
- Conduct workshops on topics of central interest to the Programme Task Force and the aquatic effects research community

1.3 Programme Status

The countries of Europe and North America have representatives in the Programme (Table 1.1). Data from 200 sites in 14 countries are available in the database (Figure 1.1 and 1.2); approximately half of these have both biological and chemical data, the remaining have only chemical data (Appendix A).

Table 2.1 Programme status as of 1.1.1994

Country	Participating on Task Force Meetings	Data Reported	Participating Intercalibration
Austria		*	*
Belgium	*	*	*
Bulgaria	*		*
Canada	*	*	*
Czech Republic	*	*	*
CIS	*	*	*
Denmark	*	*	*
Ireland	*	*	*
Finland	*	*	*
France	*		*
Germany	*	*	*
Hungary	*	*	*
Italy			*
The Netherlands	*	*	*
Norway	*	*	*
Romania	*		*
Spain			*
Sweden	*	*	*
Switzerland	*		
United Kingdom	*	*	*
USA	*	*	*
Sum	18	15	20

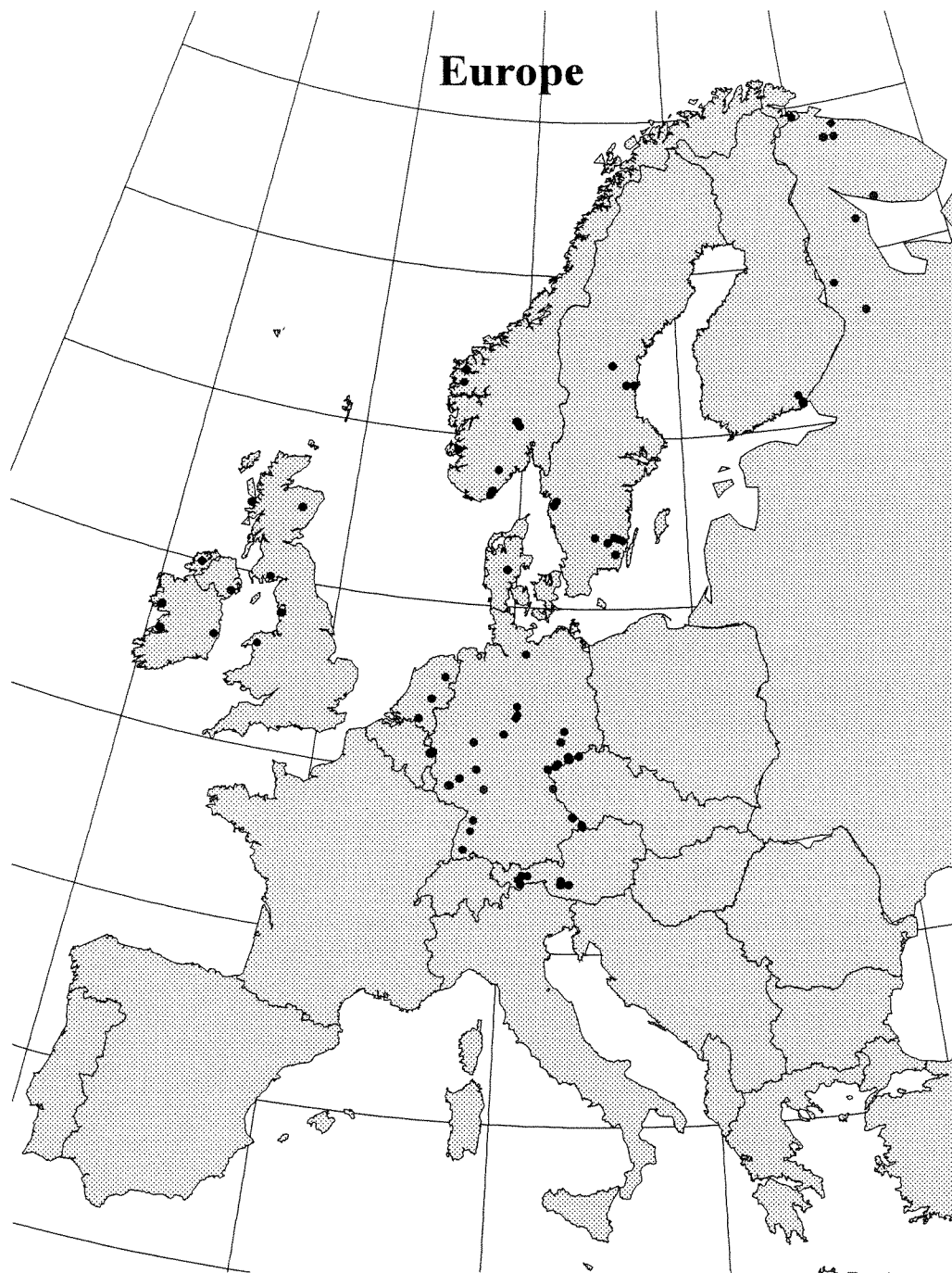


Figure 1.1. Map showing location of monitoring sites in Europe.

North America

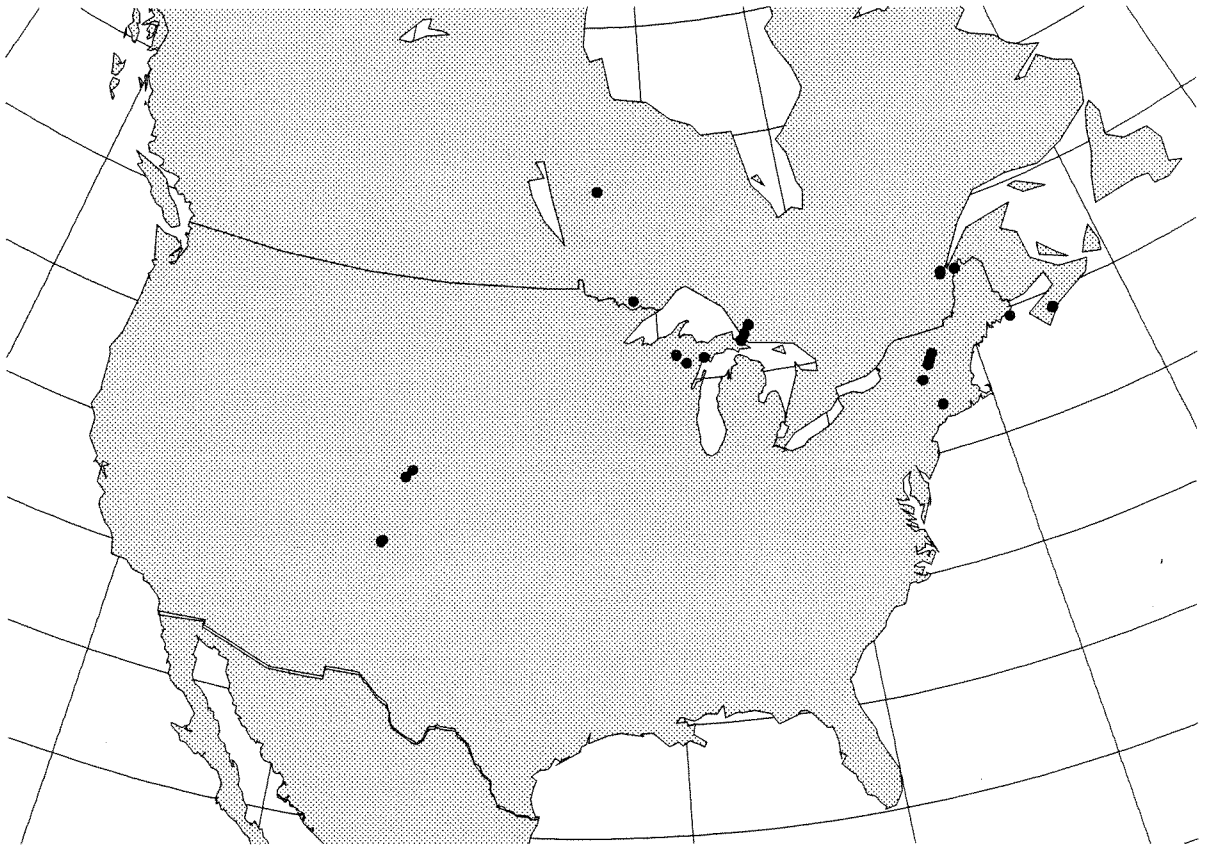


Figure 1.2. Map showing locations of monitoring sites in North America.

1.4 Relationships between chemistry and biology - dose/response

Watersheds that are undisturbed and unpolluted by man, have a water quality which has allowed aquatic life to evolve over thousands of years. In these watersheds the aquatic systems have equilibrated to a trophic status determined by the catchment, and they are usually characterised by a high level of species diversity. These ecosystems would be considered ecologically healthy. Watersheds impacted by man generally have a reduced level of species diversity. They are typically unbalanced and often dominated by a few species (May, 1976, Reynolds et al. 1993).

Presently, the number of systems where water quality has been damaged by pollution from man is growing. It is therefore a great need to protect and restore all types of freshwater ecosystems. Acidic deposition has damaged fish populations in large areas of Scandinavia and in other parts of the world. However, damage to other aspects of aquatic ecosystems is evident over a much broader area and can be measured through surveys of water chemistry and biota. Where undisturbed ecosystems still exist they can serve as references.

The effects of acidic episodes are most evident on invertebrates in rivers and streams. Such localities are therefore excellent for both short- and long- term monitoring. In lakes the densities of sensitive species are usually lower than in running waters. Further the effect of acid episodes are weaker in lakes, due to the fact that acidic stream inflows are mixed with less acid lake water. Thus short-term effects on invertebrates are seldom seen in lakes. Lakes are, however, well suited for long-term monitoring of chronic acidification.

In seepage lakes acid episodes caused by precipitation are unlikely to occur. Acid episodes can, occur in seepage lakes, but these episodes will, as a rule, be caused by processes within the lake itself or in the surroundings. Acid deposition will, however, erode its alkalinity reserves in this type of lakes, due to acidification of the soils and the ground water. Monitoring of these systems can therefore only be used for long-term trend studies.

By combining biology and water chemistry it is possible to describe simultaneously both the ecological system and the water quality in different catchments. There exist models from which water quality in different types of waterbodies can be predicted, based on the amount of acidic deposition (Henriksen et al. 1988). When the current knowledge regarding the relationship between deposition and water chemistry, and between biology and water quality, are combined, it is possible to predict the ecological status of an ecosystem versus the load of acidic components. These models can be used to predict a critical load, or the highest load that will not lead to a specified ecological damage.

The definition of critical load for atmospheric inputs of an acidifying compound is the highest load that will not cause chemical changes leading to long-term damage to ecosystems structure and function (Sverdrup et al. 1990). Whether or not a site receives deposition in excess of the critical load, can be tested over the long-term by the use of a biological index, the acidity score system or Raddum index. The system is based on the tolerance of species to acid water expressed by an index. Species with the lowest tolerance to acidification (i.e. the first to go extinct when pH starts to fall) get the index 1. The most tolerant species, however, survive in strongly acidified localities, and are given index value 0. The acidity index for a river or a lake will be a number between zero and one, which represents the average index of all locations sampled. For more information see Raddum et al. (1988) and Fjellheim and Raddum (1990). The definition of critical load suggests that an index close to one for freshwater ecosystems, except in naturally acidic systems.

The method of biological acidification index first developed in Norway for acid sensitive waters, has been further adapted to cover sites of different climatic zones. Substantial effort has been expended by the USSR in providing possibilities for the testing of different biological methods. The USSR hosted a workshop on biological methods, in conjunction with the third Programme Task Force meeting in Togliatti 14 October 1987, and invited members to a workshop on the comparison of biological methods in the Caucasus in May 1990.

1.5 Long-term trends in water chemistry

Long-term trend analysis on water chemistry is a method that is mainly used to reveal trends in a single catchment. Long-term trend analysis on the ICP dataset is the first attempt, as far as we know, to try to discover regional patterns both in Europe and North America in long-term changes in surface water chemistry, due to changes in deposition.

The monitoring sites in the ICP programme include a wide range of watersheds; small streams, big rivers, seepage lakes and drainage lakes of variable size. In addition there is also a wide range in physical characteristics such as climate, topography, hydrology, soil type and depth and amount of acid loading. Although the sites are different in many ways, it is still possible that the same mechanisms can influence on the chemistry in all of the sites. On the other hand, even if some sites show the same trends, it does not necessarily mean that they are controlled by the same mechanisms.

Trends in surface water chemistry that result from atmospheric acidification are expected to occur at low magnitudes. Climatic cycles such as droughts and wet periods often have a large, though short-term effect on surface water chemistry. Thus the short periods of record, often as short as 5 years for many of these sites, is too short, ecologically speaking, to document long-term changes from relatively small atmospheric inputs. Therefore, the trends reported here should be considered as preliminary results. However, these results are still important, as they outline the limitations of our understanding of these systems.

2. Sampling methods and data quality control

2.1 Presentation of the database

Sites included in the Programme are monitored by each country individually, although the monitoring methods used have been harmonised across participants, and intercalibration studies are completed each year. The Programme Centre and the Programme database are located at the Norwegian Institute for Water Research (NIVA). The data base is hierarchically built, containing water chemistry and biological data. For some stations deposition and hydrological data also exist. For every country there is a separate dataset, and new datasets are easily established. Each dataset has its own key parameters. In Appendix A, the data set names are listed under "file", the special codes for the stations are listed under "code", and the data periods are listed under "year". The system is made for the UNIX operating system. The data base also contains site specific information, where characteristic data for the rivers, lakes and catchments are noted. These data were presented in the Data Report 1988.

The biological data consist mainly of benthic invertebrates both from running water and lakes. Zooplankton data are present from lakes in Germany. Information on phytoplankton is delivered from Sweden, while diatoms are available both from Germany and Netherlands. Monitoring of phytoplankton is new and at this time the data only cover a short time period and a few localities. It is too early, therefore, to analyse the value of these data for the programme. The diatoms are well known as acidification indicators (Charles et al. 1989). Information on diatoms from localities in the Netherlands, Germany and United Kingdom are published by van Dam (1988), Steinberg and Putz (1989) and Battarbee et al.(1988). These data are available for the Programme for later analysis and represents a starting point for a long trend monitoring.

2.2 Water chemistry monitoring

Investigation of water chemistry is carried out by each member country, and both lakes and rivers are sampled and studied. The sampling objective is to obtain a measurement that is representative of mean conditions of the lake or river for the period of sampling. The analytical programme includes pH, alkalinity, SO_4^{2-} , NO_3^- , DOC, and all other major ions. Ammonium, Al, total N, total P, dissolved O, silica and metals may be included. Standard procedures for low ionic strength waters are used.

Quality assurance and quality control are the responsibility of the National Focal Points. The Programme Centre will, however, to ensure data quality and correct technical transfer of data, do data quality control according to the following:

1. looking for outliers
2. looking for continuity in time series
3. ionic balance

Data is not excluded without approval by the originator. The objective is to create a good quality database for efficient use in the near future as well as in the long run.

In particular the ionic balance is done by a data programme made in two versions, the first including all major ions, the second also including Al, NH_4^+ and TOC. The first set of equations are the following:

Sum anions	: SAN =	$\text{ECI}^- + \text{ENO}_3^- + \text{ESO}_4^{2-} + \text{E-ALK}$
Sum cations	: SKAT =	$\text{ECa}^{2+} + \text{EMg}^{2+} + \text{ENa}^+ + \text{EK}^+ + \text{EH}^+$
Difference cations/anions	: DIFF =	$\text{SKAT} - \text{SAN}$
Difference in percent	: D-PRO =	$\text{DIFF in \% of SKAT (DIFF*100/SKAT)}$

"E" in front of the chemical component means that the component is given on an equivalent basis. For samples where analysis of Al, NH_4^+ and TOC are present, the second set of equations is used:

Sum of anions	: SAN2 =	$\text{SAN} + \text{AN}$
Sum of cations	: SKAT2 =	$\text{SKAT} + \text{ELAL} + \text{ENH}_4$
Difference cations/anions	: DIFF2 =	$\text{SKAT2} - \text{SAN2}$
Difference in percent	: D-PRO2 =	$\text{DIFF2 in \% of SKAT2 (DIFF2*100/SKAT2)}$

AN⁻ is calculated from the TOC value taking into account weak organic acids. The equation used is:

$$AN^- = 4.7 - 6.87 * \exp(-0.322 * TOC)$$

The equation is based on empirical data from Norwegian sites. Tests have shown good agreement with analytical results. Other equations (Oliver et al. 1983) taking into consideration the pH-dependent dissociation of organic matter might also be used with minor differences.

In order to check the ion balance, all of the necessary variables for calculating the sums of cations and anions must be analysed. For good analytical results, the difference in % between sum cations and anions should be < 10%. A further check of the ion balance is made by comparing the measured conductivity to the conductivity calculated from the measured ions. Also a check of non marine Na (Na*) may indicate possible problems in Cl⁻ analysis.

ANC (acid neutralising capacity) used in this report is defined as follows:

$$ANC = ECa^{2+} + EMg^{2+} + ENa^+ + EK^+ - ESO_4^{2-} - ECl^- - ENO_3^-$$

2.3 Biological monitoring

2.3.1 Sampling methods for invertebrates

Invertebrates are small animals living in/on the bottom of lakes and streams (benthic fauna) or in the open water (zooplankton). They have a wide distribution and usually occur in high numbers, both with respect to species richness and density. Invertebrates can occupy almost every type of habitat in fresh water. The different species, however, have specific tolerance limits related to water chemistry, physical parameters, nutrient levels, etc. For this reason, analysis of invertebrate communities has become a well established tool for many types of environmental investigations (Johnson et al. 1993).

Invertebrates are sampled from lakes and rivers by each country. Sampling sites are chosen to represent average conditions in a specific region/geographical area and the important ecotypes of that region. For example, running water sites dominate in the mountain areas from Norway and Germany, while in Sweden and Finland lakes are the dominant choices for monitoring.

For the biological work the most important task is to obtain information about the species composition and relative abundances in each locality. The main habitats are sampled either by quantitative or qualitative methods. Quantitative sampling in lakes is done with an Ekman dredge or a core sampler. In running water a Surber sampler is used. Qualitative sampling is carried out by the kick technique. The methods are all well known standard methods. More information about sampling strategy is found in the Manual for Chemical and Biological Monitoring (1987).

2.3.2 Sampling methods for diatoms

Diatoms are microscopic, unicellular algae. They are abundant in a wide range of freshwater, brackish, and marine environments and may be identified with a high degree of taxonomic precision. Their use in paleolimnological studies of lake acidification is well established, as is their role in the biological monitoring of organic pollution.

In most lakes and streams, diatoms occupy a variety of habitats, and may be found on the surfaces of stones or macrophytes, attached to sand grains or living on the sediment surface as well as in the open water column. Within a single lake or stream, species distribution and abundance of diatoms is strongly controlled by habitat and water quality. Within a single habitat species composition may vary, for example, between different macrophyte hosts, or along a sediment particle size gradient. At the sites in the ICP on

waters these sources of variation are as far as possible held constant. To minimise differences caused by habitat variation (i.e. macrophyte hosts, sediment particle size etc.) physical variables are held as constant as possible at each sampling site.

At each stream site, ten cobble size stones are selected from pools in three discrete locations over a 50 m reach. Stones are collected from a depth below that of minimum flow. Diatoms are then removed from the stones by brushing them. Samples are placed in vials and preserved in Lugols Iodine preservation. At lake sites, ten cobble size stones are selected from a depth of 20 - 30 cm from 2-4 surveyed locations around the shore.

Sediment samples are used to provide diatom samples integrated both spatially, (i.e. across all diatom source communities), and temporally (over a period of months for sediment traps, or years for surface sediments).

A standard sample preparation technique is described by Batterbee (1986) or Stevenson et al. (1987). Three hundred to five hundred diatom valves are counted from each sample and identified to species level where possible. Results are compared by using accepted protocols for taxonomy and nomenclature (Kingston 1986; Stevenson 1991). Diatom data for each site are presented to the data centre in three forms:

1. A species list, giving relative abundance (percentage) of all taxa greater than 1.0% of the total count.
2. Summarised by pH preference groups; taxa are divided into a number of pH groups, and the percentage of the assemblage represented by each group is listed. The most frequently used classification scheme is that of Hustedt (1937-39). This system divides diatoms into 5 classes:

- (1) Alkalibiontic: occurring at pH values > 7
- (2) Alkaliphilous: occurring at pH about 7 with widest distribution at pH > 7
- (3) Indifferent: equal occurrence on both sides of pH 7
- (4) Acidophilous: occurring at pH about 7 with widest distribution at pH < 7
- (5) Acidobiontic: occurring at pH values < 7 with optimum distribution at pH 5.5 and under.

The diatom can be classified to their pH optimum. Four classes are identified:

- (1) pH optimum less than 6
- (2) pH optimum between 5 and 6

- (3) pH optimum greater than 6
- (4) Unknown

3. Diatom inferred pH is calculated by use of the index B method of Renberg and Hellberg (1982) or by weighted averaging (ter Braak and van Dam 1989; Birks et al. 1990).

2.3.3 Sampling methods for fish

Monitoring the status of fish populations became part of the Programme in 1992. Most fish species can live for many years and they usually have life stages with differing sensitivity to acidic water. For example salmonids are most sensitive just after hatching, just prior to the smolt stage and after spawning. Since changes in these stages cannot always be detected in the same year as they occur, the monitoring programme on fish emphasises use of methods that can detect changes at the earliest possible time (Amendment to the Manual, 1992)

Since 1976, Norwegian studies have used a standard investigation programme to describe the fish population status/changes in acidified lakes. Two methods have been used; interview-method where information is given by local lake-owners etc. having first hand information of fish population status and test fishing-method, using standard gill-net series and sampling procedures to describe fish population size, age composition, growth pattern, food organisms etc. Different countries use different traps for catching fish.

Monitoring fish status is new in the programme and only Norway and Canada have delivered data to the programme. We have therefore no data to report from the fish programme at this time.

2.4 Data quality control

Data Quality Control is an important part of the programme. Annual laboratory intercalibration exercises have been established, involving 30 laboratories from 18 countries in the 7th intercalibration. Hence this has become a routine control of normal laboratory practice in the participating laboratories. The Youden method (two sample approach) is used to assess laboratory performance. Annual reports show how target accuracies are met for most ions in natural waters.

Quality control of biological parameters commenced in 1992. Participating laboratories receive unknown invertebrate samples of their own fauna. Parallel diatom samples from lakes in different countries have been distributed in the same way and examined by diatom experts. The identification of species is generally accurate. In the presence/absence evaluation system, the misdetermination and lack of determination of a small portion of the test samples had no influence on the evaluation of the acidity level. However, errors and lack of species identifications are of high importance when using models based on the relative occurrence of species. Future quality control evaluations will take this into consideration.

3. Methods

3.1 Dose/response

The effect of acidic episodes is best seen among invertebrates in rivers and streams. The fauna of rivers and streams provide both short and long-term warning and indications of acidification. In lakes the strength of acid episodes are weaker and animals can avoid acid surface water by seeking refuges in deeper parts. The biological changes in lakes are therefore more indicative of chronic conditions which persist over the long-term.

The importance of keeping sampling sites, techniques and timing constant has been emphasised in the Manual for Chemical and Biological Monitoring (1987). Due to the life-cycle pattern of invertebrates, it is recommended that sampling should be done during spring and fall at a minimum. Presence or absence of indicator species will be reflected by the acidity index which is calculated for each site. Changes in acidification index through this is indicative of faunistic response to differences in water quality. The most acid sensitive species disappear very early in the acidifying process (indicator species), while others are more tolerant. There is a gradient in the tolerance of invertebrate species/ groups in waters of varying acidity. If the sampling season changes from year to year, the value of the data will decrease due to lack of homogeneity. This is also the case if the sampling site or the number of sites change over the years. In such cases the quality of the data will decrease and interpretation will be more difficult.

The sorting and identification of species is another key process in monitoring work. Effort must be allocated to search for sensitive species in qualitative samples, if these are subsampled. By quantitative sampling, all individuals are by definition analysed, so species will not be overlooked. Identification of species, especially the sensitive ones, is important for stating the presence/absence situation. The taxonomist must be familiar with these groups. The tolerant groups are also of importance, especially for analysing the community structure which in many cases is of great support when stating the level of acidity. The occurrence of rare species is often of high interest for taxonomists, but these species have in most cases reduced importance for the assessing of acidity.

Information on community structure is also important for further development of the index in order to assess acidification. Inclusion of multivariate statistical analyses, will sort the information in a more precise way, and predictions about effects of acidification in the different types of ecosystems will be much more accurate.

A long species list at a station normally indicates good quality data. If the list only contains a very few names, it can indicate either an exposed situation at the locality or poor quality of data. In these cases samples should be given close attention so no species are overlooked.

Many of the most important species used in the index system are widespread, quite common and well known among zoologists. The chance of misidentifications of important species are in fact very low. Further, the effect on the overall index determination of a single misidentification is usually minor. This is because presence of one sensitive species has the same meaning in the index system as record of many other sensitive species. Through this, possible fault made during identification of sensitive species, will have no influence on the estimation of the acidity index as long as one of the species is correctly determined. The risk of stating a wrong index is therefore low in most cases. The difficult cases are when a site has an acidity near the shift between two acidity levels. A wrong index determination of a site will, however, have reduced influence on the mean index of a watershed, if the number of sites is as recommended (>15 sites).

Aquatic invertebrates and chemistry information from Western Norway in 1988, 1989 and 1990 and data from other parts of the country have been analysed statistically (Larsen et al. in prep.). Some results from this study are presented in this report. In total the material consisted of 180 samples with 127 different invertebrate taxa and associated water chemistry (pH, Ca²⁺, Al_{tot}, ANC and conductivity). The data were log₁₀ transformed and analysed with the use of a detrended correspondence analysis (DCA) (Hill and Gauch, 1980) and by a canonical correspondence analysis (CCA). The analysis consisted of numerical ordination, based on the individual species, related directly to the measured environmental parameters.

3.1.1 Evaluation methods for invertebrate fauna - the Raddum Index

The number of species in a watershed is determined by several factors. These factors are both direct and indirect in nature. Direct effects relates to the physical and chemical status of the water in the locality. Different chemical compounds have to be within a certain range of concentrations in order to meet the demands of each species. If concentrations exceed the tolerance, the animal will die. Tolerance limits exist also for physical factors, such as temperature, water current, bottom substrate, etc. Biotic effects or indirect effects such as predations and competition for food, can both influence on the presence and absence of

species. For this reason monitoring is done in systems where acidification is expected to be the most relevant species-determining variable.

Based on field observations and experiments in water with low ion- and humic content, the tolerance of different species to acidity is determined and expressed by a number, an acidity index. The species can obtain the index 0, 0.25, 0.5 or 1. Species with very low tolerance to acid water will go extinct when the pH falls below 5.5. These species are given the index 1. More tolerant species can live in water with acidity down to pH 5.0. Those species are given the index 0.5. Species tolerating pH reductions to 4.7 obtain the index 0.25, while very tolerant species that can live in water with pH < 4.7, are given the index 0. To some extent the sensitivity of individual species is altered by Ca²⁺ concentrations in the water (Lien et al. 1992). Species richness, density biomass are usually positively correlated with increasing Ca²⁺ concentrations.

A computer programme assigning an index to a sample searches for species in a hierarchical order (first species with index 1, then species with index 0.5 etc.) and this is used to index each site. The index can be computed for a combinations of sites, watersheds, regions or countries. Comparisons of the mean index between watersheds or seasons should be based on 15 sites, representing both tributaries as well as the main river or lake. Many of the watersheds/areas monitored have a lower number of sites. The strength of the data with a low number of sites is reduced.

The acidification index does not take into account sublethal effects which obviously are very important for evaluations of acidification. However, very little is published about such effects on invertebrates and many different factors can reduce the abundance of sensitive species. Changes in the densities of these species are therefore difficult to relate to acidity without other environmental information. Raddum and Fjellheim (1984) investigated the common sensitive mayflies (M) and tolerant stoneflies (S) occurring in Norwegian running water and found that the ratio M/S was > 1 in water with pH > 6 in localities situated below 700 m above sea level. This information can be used when the sensitive mayflies are present, acidification index 1. In these cases the index 1 can be defined as index 0.5 + the ratio (M/S). In sites with damaged mayfly fauna the ratio will be < 1 and the sum will be a number between 0.5 and 1. If the sum 0.5 + B/S > 1, the index is set to 1. This method will make a smooth "new" acidification index between 0.5 and 1. This new index has been used in studies of river Nausta and has been shown to correlate well with changes in water chemistry (Raddum and Fjellheim 1994). Although this new index has not been tested on other watersheds, initial results seem promising and work will continue on its developments.

The most useful feature of the acidification index technique is not only its ability to pinpoint the acid sensitivity of a specific location, but rather its ability to show how the fauna changes over time in response to long-term trends in acidification. Consistently increasing index over a number of years indicate the recovery of aquatic communities from the effects of prior acidification. This is shown in limed watersheds (Fjellheim and Raddum, 1992). The index can be tested through the changes in relative densities of species over time. Variations in the density of a species are tested statistically and significance levels are indicated (Wilcoxon Matched pairs signed rank test). Occurrence of a species at different sites is tested in the same way. Tested changes among sensitive species can be used as a test of the trend in acidification, expressed by the acidity index.

3.2 Long-term trends in chemistry - Data Analysis Methods

3.2.1 Data Description

The monitored sites in the ICP-programme are located in regions where there is acidic deposition, and geology that renders surface water sensitive to its effects. Data from 67 sites are used in long-term trend analysis on water chemistry including 24 rivers and 43 lakes, in 8 countries (Canada, Denmark, Finland, Germany, The Netherlands, Norway, Sweden, and the U.S.) (Table 3.2.1). These sites were chosen based on the number of years of monitoring and on the number of data points for the site. Selection criteria were a minimum of 5 years of existing data with at least 14 data points in the 5-year period. At the time when the trend analysis was conducted (1991), many sites had only data for 4 years. A list of the sites included for these analyses is attached (Table 3.2.1). Data from the monitoring sites were examined for internal consistency by evaluation of charge balance where complete ion chemistry was available, and identification of outliers in intervariable relationships, such as the relationship between alkalinity and Ca^{2+} . Acceptable analytical uncertainty was defined as difference between sum cations and sum anions $< 10\%$.

Trend tests were applied to the variables of interest at each site, whenever data availability met the above criteria. The variables of interest for trend tests were alkalinity, pH, SO_4^{2-} , NO_3^- , NH_4^+ and Ca. Alkalinity and pH were chosen because they change with additions of acid or base to the system. SO_4^{2-} and NO_3^- are strong acid anions and NH_4^+ are a strong base that derive from atmospheric sources. Calcium is a major base cation derived mainly from cation exchange and weathering reactions in soils.

Surface water is influenced by the chemistry and amount of precipitation. Precipitation and deposition chemistry were seldom available at the same site as the surface water chemistry, so data from the nearest deposition site from existing networks were obtained and paired with the surface water sites. Surface water sites that did not have deposition monitoring sites within 160 km were not paired with a deposition monitoring site. In Canada deposition data were from the CAPMoN (Vet et al. 1989) network, while in Europe most of the data were obtained from EMEP (Schaug et al. 1993). For the U.S., data were obtained from the National Acid Deposition Program, or from Map3S (Simpson and Olsen, 1990) and precipitation volume was estimated from local meteorological sites in the U.S. (Climatedata, 1988). Trend analyses were made for monthly deposition, average concentration, and precipitation volume.

Table 3.2.1 A list of the ICP sites included in the trend analysis. Only sites with at least five years of data and 14 data points per variable were included.

Country	Area	Site Name	Station id.	Long./ Lat.	Lake/ River	Years Monit	Period Monit	
Norway	Buskerud	Langtjern		9°43'0" 60°22'0"			72-91	
	Oppland	Aurdøla	90 1	9°30'0" 60°30'0"	R	5	86-91	
	Telemark	Storgama		8°32'0" 59°10'	R	17	74-91	
	Aust Agder	Birkenes			8°15'0" 58°23'0"	R	19	72-91
			Tovdalselva	7 1	8°8'0" 58°15'0"	R	11	80-91
	Rogaland	Vikedalselva	32 9	5°58'0" 59°32'0"	R	9	82-91	
	Sogn og Fjordane	Gaular			° °	R	11	80-91
Nausta			34 1	5°53'0" 61°34'0"	R	11	80-91	
Trodøla			34 5	5°56'0" 61°57'0"	R	7	84-91	
Sweden		Alsteraaen,Getebro	S02	16°39'47" 57°5'55"	R	5	85-90	
		Alsteraaen,Stroemsborg	S03	15°39'47" 57°5'55"	R	6	85-90	
		Fraecksjoen	S11	12°11'2" 58°8'55"	L	5	84-90	
		Aneraasen,Haersvatn	S12	12°1'56" 58°1'14"	R	6	84-90	
		Delangeraaen, Iggersund	S01	17°5'23" 61°38'19"	R	6	84-90	
Finland		Hirvilampi	SF01	27°55'0" 60°41'0"	L	12	78-90	
		Kivijaervi	SF04	27°40'0" 60°57'0"	L	28	62-90	
		Maekelampi	SF03	27°53'0" 60°44'0"	L	12	78-90	
		Vuorlampi	SF02	27°55'0" 60°44'0"	L	12	78-90	
Denmark		Skaerbaek	SK110B	9°25'22" 56°5'0"	R	14	76-90	
FRG	Harz	Alte Riefenbeck	AltR1	10°23'20" 51°47'13"	R	4	86-90	
		Alte Riefenbeck	AltR2	10°22'36" 51°46'26"	R	4	86-90	
		Alte Riefenbeck	AltR3	10°23'10" 51°45'44"	R	4	86-90	
		Varleybach	Var1	10°20'26" 51°53'16"	R	4	86-90	
		Varleybach	Var2	10°19'46" 51°52'43"	R	4	86-90	
		Grosse Schacht	Sch9	10°22'3" 51°66'27"	R	4	86-90	
	Taunus	Rombach	Rom2	8°26'14" 50°12'0"	R	4	86-90	
	Hunsrueck	Traunbach	Tra1	7°6'59" 49°53'12"	R	8	82-90	
	The Netherlands		Achterste Goorven A	AGA	5°12'36" 51°33'52"	L	15	78-90
		Achterste Goorven B	AGB	5°12'51" 51°33'53"	L	12	78-90	
		Achterste Goorven E	AGE	5°12'51" 51°33'57"	L	11	78-90	
		Gerritsfles	GER	5°49'6" 52°9'39"	L	11	80-91	
		Kliplo	KLI	6°26'19" 52°50'7"	L	10	80-90	

Country	Area	Site Name	Station id.	Long./ Lat.	Lake/ River	Years Monit	Period Monit
Canada	Nova Scotia	Beaverskin Lake	C14	-65°19'58" 44°18'28"	L	10	80-90
		Kejumkujik Lake	C13	-65°13'13" 44°22'9"	L	9	83-91
		Little Red Lake	C12	-65°23'43" 44°20'20"	L	9	80-89
		Mount Tom Lake	C10	-65°18'0" 44°22'5"	L	9	80-89
	Quebec	Lac Bonneville	C07	-71°7'0" 47°19'0"	L	8	82-90
		Lac Josselin	C06	-71°40'0" 47°22'0"	L	8	82-90
		Lac MacLeod	C09	-70°58'40" 47°20'0"	L	8	82-90
		Lac Veilleux	C05	-71°34'30" 47°23'40"	L	8	82-90
		Laflamme Lake	C08	-71°7'0" 47°19'0"	L	9	81-90
	Ontario	Batchawana Lake	C01	-84°23'33" 47°3'46"	L	10	80-90
		Little Turkey Lake	C03	-84°24'38" 47°2'43"	L	10	80-90
		Turkey Lake	C04	-84°23'47" 47°3'2"	L	10	80-90
		Wishart Lake	C02	-84°24'12" 47°2'47"	L	10	80-90
USA	Maine	Little Long Pond	1W1132	-68°4'41" 44°38'15"	L	8	82-90
		Tilden Pond	1E1133	-68°4'19" 44°38'5"	L	8	82-90
	New York Adirondack	Arbutus Lake	1A1052	-74°14'31" 43°59'15"	L	8	82-91
		Constable Lake	1A1017	-74°47'46" 43°50'0"	L	7	
		Dart Lake	1A1106	-74°51'27" 43°47'50"	L	9	
		Heart Lake	1A1102	-73°58'8" 44°10'56"	L	9	
		Rondaxe Lake	1A1110	-74°54'22" 43°45'50"	L	9	
		Moss Lake	1A1109	-74°51'0" 43°47'10"	L	9	
	New York Catskill Mt.	East Branch Neversink	143010	-74°27'19" 41°57'49"	R	7	83-90
		High Falls Brook	143105	-74°31'19" 41°58'33"	R	7	
	Michigan	Andrus Lake	2B3082	-85°2'35" 46°42'0"	L	7	83-90
		Buckeye Lake	2B2102	-85°44'19" 46°27'57"	L	7	
		Johnson Lake	2B1047	-85°2'38" 46°25'30"	L	7	
	Minnesota	Cruiser Lake	2A2063	-92°48'19" 48°29'54"	L	7	83-90
	Wisconsin	Luna Lake	2C2062	-88°57'35" 45°54'19"	L	7	83-90
		Sand Lake	2C1068	-89°41'15" 46°6'14"	L	7	
		Nichols Lake	2C1069	-89°39'5" 45°43'28"	L	7	
Colorado	White Dome Lake	4E2071	-107°33'11" 37°42'32"	L	5	85-90	
	Seven Lakes	4E2009	-106°40'52" 40°53'44"	L	5		
	Summit Lake	4E2060	-106°40'48" 40°32'45"	L	5		
	Upper Sunlight Lake	4E2070	-107°33'11" 37°42'32"	L	5		

3.2.2 The Statistical Analyses

3.2.2.1 Trend Tests

The seasonal Kendall tau (SKT) test for trends (Hirsch, et al. 1982) was applied to the data. In this nonparametric test, all possible pair-wise comparisons are ranked within a season. The ranks (increase, tie or decrease) for each season are summed. The ranks across all seasons were then combined as *tau* (τ), to evaluate whether a monotonic trend exists over time in the data. If the ranks sum to zero, then no trend exists in the data; if the sum is negative then a decreasing trend is present, while a positive sum of ranks indicates an increasing trend. With sufficient sample sizes (> 5 years), the tau statistic approximates the z-statistic from a normal distribution. The z-statistics can be used as indication of the strength of the trend, or how confident we are that there is a non-zero trend. The negative and positive z-scores preserve information about the trend direction.

Trends in concentrations are dependant on discharge. For the stream sites stream data were also analysed using the SKT when no significant correlation was found between stream discharge and concentration of the variable of interest, or when no discharge data were available. For the stream sites where the influence of flow on concentration could be quantified and was found significant, Alley's test (1988) was applied to detect trends that did not result from stream flow. Alley's test differs from the SKT by first removing the effects of flow from both concentration and time, then applying the SKT to the residuals.

Both the SKT and Alley's modification are robust for data characterised by seasonality and departures from normality and, with the serial correlation correction, for data with moderate levels of serial correlation. For most statistical tests, the probability of accepting the null hypothesis when the alternate hypothesis is true, referred to as significance level, is set by the scientist. In this case, we have chosen a level of 0.10. However, although often ignored, the probability of detecting a trend when it infact exist, or power of the test, is also of importance in interpreting trends in data. It is not easy to determine the power of nonparametric tests; furthermore power is impossible to quantify in any situation where the null hypothesis is a non-specific "not equal to". According to the test performance on simulated datasets, there is little loss of power using the SKT compared to parametric methods (Hirsch et al. 1982 and Hirsch and Slack, 1984), although the use of the serial correlation correction, which is generally low in the seasonally sampled lakes, combined with the smaller than recommended sample sizes (Hirsch and Slack, 1984), results in some expense of power. Therefore, in the results reported here, failure to detect a trend is not a reliable indication of no existing trend, however, trend detection at the 0.1 significance level provided at least 90% confidence that detected trends were significantly different from zero.

The SKT test was applied separately to each variable at each site, at a significance level of $p < 0.1$, without consideration for the fact that multiple tests were performed on the data set from each lake. Z-values from the normal approximation of the SKT were used to calculate correlations between observed trends and the other principal physical and chemical characteristics. Z-score values generally ranged from -3 to +3, exceeding these bounds when the tau statistic greatly exceeds its variance, indicating that the trend is very strong. Positive values indicate increasing trends, negative values decreasing trends, and near-zero values no apparent trend. This approach allows inclusion of trends from all sites in the analysis, without defining trend significance. Z-scores < -1.645 or > 1.645 are considered to be significant trends. Slope values for trends can be determined following a nonparametric approach (Hirsch et al. 1982). The slope estimates are the median of all the 2 point slope estimates from within-season pairs.

3.2.2.2 Correlations

One way of inferring what processes may be responsible for the observed trends is to evaluate the trend slopes for several variables at a given site. Due to the aforementioned problems with this approach, we have developed an alternative approach that is designed to look at region-wide processes and characteristics, and determine their relations to observed trends. For this, z-scores for trends in alkalinity, pH, Ca^{2+} , SO_4^{2-} , and NO_3^- were correlated with median site values of alkalinity, $\text{Ca}^{2+} + \text{Mg}^{2+}$, SO_4^{2-} , pH, watershed area, and with lake retention time at each site. Z-scores for surface water trends were also correlated to z-scores for both precipitation concentration and deposition load of SO_4^{2-} , Ca^{2+} , and H^+ and to z-statistics for trends in precipitation volume. The results of these correlations were used to determine whether deposition or precipitation trends were significantly related to surface water trends, or whether increasing or decreasing trends in surface waters occurred in waters of particular chemical ranges.

3.2.2.3 Cluster Analysis

Clustering techniques are used in exploratory data analysis to group sites based on the relationships among a large number of variables. The advantage of this approach is that observations in the data set, in this case lakes and streams, are clustered in multi-dimensional space based on a number of characteristics. Using Ward's technique (SAS, 1985), clustering was performed on the following site characteristics; median $\text{Ca}^{2+} + \text{Mg}^{2+}$, SO_4^{2-} , and pH, and in a second analysis, on the z-scores from the SKT trend analysis for $\text{Ca}^{2+} + \text{Mg}^{2+}$, SO_4^{2-} , pH, and NO_3^- . The number of resultant clusters in each analysis was determined from a plot of the cluster dendrogram.

The clustering on median site values was used as a quantitative approach to group the sites according to their chemical characteristics. For this exercise, European and North American sites were treated as separate groups. Clustering was performed on the median chemical variables of each site in an attempt to group the sites in a way that might predict trends. Although other variables may provide additional information, thus increasing the cluster resolution and predictability, they were not always available for all sites; missing values mean that the site would not be included in the analysis. Therefore only $\text{Ca}^{2+} + \text{Mg}^{2+}$, SO_4^{2-} , and pH were the only variables used in this analysis for both Europe and North America.

Clustering analysis was also done on the Z-scores from trend analysis for $\text{Ca}^{2+} + \text{Mg}^{2+}$, pH, SO_4^{2-} , and NO_3^- , in order to group the sites according to the trends observed at the sites. Alkalinity was not used for this analysis, because it is not measured at all sites. Any observations with missing values were excluded from the analysis. In order to avoid missing values for trends in NO_3^- , the trend was assumed to be zero when NO_3^- was consistently below detection limit.

4. Deposition of sulphate in Europe the last 100 years

A main cause for acidification of rivers and lakes are the deposition of SO_4^{2-} from anthropogenic sources. In Europe and North America, natural emissions are negligible when compared to the anthropogenic emissions (Semb, 1987). The principal source of man made sulphur emissions comes from the combustion of fossil fuels.

The economy and technology of the 19th century and beginning of the 20th century was built on coal. At first, bituminous coal (hard coal), through its dominant use in steam engines, was consumed in vast quantities as preferred fuel for the iron and steel industries, railways, steam ships, and thermal power stations. Brown coal, which had low calorific value and high water content, attracted little attention until the late 19th century, but its occurrence in shallow surface deposits favoured its use for local consumption. In the 1870's a method of compressing it into briquettes and expelling most of the water made its transport and use more practical. By 1900 brown coal was being burned in thermal-electrical stations. Small fields of oil had been known since the early part of 19th century, but the technology was too simple to exploit and was not developed until the 1930's. Oil had a greater calorific value, it was easier to transport and use than coal, and gave a large number of by products. These advantages together with a number of economic factors gradually made oil the fuel of choice in the early 20th century.

After the second world war, problems of air pollution caused by the combustion of coal began to draw public attention as events like the notorious London smogs became more common. Subsequent abatement strategies encouraged substitution of "clean" fuels for coal, building of high chimneys, and increased use of electricity generated in large thermo-electrical power plants. The fact that high chimneys increased the long range transport of pollutants and contributed substantially to regional acidification was not evident until the late 1960's when the local pollution problems began to merge into a pan-European one. Severe damage of rivers and lakes was detected in southern Scandinavia and this initiated extensive discussions on the environmental effects of pollution on regional scale (Odèn, 1968). In the 1970's most European governments launched national pollution control programmes and recognised internationally the urgent need for a pan-European action. These discussions have led to reduction in SO_2 emissions for most European countries during the 1980's. As a consequence the deposition of SO_4^{2-} has decreased throughout most of Europe during the same period.

Examples of development in depositions of oxidised sulphur at two EMEP grid squares (southern Norway and the border between The Netherlands, Belgium and France) the last 100 years (Figure 4.1, from Mylona, 1993). The deposition amount for this two areas differ with a factor of about 1000 (note the scale on the y-axis), but both sites show the same pattern, with increasing deposition from 1880, on to the late

1970's and thereafter a reduction in the deposition amount. The ICP on waters programme have data mostly from 1980 up to today which is only a very short period in ecological context. Responses in surface water chemistry in a catchment to reduced deposition may be influenced by earlier deposition amounts. This is background for much of the work on long-term trend work.

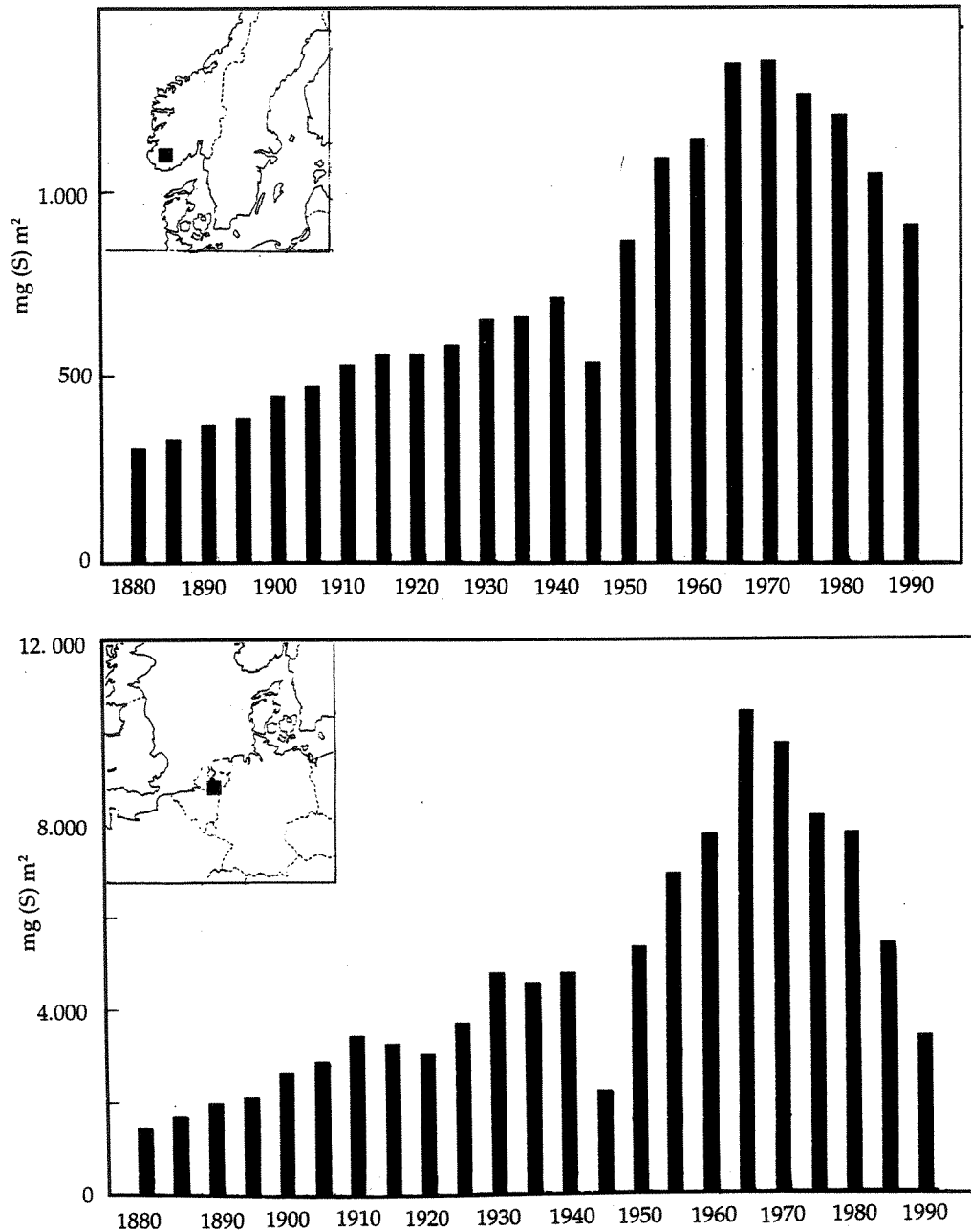


Figure 4.1 Calculated historical deposition of oxidised sulphur at two EMEP grid squares in southern Norway and on the border between The Netherlands, Belgium and France. Note the different scale on the two figures (From Mylona, 1993).

5. Chemical description of the monitoring sites in the Programme

The location of the monitoring stations in the different member countries and yearly mean values for the different stations for the years 1987 to 1992 are shown in Appendix B.

5.1 The Nordic countries

5.1.1 Norway

Large areas in Norway are underlain by slow-weathering bedrocks partly covered with thin layers of podzolic soil. Consequently, most of the lakes in Norway are very sensitive to acidification by atmospheric input. Most lakes in southern Norway are acidified or no longer have resistance against acidification. Nine of the Norwegian stations are on rivers and streams and one is a lake outlet. The stations are all in areas with yearly mean pH in precipitation < 4.7 . Four of the ten stations have mean pH < 5 . The other six have mean pH < 6 . The Norwegian stations are all low in Ca^{2+} indicating that they are sensitive to acidification. All stations have mean SO_4^{2-} concentrations below 10 mg/L. Alkalinity is < 0 $\mu\text{eq/L}$ on 5 of the stations and low (< 30 $\mu\text{eq/L}$) on the other 5 stations. Alkalinity is not measured when pH < 5.3 .

Central Norway

The acidification situation in central Norway is now stable and shows some tendency for improvement. There has been 30-40 % decrease in SO_4^{2-} in precipitation from 1974 to 1991 and the yearly mean pH value has increased from 4.2 to 4.5 in the same period (SFT, 1993). The decrease in SO_4^{2-} deposition has resulted in a decrease in SO_4^{2-} in runoff in the same period.

Two of the stations are strongly acidified with pH < 5.0 and alkalinity < 0 $\mu\text{eq/L}$. Calcium at these two stations are low, < 1 mg/L. The third station, Aurdøla, has pH > 6 and positive alkalinity at about 30 $\mu\text{eq/L}$. Precipitation in this watershed is moderate and the catchment area is big. Year to year variations in this river is therefore small and SO_4^{2-} and NO_3^- concentrations in runoff have been stable through the monitoring period (SFT, 1993).

Southern and south-western Norway

The sites in the southernmost part of Norway receives the highest yearly loading of SO_4^{2-} of all the Norwegian stations. The acidification situation in this area is serious, but stable, and the concentrations of SO_4^{2-} in surface waters today are the lowest that have been measured since the start of monitoring in 1980 which reflects the reduced deposition of SO_4^{2-} over the same period (SFT, 1993). Nitrate concentrations are highest in southern Norway (about 200 $\mu\text{g N/L}$) and lowest in northern Norway (about 10 $\mu\text{g N/L}$).

The Birkenes catchment is situated on granitic bedrock with low weathering rates, thin soil and low buffer capacity. At Birkenes there has been a 40 % decrease in SO_4^{2-} in precipitation from 1974 to 1991, while NO_3^- and NH_4^+ have had a small but not significant increase during the same period. Sulphate in runoff water has decreased parallel to the decrease in SO_4^{2-} in deposition. This decrease has not been followed by an increase in pH in runoff, probably because it has been accompanied by a significant decrease in base cations in the same period (SFT, 1993).

Western Norway

The rivers in western Norway have pH between 5.5 and 6, but Ca^{2+} is low, < 0.5 mg/L and so is alkalinity (< 10 $\mu\text{eq/L}$) which indicate that these rivers are very sensitive to acidification. In western Norway there has been no decrease in the total amount of SO_4^{2-} in deposition, consequently SO_4^{2-} does not show any decreasing trends in this area (SFT, 1993).

Northern Norway

Dalelva is situated in the northernmost part of Norway. Air pollution in this area originates from the Russian industry on the Kola peninsula. The deposition history in this area differs from that of the rest of Europe by the relatively recent influence of acid rain. Yearly mean pH in Dalelva is 6.0, Ca is about 1.5 mg/L and alkalinity is about 30 $\mu\text{eq/L}$. The SO_4^{2-} concentrations is about 6 mg/L, which is in the same range as Birkenes which receives the highest SO_4^{2-} loading in Norway.

5.1.2 Sweden

In most parts of Sweden, deposition of SO_4^{2-} gave rise to a sharp increase in SO_4^{2-} levels in lakes and rivers between the beginning of the century and the middle of the 1970's. By the late 1970's SO_4^{2-} concentrations in aquatic systems levelled off, then began to fall probably due to declining SO_4^{2-} in deposition (Monitor 1991). In southern and central Sweden SO_4^{2-} concentrations in surface waters have declined between 20-40% since then, which means they are now lower than in the middle of the 1960's. Base cation levels have also fallen substantially, consequently there has been no clear long-term trend in

either alkalinity or pH the last 20 years (Monitor 1991). In southern part of the mountainous regions of Sweden, however, the scale of acidification damage appears to have been increasing until as recently as the 1980s. The reason there has been no improvement may be that the beneficial effects of reductions in SO_4^{2-} in recent years have been offset by continuing acidification of soils and the associated decline in their capacity to neutralise acid inputs (Monitor 91). In Sweden, 4 stations are on rivers and 7 are on lakes, and the stations are located in three different areas.

South-eastern Sweden

The four stations in south-western Sweden have pH values between 6 and 7, Ca^{2+} concentrations are between 2-7 mg/L and SO_4^{2-} concentrations are high at about 10 mg/L.

South-western Sweden

The lakes and rivers in southern Sweden are the most acidified. Two of the three stations in this area have pH at about 4.5, Ca^{2+} is low (<1 mg/L) and SO_4^{2-} concentrations are high (>10 mg/L). Nitrate are mainly < 200 $\mu\text{g N/L}$.

Eastern Sweden

The three stations in eastern Sweden have pH between 6 and 7, Ca^{2+} concentrations between 1.5 and 4 mg/L. Sulphate concentrations are low (< 5 mg/L) and NO_3^- concentrations are low to moderate, up to 150 $\mu\text{g N/L}$.

5.1.3 Finland

The proportion of acidic lakes ($\text{ANC} < 0$ $\mu\text{eq/L}$) in Finland is 12 percent according to the lake survey conducted in 1987 (Forsius et al. 1990). The acidic lakes are spread around the whole country, even if the frequency is highest in the south-western, central and eastern regions. The acidic lakes are on average clearly dominated by organic anions (mean 88 $\mu\text{eq/L}$ vs. 54 $\mu\text{eq/L}$ of SO_4^{2-} and $\text{NO}_3^- < 1$ $\mu\text{eq N/L}$). In Finland, the total number of lakes over 1 ha area is about 56 000.

During the 1980's sulphate deposition decreased in southern Finland. At the same time, NO_3^- deposition increased and the deposition of base cations decreased. The result was an increase in rainwater acidity despite lower SO_4^{2-} deposition (Laurila 1990). In response to changing deposition, concentrations of SO_4^{2-} and base cations generally decreased and conductivity declined in small low-ionic lakes in the period 1979-1989. Apparently in these kind of lakes, acidification has continued despite decreasing SO_4^{2-} deposition (Roila 1992).

The lakes that have acidified anthropogenically are located mainly in high deposition areas of southern Finland. These lakes have on average lower concentration of organic matter and clearly the dominating anion is SO_4^{2-} . The watersheds are characterised by exposed bedrock. The three small lakes of the Finnish stations, Hirvilampi, Vuorilampi and Maekilampi are of this type. Of the two former, also diatom records exists showing the pH decline (Tolonen et al. 1986). Maekilampi has been limed in spring 1991 and the monitoring must be reconsidered.

All four Finnish stations are situated in the south-eastern corner of the country. The three small lakes have mean pH values between 5 and 6, and the mean SO_4^{2-} concentrations are all close to 10 mg/L. The bigger lake, Kivijaervi (SF-04), has a long record starting 1962 and the mean pH value is close to 7. All stations have Ca^{2+} values higher than 2 mg/L. The mean NO_3^- concentrations are low to moderate up to 330 $\mu\text{g N/L}$. The declining trends of SO_4^{2-} and pH in Hirvilampi and Maekilampi follow the same but more pronounced trend in deposition. The NO_3^- and Ca^{2+} trends in these lakes are weak and do not follow uniformly the trends in deposition (Roila, 1992).

The Finnish data in the ECE database suffers from missing values particularly in the early 1980's. Detailed information about individual stations is found in some publications (diatoms; Tolonen et al. 1986, heavy metal history; Verta et al. 1989, bottom fauna; Meriläinen and Hynynen, 1990), but in most cases the data from these lakes have been presented as a part of a lake population.

5.1.4 Denmark

The Weichel glaciation has divided Denmark into two regions with different susceptibility to acidification. East of the Weichselian terminal moraine, soils are usually clayey and calcareous, and their streams are alkaline and resistant to inputs of acidifying components. Trend analysis of pH and alkalinity of water samples taken over 15 years in two streams with alkalinity above 100 $\mu\text{eq/L}$ showed no trends of acidification (Rebsdorf et al. 1990).

West of the terminal moraine soils are sandy and leached and alkalinity is lower (mean 25 $\mu\text{eq/L}$). Although streams in such areas have medium alkalinity and are believed not to be vulnerable to acidification, there are documented significant decreases in pH and alkalinity in streams in this area over 12 years (1977 - 1988). Trends of pH and alkalinity in 4 western streams with mean alkalinities between 0 and 50 $\mu\text{eq/L}$ showed annual decreases in pH and in alkalinity (Rebsdorf et al. 1991).

Denmark's two river stations are situated west of the terminal moraine in the acid sensitive area of Denmark. Mean pH values are 6.5 indicating that these sites are not acidified. Both Ca^{2+} and SO_4^{2-} are high (ca. 10 mg/L and ca. 20 mg/L respectively). The NO_3^- values are very high (>3000 $\mu\text{g N/L}$) indicating that the NO_3^- saturation point of the watershed has been reached. Atmospheric deposition of NH_4^+ and NO_3^- plus agricultural influence on the station in Skaerbaek, Sepstrup Sande are the likely causes of this nitrogen saturation (Rebsdorf et al. 1991).

5.2 Countries on the European continent

5.2.1 Germany

In Germany 72 stations in 9 acid sensitive areas have been monitored since 1986. In 1990-1991, 43 of these stations were closed and 9 new station in 4 areas in former GDR were established instead. Altogether 40 stations in 13 acid sensitive areas in Germany now deliver data to the ICP programme. In Appendix B only data for the active stations in former FRG are presented (30 stations).

In general, results from the monitoring programme show improvement with regard to the acidification status which is also correlated with a decrease in SO_4^{2-} depositions (Zahn, 1990). Decreasing SO_4^{2-} concentrations in surface waters are in many cases correlated with increases in pH and decreasing Al-concentrations. Decreases of SO_4^{2-} concentrations were specially evident in the Black Forest and Hunsrück areas. Dry summers in 1988 and 1989 in combination with the mild winters with no or little snow, caused a strong decrease in pH and SO_4^{2-} due to greater influence of base flow in streams in Rothaargebirge, Kaufanger Wald and Gräfenbach in Hunsrück. An increase in N-deposition has resulted in increased NO_3^- concentrations in runoff (Zahn, 1990).

Of the 30 stations, presented in Appendix B, one is extremely acid with mean pH value < 4 (Schmerbach, Sch3). Ten stations have mean pH values between 4 and 5, the rest of the stations have mean pH values above 5. Many stations rich in Ca^{2+} have buffering capacity which provides some protection against transient pH depressions. The most affected areas have extremely high mean concentrations for SO_4^{2-} , NO_3^- and Al (Zahn, 1990). Ten of the stations have mean SO_4^{2-} concentrations higher than 10 mg/L, seven have concentrations higher than 20 mg/L, and three have mean SO_4^{2-} concentrations higher than 30 mg/L. Nitrate concentrations is above 800 $\mu\text{g N/L}$ for 25 of the stations. The highest mean NO_3^- concentrations are measured in Waldnaab in East Bavaria where yearly mean values up to 6650 $\mu\text{g N/L}$ have been registered.

5.2.2 Belgium

Belgium has 8 stations in the programme, 2 lake and 6 river stations. One lake station and 2 river stations are situated in the Eupen area, and 1 lake and 4 river stations in the Jalhay area. Seven of the 8 stations have mean pH values below 5. Only one station has a mean pH value higher than 6. Sulphate is high (>10 mg/L), indicating that these stations are located in areas with high deposition rates. The Ca^{2+} values are all higher than 2 mg/L, but this buffering is not enough to maintain pH values over 5. For the station Les Hes (B07), where the yearly mean Ca^{2+} concentration is between 5.9 to 6.6 mg/L, the mean pH range from 5.7 to 6.1, despite SO_4^{2-} concentrations between 14.3 and 16.8 mg/L and $\text{NO}_3^- > 1000$ $\mu\text{g/L}$ that are among the highest mean values registered for the Belgian stations.

5.2.3 The Netherlands

The most important group of acid-sensitive surface waters in the Netherlands are moorland pools. Although they were once oligotrophic they suffered from eutrophication due to changes in land use (from about 1900 to 1950) and thereafter from acidification (van Dam, 1988). Ammonium from agricultural sources contributed significantly to this acidification. The observed acidification in the present century is partly due to changes in management of moorland pools (van Dam, 1988). The moorland pools are all in non-calcareous sandy areas and are more susceptible to acid precipitation than ground-water fed lakes in the same area. The stations in The Netherlands are all seepage ponds, and all have acid waters.

The three Dutch sites differ in water chemistry. The station Kliplo has $\text{pH} > 5$, the three stations at Achterste Goorven have mean yearly pH values between 4.5 and 5 and Gerritsfles has $\text{pH} < 4.5$. Sulphate values are also high, especially for the three stations in Achterste Goorven, which all have mean sulphate concentrations higher than 10 mg/L. Chloride is lowest in the pool with the shortest residence time (Gerritsfles) and highest where the residence time is longest (Goorven) (van Dam, 1988). Also the total concentration of ions is highest in the latter pool. Sulphate is a dominant anion only in the surface waters of Goorven. In Kliplo, Cl^- dominates, while in Gerritsfles, SO_4^{2-} and Cl^- concentrations are equal. Nitrate is almost absent in the pools and HCO_3^- is of minor importance. Concentrations of Al and H^+ are high in Goorven and low in Kliplo (van Dam, 1988).

Some of the moorland pools have measurement records of pH, SO_4^{2-} and ammonium which extend from 1925-1930 and from 1974 up to now. Analysis shows that pH has decreased significantly over the last six decades (van Dam, 1988).

5.2.4 Austria

The Austrian stations are high mountain lakes (ca. 2000 m above sea level) situated on crystalline bedrock with low weathering rates. These lakes are therefore extremely sensitive to acidic inputs. Alpine lakes in Tyrol and Carinthia show signs of acidification. Data from 1938-1941 in Carinthian lakes are distinctly higher in pH and alkalinity than those from 1980-1986 (Psenner, 1989). Nitrate concentrations increased over the same period. The long-term diatom record for these lakes shows that acidobiontic species have only appeared recently (Psenner, 1989).

The 13 Austrian stations are situated in three different regions. Four of the stations show pH values < 5, three are between pH 5 and 6 and six of the stations show pH > 7.0. The stations with the lowest pH values (< 6) also have the lowest Ca²⁺ (< 2 mg/L) and SO₄²⁻ concentrations (< 3 mg/L), while higher pH stations also have higher Ca²⁺ and SO₄²⁻ concentrations. There seems to be a very good correlation between increasing Ca²⁺ and SO₄²⁻, but also with increasing pH in the Austrian stations. Nitrate is moderately high with yearly mean values mostly ranging from 100 to 400 µg/L with 3 stations with values > 1000 µg/L.

5.3 UK and Ireland

5.3.1 The United Kingdom of Great Britain and Northern Ireland

Large areas of UK are underlain by slow-weathering bedrocks, making the water sensitive to acidification. This combined with high SO_4^{2-} depositions through the last 150 years, have acidified many lakes especially in Scotland, Mid England and Northern Wales.

The UK has six lake stations, where four are situated in Scotland, one in England, one in Wales and one in Northern Ireland. Of the six UK stations two show mean pH values < 5 . Two stations have mean pH values between 5 and 6, and three stations have a mean pH value higher than 6. The sulphate values are all below 4 mg/L. Calcium for all the stations are below 1.0 mg/L, indicating that these lakes are situated in areas sensitive to acidification.

5.3.2 Ireland

In Ireland the principal areas with acid sensitive water bodies lie along the western seaboard and in Co. Wicklow. These areas are underlain by slowly weathering, bedrock, mostly granite and have a peaty or peaty podzolic soil overburden. The surface waters in these areas are characterised by their low alkalinity and consequential poor capacity to neutralise acid. A baseline investigation of the physio-chemical and biological characteristics of lakes in these areas, initiated in 1984-85, showed that the west coast lakes were unimpacted by artificial acidity, whereas the biological communities recorded in Glendalough Lake Upper on the east coast were consistent with those of a lake under acid stress (Bowman, 1986).

Overall there is little evidence that artificially acidified precipitation is directly responsible for surface water acidification in Ireland and the incidence of this effect is low. In Ireland the data show low acidification damage at all investigated sites except at two locations at Glendalough Lake Upper (Bowman, 1991).

In Ireland there are eighteen stations in 5 lake systems. Three have mean pH values < 5.0 . Six stations have mean pH values between 5 and 6, and nine stations have values between 6 and 7. The Ca^{2+} values are mostly higher than 2 mg/L, and the SO_4^{2-} and NO_3^- values are not very high for the same stations. All mean SO_4^{2-} concentrations are below 5 mg/L.

5.4 North America

5.4.1 Canada

In Canada the programme includes four river and ten lake stations which are situated in four different regions. With the exception of the four stations in the Algoma region, Ontario, (C01 - C04), the Canadian stations are placed in sensitive areas with low Ca^{2+} values. Six of the fourteen stations have Ca^{2+} values below 1.0 mg/L. As a result, many of them show mean pH values < 6 , despite relatively low SO_4^{2-} and NO_3^- mean values. Two stations have mean pH values < 5 , and seven stations have values between 5 and 6. Five stations have mean pH values between 6 and 7. None of the stations have mean SO_4^{2-} concentrations higher than 10 mg/L. The stations in the Algoma region have mean NO_3^- concentrations between 250 $\mu\text{g N/L}$ and 450 $\mu\text{g N/L}$ the others have concentrations lower than 100 $\mu\text{g N/L}$. The Nova Scotia stations (C10 - C14) all have mean NO_3^- values close to 10 $\mu\text{g N/L}$.

Biological monitoring and additional chemical monitoring is done in lakes in five major region across the country. River monitoring is also done in one of these regions. The Experimental Lake Area(ELA) is the most westerly of these monitoring sites, and acts as the baseline reference for the programme, because it contains acid sensitive lakes, but lies in a zone of relatively low SO_4^{2-} deposition ($< 10 \text{ kg/ha/year}$, wet). Data from ELA span the period from 1986 to present and are now being contributed to the ICP on waters. It is expected that biological data from other Canadian monitoring sites will be contributed in the future. ELA has also participated in chemical and biological intercalibration exercises conducted by the programme centre at NIVA.

5.4.2 USA

In US there are 22 stations in 7 regions in 6 different states. The lakes and streams are dilute systems with low Ca^{2+} values, mostly $< 2 \text{ mg/L}$ and low alkalinity, selected to represent acid-sensitive aquatic resources. Despite this, pH is high compared to many of the acid sensitive European stations. One station has yearly mean pH < 5 , 8 stations have pH between 5 and 6 and 10 station have pH between 6 and 7 and three stations have pH > 7 . According to SO_4^{2-} and NO_3^- values, the most affected lakes are situated in the western part of the US in the Adirondacks and Catskill mountains in New York (SO_4^{2-} , 3-6 mg/L, NO_3^- , 200-600 $\mu\text{g N/L}$). Lakes in Maine, Michigan, Minnesota and Wisconsin are also affected by SO_4^{2-} but the NO_3^- influence is much less here. Lakes in Colorado seem to be unaffected by airborne SO_4^{2-} , but the NO_3^- values in this area are higher than expected.

Maine

The two lake stations in Maine are underlain by granites. Yearly mean pH is about 6, Ca^{2+} about 1 mg/L and SO_4^{2-} < 4 mg/L. Long-term trend results (Kahl et al. 1993) show that there have been an increase in Alkalinity of < 2 $\mu\text{eq/L/year}$ and a decrease in SO_4^{2-} of < 1 $\mu\text{eq/L/year}$ in a period from 1982 to 1989. The sum of base cation also increased in the same period. These changes in aquatic chemistry were coincident with decreased concentrations of all solutes in precipitation during the 1980's. Other data on lakes and streams in Maine collected between 1930's to 1990 generally confirm these trends and further indicate that larger increases in alkalinity may have occurred in some lowland lakes since 1940.

New York - Adirondacks and Catskill

In a survey from 1982-1989 in New York, concentrations of SO_4^{2-} in surface waters generally decrease, while concentrations in NO_3^- generally increase consistent with decrease in SO_4^{2-} deposition in western U.S. (Murdoch and Stoddard, 1993). Reductions in SO_4^{2-} have not coincided with a recovery in alkalinity. Rather alkalinity has declined in some Adirondack lakes. This pattern is most likely due to increasing concentrations of NO_3^- that occurred in most of the Adirondack lakes (Driscoll and van Dreason, 1993). Combined acid anion concentrations were generally unchanged throughout the period of record, indicating both that the status of these streams with respect to acidic deposition is unchanged, and that NO_3^- is gradually replacing SO_4^{2-} as the dominant acid anion.

Michigan and Wisconsin

Lakes in Upper Midwest sampled between 1983-1989 all showed negative trends in SO_4^{2-} consistent with regional decline in SO_4^{2-} depositions. However, these trends occurred predominantly in higher alkalinity (100 to 225 $\mu\text{eq/L}$) non-seepage lakes and were associated with increases in alkalinity and pH in only one of the 8 lakes that were monitored (Webster et al. 1993). Alkalinity decreased in a second groups of lakes, usually correlated with decreased ($\text{Ca}^{2+}+\text{Mg}^{2+}$), a response that was associated with severe drought.

Colorado

The stations in Colorado are high elevation sites (ca 3000 m). The underlying geology consists of granite, quartzites or gneiss. Soils in these areas are very shallow, if existent, and tree coverage and other vegetation is also minimal. The predominate surface water trends observed in this region are increases in SO_4^{2-} . The lakes from this state are located in two different areas of the state, and the mechanisms for surface water increases differ among the two areas. In the northerly Mt. Zirkel area, surface water increases in SO_4^{2-} track recent increases in wet deposition of SO_4^{2-} . In the more southerly Weminuche region, S isotope studies indicate that local increases in surface water SO_4^{2-} result from watershed sources (Turk et al, 1993).

5.5 Participating countries recently reporting

5.5.1 Russia

Up to 1990 the situation in the Kola peninsula remained unstudied with regard to the acidification of freshwater. According to climatological and geological conditions, surface waters of the major Kola North territory are sensitive to acidification. Results from investigations in 1990-1992 (Moiseenko, unpubl.) show that SO_4^{2-} typically ranges between 3.5 - 5 mg/L and up to 10 mg/L close to emission centres. Assumed natural SO_4^{2-} concentrations in this area is about 0.5 - 0.75 mg/L. Of 370 investigated lakes, 4.8 % had $\text{pH} > 5$, while 10.5% had $\text{pH} < 6$. More than 30% are at the risk of acidification ($\text{ANC} < 30 \mu\text{eq/L}$).

A trend analysis of alkalinity in big rivers from 1960 to 1990 revealed a decrease in alkalinity from 1976. This is when the "Severonickel" company utilised an ore with higher sulphur content, that caused higher sulphur emissions into the atmosphere and intensified the water acidification processes.

From CIS, the ICP programme receive a yearly data report on the ecological conditions on 350 sites in 6 republics (Russia, Ukraine, Kazakhstan, Uzbekistan, Latvia, Estonia) (Annual report on the state of surface water ecosystems, according to hydrobiological indicators). The data report include an evaluation of the water quality based on biological parameters such as presence and number of indicator organisms.

These data does not fill immediately into our database as they are not similar in format as the other data, and has to be treated separately. For reference purposes water chemistry data from the Lake Baikal area has been added to the programme database.

5.5.2 The Czech Republic

A regional survey in Bohemia and Moravia from 1984 to 1990 (Vesely et al. unpublished) show that 5% of the areas are chronically acidified with $\text{pH} < 5.5$ and $\text{ANC} < 0 \mu\text{eq/L}$. About 50% of surface waters above 700 m above sea level have $\text{pH} < 6.0$. Acidified areas increase in size during wet periods such as spring thaw and heavy rain storms. The anthropogenic acidity of waters is mainly caused by acidic deposits from burning brown coal, while an other important source of acidification results from $\text{NH}_3\text{-N}$ depositions in several mountainous area (upto 50% in some localities).

The Czech republic has until now delivered data from yearly lake surveys of 100 lakes in Slovakia. These data also has to be treated separately at present, however, time serious are developing from now.

5.5.3 Hungary

The natural buffering capacity of surface water in Hungary is high, owing to high bicarbonate concentrations (Likcsò and Zotter, 1985). The programme centre receives data from one station in Hungary. The station is a water inlet to a drinking water reservoir named Csórrét. The pH in this water is about 7 and Ca^{2+} is about 50 mg/L.

6. Results

6.1 Dose - Response

In the "Three year report" the relationship between acidity index, pH and ANC was tested for all stations where water chemistry and invertebrate data were present. For ANC there was a correlation with the acidity index ($r=0.65$) (Figure 6.1.1) in spite of the heterogeneous material. The correlation between pH and the index was stronger than for ANC ($r=0.81$) (Figure 6.1.2). The presence of outliers may be explained by the high seasonal variability in water chemistry. Strong episodes, which have only a minor influence on the mean chemical values used in the diagrams, may have a strong influence on sensitive species. The analysis was also run on Norwegian data from the watersheds Nausta, Gaular and Vikedal. Because of the large homogeneity of these watersheds the correlation ($r=0.88$) between ANC and the acidity index is much better than for all of the stations. In the present report we have continued to investigate the influence of water chemistry on the invertebrates fauna of lakes and rivers by using detrendend correspondence analysis (DCA) and canonical correspondence analysis (CCA) (Larsen et al. in prep.).

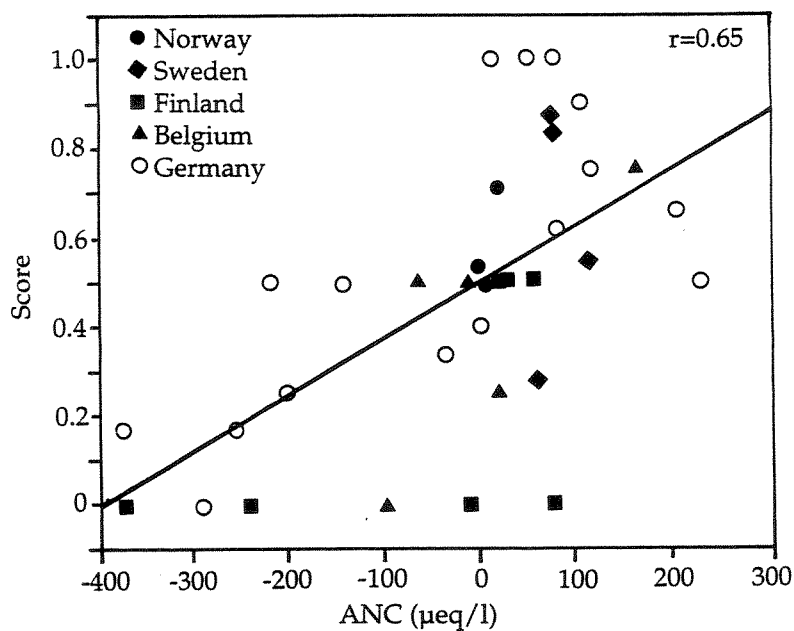


Figure 6.1.1 Correlation between ANC and acidity index

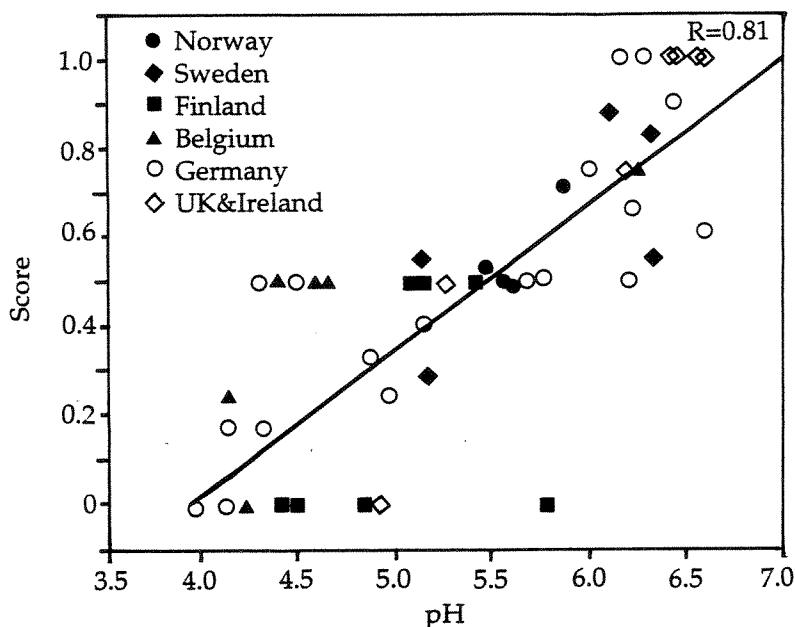


Figure 6.1.2 Correlation between pH and acidity index

6.1.1 Composition of invertebrate communities - examples from Norway

Detrended correspondence analysis (DCA) and Canonical correspondence analysis (CCA) were conducted to explain the distribution of invertebrates in Norway with respect to the chemical environment. DCA analysis on the invertebrate data showed one dominant axis of variation ($\lambda_1 = 0.365$) which explained 16.3 % of the variance in the invertebrate community and a second, smaller axis of variance ($\lambda_2 = 0.144$) explaining 5.6 % (Figure 6.1.5). The first axis had a high correlation ($r = 0.845$) with the chemical data, in particular pH ($r = 0.784$) and AI ($r = -0.759$). ANC and conductivity had lower correlations with $r = 0.464$ and 0.445 , respectively. Axis two was poorly correlated with most chemical data except AI which had a significant regression coefficient ($r = -0.402$)

The CCA analysis showed similar results to DCA by yielding two significant axes of variation (Figure 6.1.4). Axis one showed variation with $\lambda_1 = 0.272$, explaining 10.6 % of the variance of the biological data. Correlations with chemical data were high (Table 6.1.1).

Table 6.1.1 Correlation (*r*) between the two axes and chemical variables in CCA analysis.

Variables	Axis 1	Axis 2
Correlations with environment	0.889 *	0.663
pH	0.821	0.026
Ca	0.084	0.557 *
ANC	0.495	-0.110*
Al	-0.802 *	0.061
Conductivity	0.489 *	0.278*

*) Monte Carlo perm.test $p=0.01$ (99 permutations)

The importance of each parameter by using single chemical variables and comparing the size of the first constrained axis with the size of the second unconstrained axis are given in table 6.1.2.

Table 6.1.2. The correlation (*r*) and importance (%) of the different variables.

	<i>r</i>	λ_1	λ_2	λ_1/λ_2	%
pH	0.828	0.235	0.210	1.2	9.2
Al	0.817	0.225	0.225	1.0	8.8
ANC	0.598	0.103	0.303	0.33	4.0
Conduct.	0.581	0.102	0.299	0.34	4.0
Ca	0.619	0.045	0.364	0.12	1.8

pH appeared to be the best explanatory variable with importance value of 9.2 %, followed by Al at 8.8 %. The other parameters had predictive values of 4% or less. These results suggest that pH and Al are the best chemical parameters for explaining invertebrate distribution in Norway, but point on that the use of average chemical values tends to cloud the importance of these parameters as determinants of habitat tolerance.

The major division, biologically, within the 127 taxa, is between the taxa associated with high ANC, pH and conductivity on one side and those connected with high Al and low pH, ANC and conductivity on the other. Of those in this second group, Al seemed to be the most important chemical determinant in influencing the presence and abundance of many of the taxa. However, the importance of Al could be incorrect as this parameter is associated with low pH. Experiments show that high Al-concentration mainly influence the invertebrates negatively, but positive reaction are also reported (Rosseland et al. 1992). No patterns were detected with respect to species diversity and water chemistry.

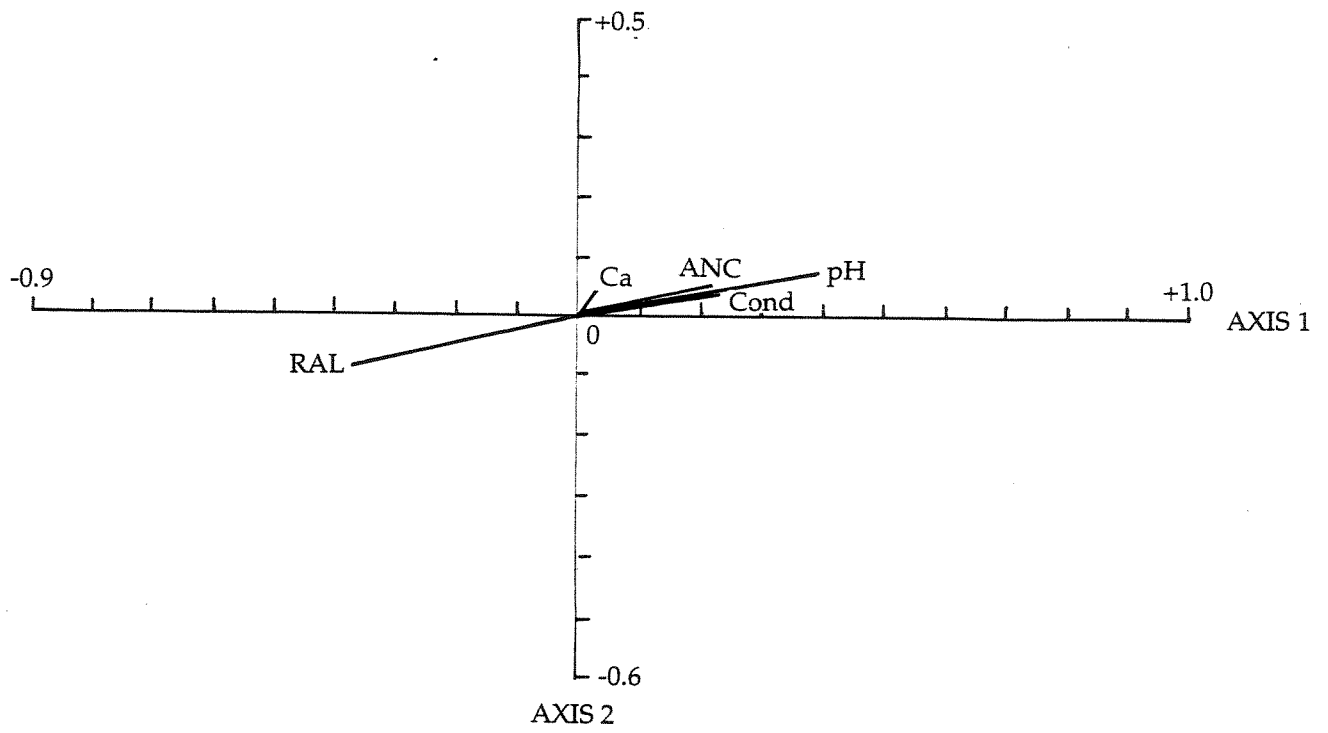


Figure 6.1.3. Detrended correspondence analysis and chemical variables.

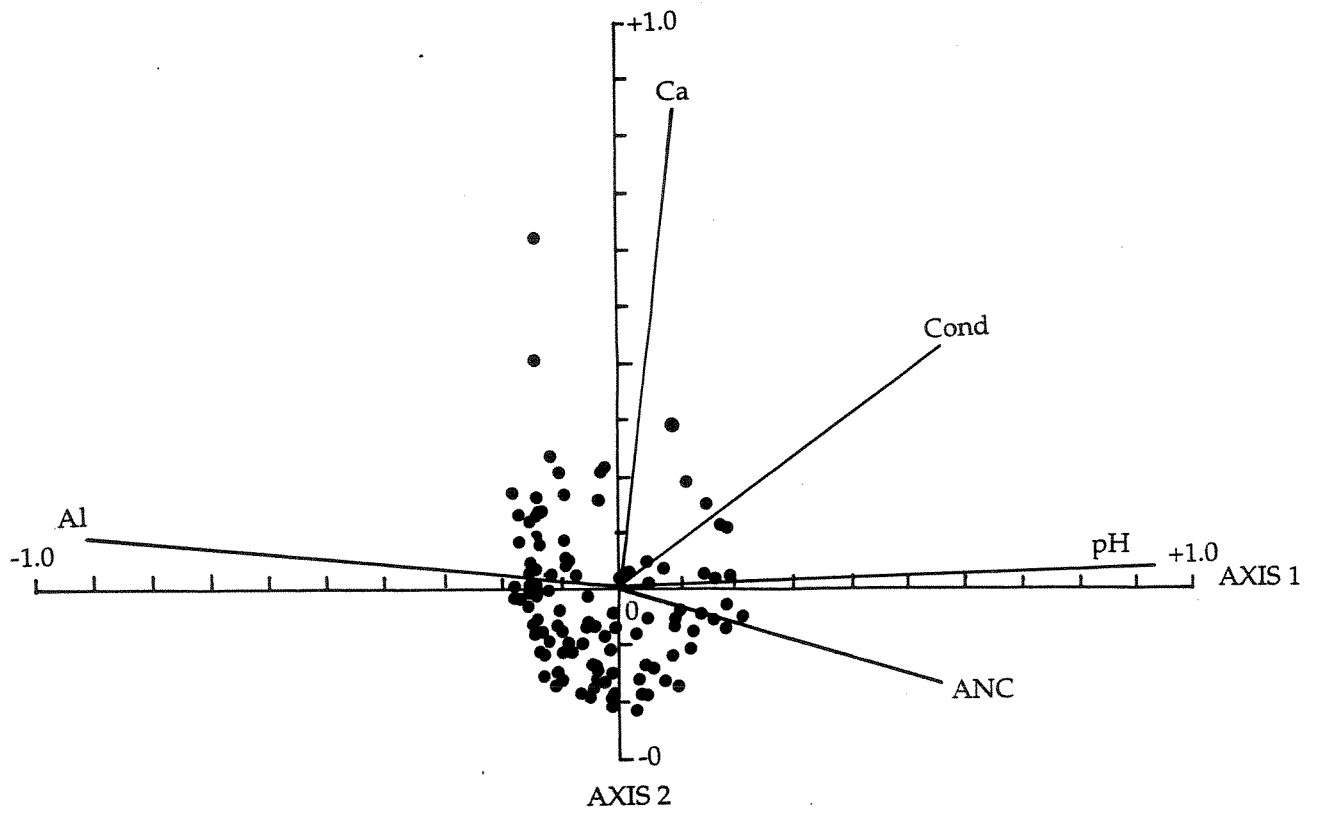


Figure 6.1.4. Canonical correspondence analysis and the chemical variables.

6.1.2 Predictions of pH from invertebrate assemblages

In an attempt to establish whether or not habitat pH could be predicted from the types and number of invertebrates that were present, data were subjected to weighted averaging regression and calibration with error estimation by bootstrapping (Birks et al. 1990). Preliminary data screening identified 8 "rogue" samples which were deleted. Final root mean square error of prediction based on bootstrapping was 0.302 pH units for this abbreviated data set. This is a similar result to that obtained by using SWAP model with modern diatom data. (Larsen et al. in prep.).

This work is preliminary and will be expanded in future publications. One problem with this type of analysis is that the critical limit of pH, Al etc. for the different taxa varies depending on conductivity, Ca²⁺ and humic substances in the water (Lien et al. 1992). The ability to predict chemical conditions (or the impact of extreme chemical conditions) from the Norwegian data only applies to sites in Norway. In other regions the invertebrate fauna composition may be different due to differences in the zoo geographic distribution patterns of species and differences in water quality (conductivity, Ca²⁺ and humic substances).

6.1.3 Critical limits of water chemistry to invertebrates

Norway

The occurrence and distribution of species in the Norwegian data set differed among sites with the same degree of acidification. This variance was due to other factors than acidity. Most important among these are other chemical factors, the physical environment, location, watershed productivity and biological interactions (predation, food availability and competition). Clearwater sites dominate in western Norway. Here the presence or absence of invertebrate species is strongly influenced by Ca²⁺ concentration the species most sensitive to acidification (i.e. index value 1). At values of pH > 5.5 the number of sensitive species present was higher in water with Ca > 0.7 mg/L than in water with Ca < 0.7 mg/L (Table 6.1.3 from Lien et al. 1992). Species occurring at higher Ca²⁺ content were primarily represented by snails, mussels, leeches and mayflies. In addition some crustaceans were found to respond in a similar way (Økland and Økland, 1986).

Table 6.1.3 The occurrence of invertebrates at acidification index 1 at high Ca^{2+} (>0.7 mg/L) and low Ca^{2+} (<0.7 mg/L). The acidification index 0.5 and 0 are independent of Ca^{2+} levels. Data are based on 71 sites in western Norway (from Lien *et al.* 1992).

Acidification index	1	1	0.5	0
Species/Group	high calcium	low calcium	varying calcium level	
Margaritana margaritifera	*			
Pisidium spp	*	*	*	
Lymnaea peregra	*			
Gyraulus sp.	*			
Helobdella stagnalis	*			
Glossiphonia complanata	*			
Baetis rhodani	*	*		
Baetis fuscatus	*			
Baetis niger	*			
Heptagenia fuscogrisea	*	*	*	*
Heptagenia sulphurea	*	*	*	
Leptophlebia sp.	*	*	*	*
Ephemerella aurivilli	*	*		
Caenis horaria	*			
Diura nanseni	*	*	*	
Taeniopteryx nebulosa	*	*	*	*
Brachyptera risi	*	*	*	*
Amphinemura sulcicollis	*	*	*	*
Amphinemura borealis	*	*	*	*
Nemoura cinerea	*	*	*	*
Protonemura meyeri	*	*	*	*
Leuctra nigra	*	*	*	*
Isoperla sp.	*	*	*	
Leuctra hippopus	*	*	*	*
Leuctra fusca	*	*	*	*
Rhyacophila nubila	*	*	*	*
Philopotamus montanus	*			
Neureclipsis bimaculata	*	*	*	*
Plectrocnemia conspersa	*	*	*	*
Polycentropus flavomaculatus	*	*	*	*
Polycentropus irroratus	*	*	*	*
Hydropsyche pellucidula	*		*	
Hydropsyche siltalai	*		*	
Lepidostoma hirtum	*			
Notidobia ciliaris	*	*	*	*
Limnephilus sp.	*	*	*	*
Oxyethira sp.	*	*	*	*
Athripsodes sp.	*	*	*	*
Hydroptila sp.	*			
Itytrichia lamellaris	*			
Apatania sp.	*	*	*	
Tinodes waeneri .	*	*	*	
Glossosoma intermedium	*			

The fauna present in waters with pH between 5.0 to 5.5 also varies, but to a lesser extent. At pH of < 5.0 the Ca²⁺ content of water had little effect on fauna compositions. This means that factors associated with acidic water, like low pH and high Al, are relevant variables for determine the presence of taxa in these localities.

The critical limits of acidic water is also modified by moderate content of humic substances, TOC, (Hobæk and Raddum ,1980; Sutcliffe et al. 1986; and Hämäläinen and Huttunen, 1990; Johnson et al. 1993). TOC appears to eliminate the toxic effects of Al (Rosseland et al. 1992). The dataset used in Lien et al. (1992) has been divided by localities into clearwater and humic sites. Comparison of these two datasets with respect to mean critical limit of pH, Ca²⁺ and ANC against the different values of the acidification index is shown in Figure 6.1.5 to 6.1.6 (modified from Lien et al. 1992). In clearwater lakes the mean critical limit of pH is about 5.8 for the most sensitive species, while a moderate increase in TOC can reduce the limit to pH 5.6 (Figure 6.1.5). The same picture as for critical limit of pH is also observed for the other indexes of acidification. Index 0 as an example, occurred at higher pH in clearwater than in humic water sites.

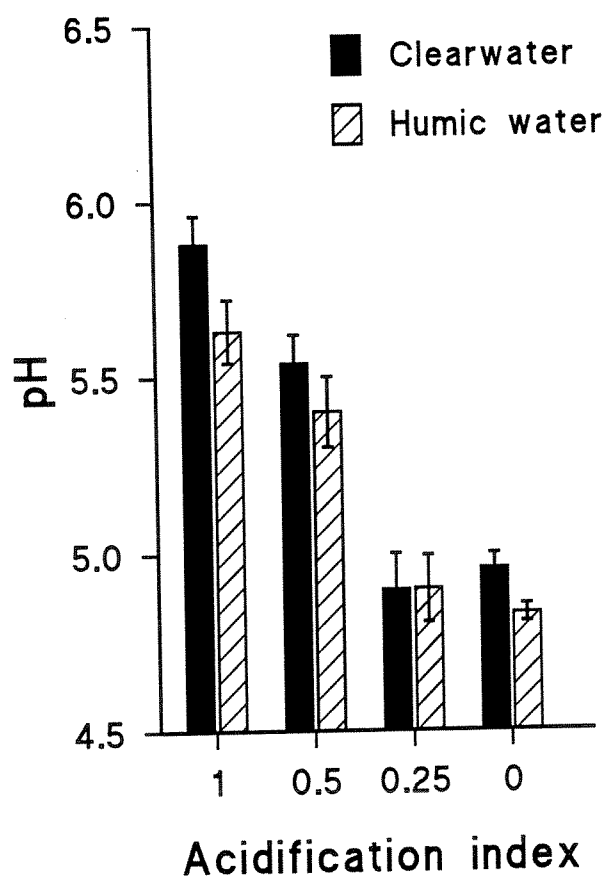


Figure 6.1.5 Distribution of mean pH in clear and humic waters among sites with different acidification index. Bars indicate one standard error unit.

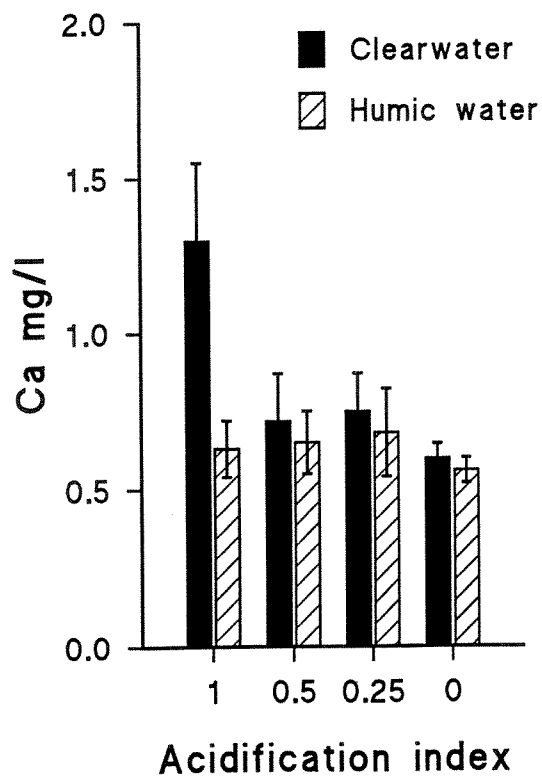


Figure 6.1.6 Distribution of mean Ca in clear and humic waters among sites with different acidification index. Bars indicate one standard error unit. Records similar to 6.1.3.

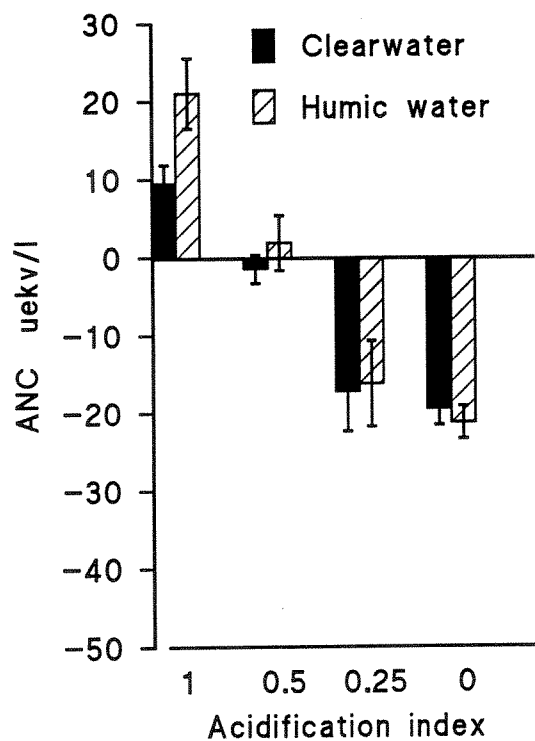


Figure 6.1.7 Distribution of mean ANC in clear and humic waters among sites with different acidification index. Bars indicate one standard error unit. Records similar to 6.1.3.

In humic water no relationship was observed between Ca^{2+} concentration and the acidification index (Figure 6.1.6). In clear water sites, however, acidification index 1 was accompanied with the highest content of Ca^{2+} . No relationship existed between other values of the acidification index and the Ca^{2+} content in clearwater.

In clearwater lakes, sites containing index 1 fauna had positive values of ANC (mean ANC > 10 $\mu\text{eq/L}$), while in humic water an ANC value of > 20 $\mu\text{eq/L}$ was required to support the most sensitive fauna (Figure 6.1.7). Animals which scored 0.5 were found in both humic and clear water at values of ANC=0. Sites with lower acidification scores had negative values of ANC \approx -20 $\mu\text{eq/L}$ (Figure 6.1.7). The above examples illustrate how humic content and Ca concentrations can influence the sensitivity of invertebrate taxa to acidification.

Watersheds in western Norway are dominated by dilute waters. Here, acid sensitive species (Index 1) have adapted to ANC levels of \approx 10 $\mu\text{eq/L}$. In most other parts of Norway the ANC is higher. These areas also have other sensitive species that require higher ANC. A minimum limit of ANC of about 20 $\mu\text{eq/L}$ is necessary for these areas.

In areas where different taxa of snails, mussel and sensitive crustaceans occur (or were historically present), critical limits of ANC are much higher. Snails are seldom recorded and gammarus is not found at all at the Norwegian monitoring sites. These invertebrates normally require pH > 6 and Ca concentrations far above 1 mg/L (Økland and Økland, 1986). At sites in other countries these species are sometimes frequent. High variation in ANC exists at sites with gammarus and snails, but usually far above 20 $\mu\text{eq/L}$. To protect this part of the invertebrate fauna, the ANC limit must be set at higher values in regions where the animals are distributed.

Other countries

The invertebrate communities in Swedish lakes were classified by Johnson et al. (1993) by use of CCA and two-way indicator species analysis (TWINSPAN) (Hill 1979). Of four environmental variables pH showed a strong correlation with axis one, in accordance with the results from Norway. Further, there it was a good agreement between the acidification index and the invertebrate communities classified by TWINSPAN.

In an attempt to further analyse critical limits of water quality we have examined the distribution of the most sensitive snails and gammarus in different countries. These invertebrates are frequent in the dataset from Germany, Ireland and Austria. They also occur in the Swedish material. The water chemistry of the Irish sites resembles the water quality found in Norway with respect to influence from the sea. However, the

alkalinity in Ireland is generally higher and negative ANC is relatively rare. The sensitive species were here recorded at sites with ANC and pH down to about 20 $\mu\text{eq/L}$ and pH 5.7, respectively, according to observations from Norway. The mean ANC and pH at sites with record of sensitive invertebrates in Ireland was 136 $\mu\text{eq/L}$ and 6.2.

The water chemistry of German rivers and streams is extremely variable. Extremes of pH below 4 and close to 8 and ANC values range from about -100 to + several hundred exists. In some cases large amplitudes in ANC and pH occurred over short periods at a site. For this reason average chemical values are of little use in interpreting biological conditions. It is often the magnitude of extreme conditions that has the highest impact on sensitive species. Therefore we chose to compare invertebrate community structures with water chemistry from a period of poor water quality that can have influenced the community before invertebrate sampling. The data were examined for relationships between acidification index and its number of taxa, pH and ANC (Figs. 6.1.8-6.1.10). These relationships were then used to define critical limits for the most sensitive invertebrates.

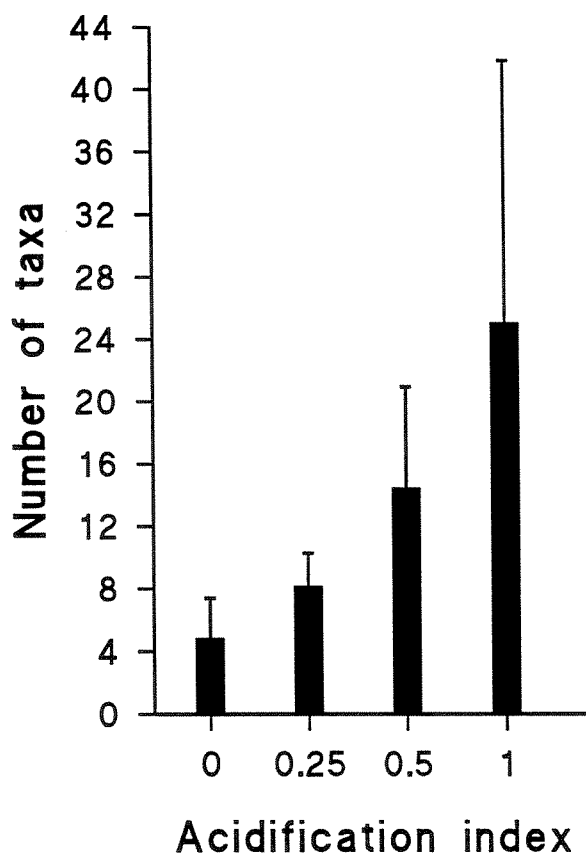


Figure 6.1.8 Acidification index against number of recorded taxa in Germany. Standard deviation is indicated.

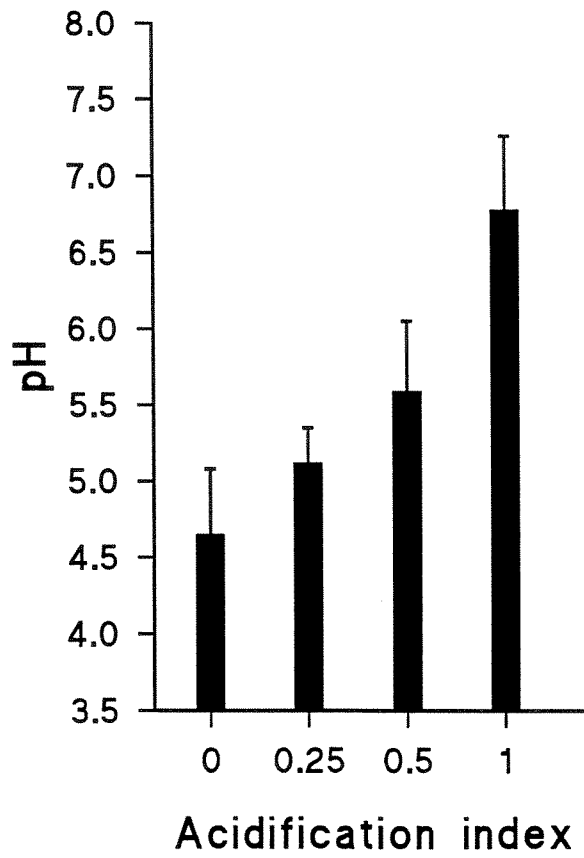


Figure 6.1.9 Acidification index against mean pH in Germany. Standard deviation is indicated.

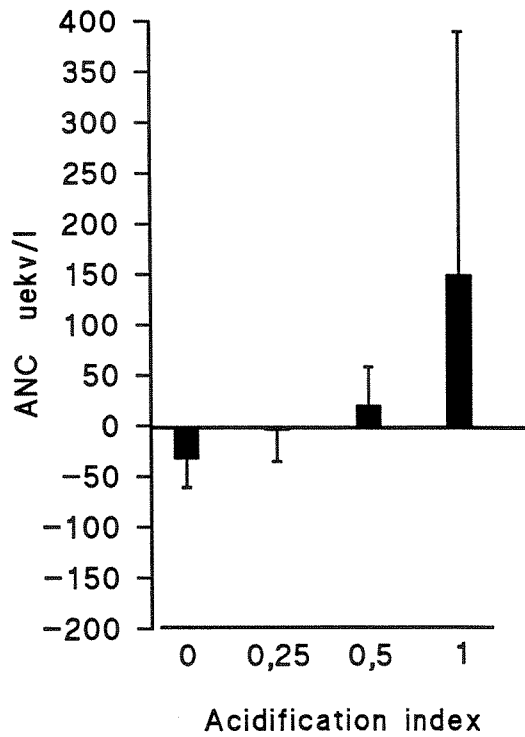


Figure 6.1.10 Acidification index against mean ANC in Germany. Standard deviation is indicated.

The mean number of taxa recorded in the samples from Germany with acidification index 0 was 4.8 ± 2.5 , a surprisingly low value (Figure 6.1.8). In Norway, Sweden and Ireland this values was 6.4, 10.7 and 12.4 respectively, indicating lower damage at this acidification level in these countries. For samples with an acidification index of 0.25 in Germany, the number of taxa was 8.2 ± 2.1 , which was significantly different from the 0 level ($p < 0.001$). Number of taxa recorded at the 0.5 level was 14.5 ± 6.4 . This was also significant different from the 0.25 level. Acidification index 1 stations, however, showed mean number of 25.1 ± 16.7 taxa, a value highly significant from the 0.5 level ($p < 0.001$).

The total number of taxa on the German taxa list is 223. Compared with this list the recorded fraction of taxa in each sample seems very low especially in samples with acidification index 0 (about 2%). The number of taxa recorded at index 1 sites was also low (11% of the possible). On the other hand, this percentage was from 10 to 25% in the other countries.

By definition, the number of taxa lessens with decreasing index. However, the strong decline in the German dataset, shown in figure 6.1.11, is much greater than the one seen in Norway and Sweden. This indicates that damage to the invertebrate community is relatively higher in Germany than in Scandinavia in strongly acidified localities.

There is a strong relationship in the German data between acidification index and the minimum recorded pH. Values for each of the scores are significantly different from another ($p < 0.001$), however, a very poor relationship exists if values of mean rather than extreme pH are used. The result is quite different from what we have seen in Scandinavia with respect to concentrations. Mean pH at index 0 was 4.65 in Germany. Further, the pH at index 0.25, 0.5 and 1 were 5.12 ± 0.23 , 5.59 ± 0.46 and 6.78 ± 0.49 , respectively. There were no significant differences between the mean pH at index 0 and 0.25 at Norwegian sites. The mean pH for these scores was 4.9. At index 0.5 and 1 the pH was 5.45 and 5.7. This suggests that the indexes are distributed over a wider range of the pH scale in central Europe than in countries with naturally low buffer capacity.

The relationship between acidification index and ANC in Germany (Figure 6.1.11) shows that a strong significant difference ($p < 0.001$) exist between the mean values of ANC at each of the index values. As with pH the index spans a much wider range of ANC in Germany than in Norway $-32.4 \pm 28.37 \mu\text{eq/L}$ at index 0 and $150 \pm 240 \mu\text{eq/L}$ at 1 vs. -30 to $20 \mu\text{eq/L}$ in Norway. The results indicate that the critical limit of ANC ($20 \mu\text{eq/L}$) proposed for Norway and Scandinavia need not apply in central Europe or to waters where the fauna is adapted to high ionic content.

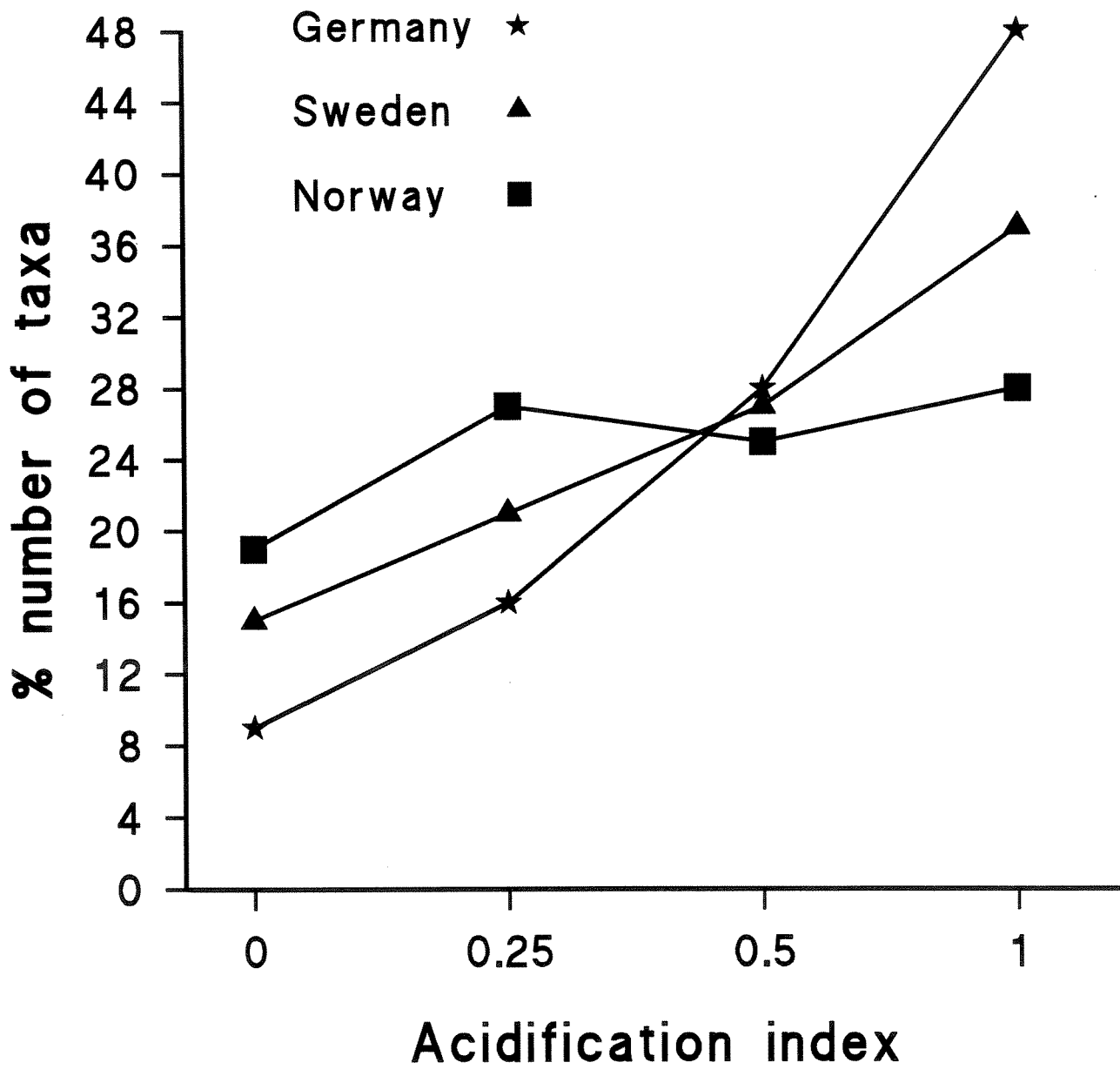


Figure 6.1.11 Decline in number of taxa at different acidification in Germany, Sweden and Norway.

The reason for this is most likely that the undamaged sites in Germany have higher pH and ANC than the corresponding sites in Norway. Consequently these sites in Germany have the potential of a higher diversity of sensitive taxa as well as a higher number of taxa in general. On the other hand, the most acidified sites in Germany often have lower pH than seen in Scandinavia. The results of this is probably that some of the species living at index 0 sites in Scandinavia have been wiped out in Germany. The potential damage with respect to invertebrate diversity is, due to this, very high in the most acidified regions of central Europe (conf. figs. 6.1.8 and 6.1.11). Consequently the damages occur at higher pH and ANC in these regions than in Norway.

The tolerance limit of pH to the most sensitive invertebrates is set to 6, based on the available data. Figure 6.1.12 shows that ANC can vary considerably at this pH. A critical limit of ANC=50 µeq/L is, however, proposed for waters in central Europe, based on the correlation between pH and ANC.

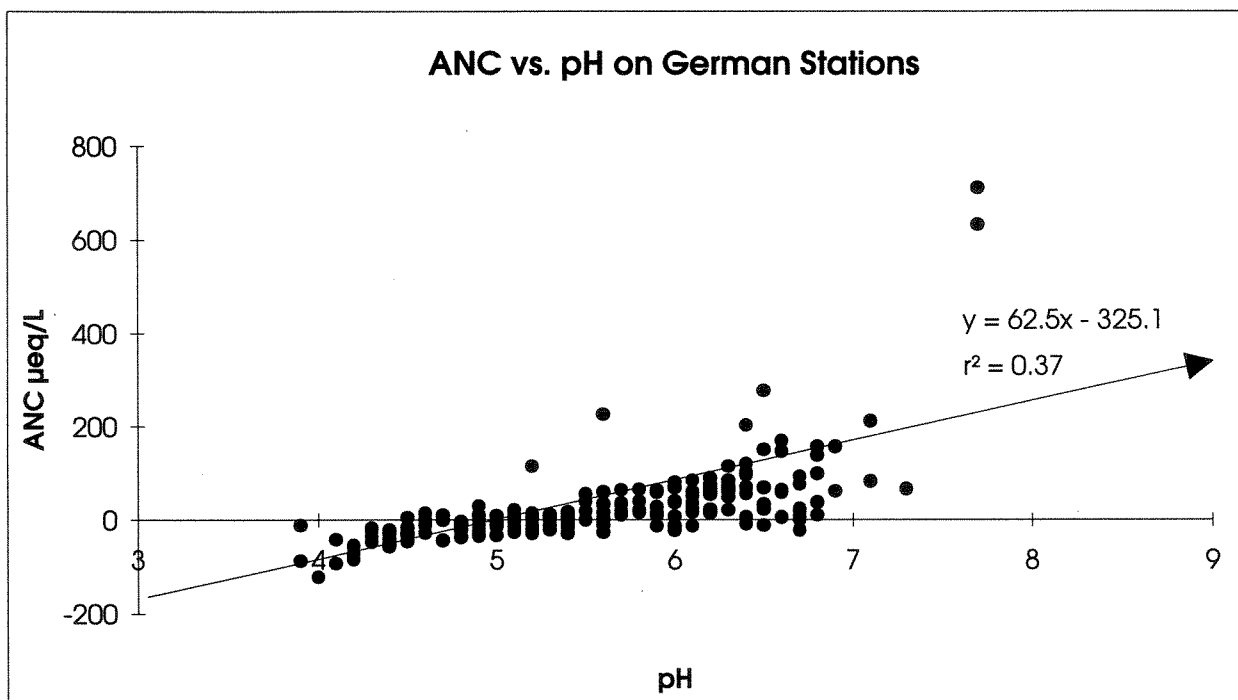


Figure 6.1.12 ANC vs. pH on stations in Germany.

Based on the results described above we have plotted pH and ANC for East Bavaria and Black Forest in Germany and indicated tolerance trends for the most sensitive species (pH=6 and ANC = 50 μ eq/L) (Figure 6.1.13). Large numbers of station values fall below these limits, indicating that biological damage from acidification is widespread in these watersheds.

The distribution of samples at different acidification scores in Norway, Sweden, Germany and Ireland is shown on figure 6.1.14. A high percentage of the samples available from Ireland have the index 1 and show low acidification of the monitored watersheds. Samples with index 1 and 0 are about equal in Norway (ca. 40% each), while relatively few samples gave the index 0.25 and 0.5. In Germany the index 1 and 0 is found in 30 and 25% of the samples, while most samples had index 0.5.

The distributions of the samples (sites) in the different countries as shown in fig 6.1.14 allow the evaluation of trends in acidification by long-term monitoring.

Ireland with its low influence of acid deposition can serve as a reference country with respect to invertebrates.

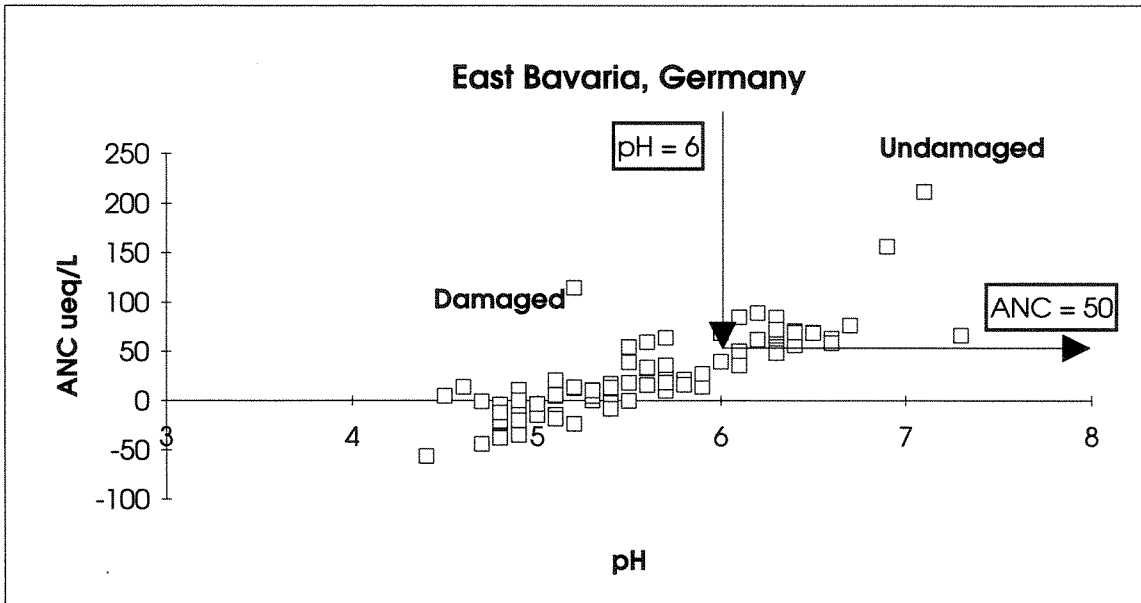
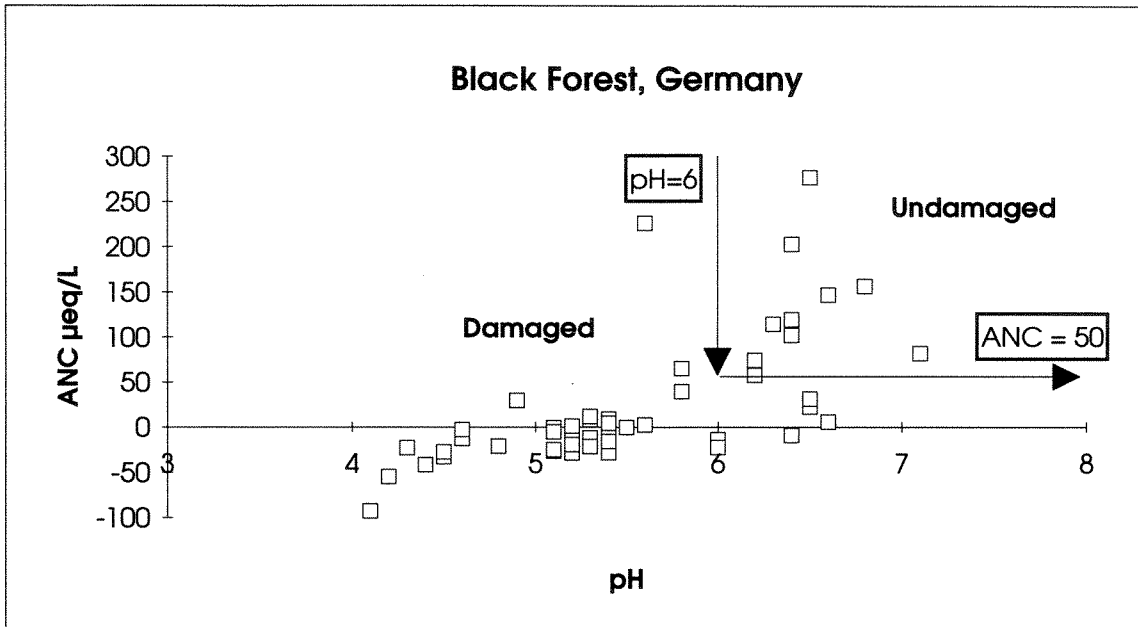


Figure 6.1.13 Scatterplot of ANC against pH for East Bavaria and Black Forest in German. The datapoints are from the period 1987-1991. The lines at pH=6 and ANC=50 indicates the lowest limits where we can expect to find undamaged fauna.

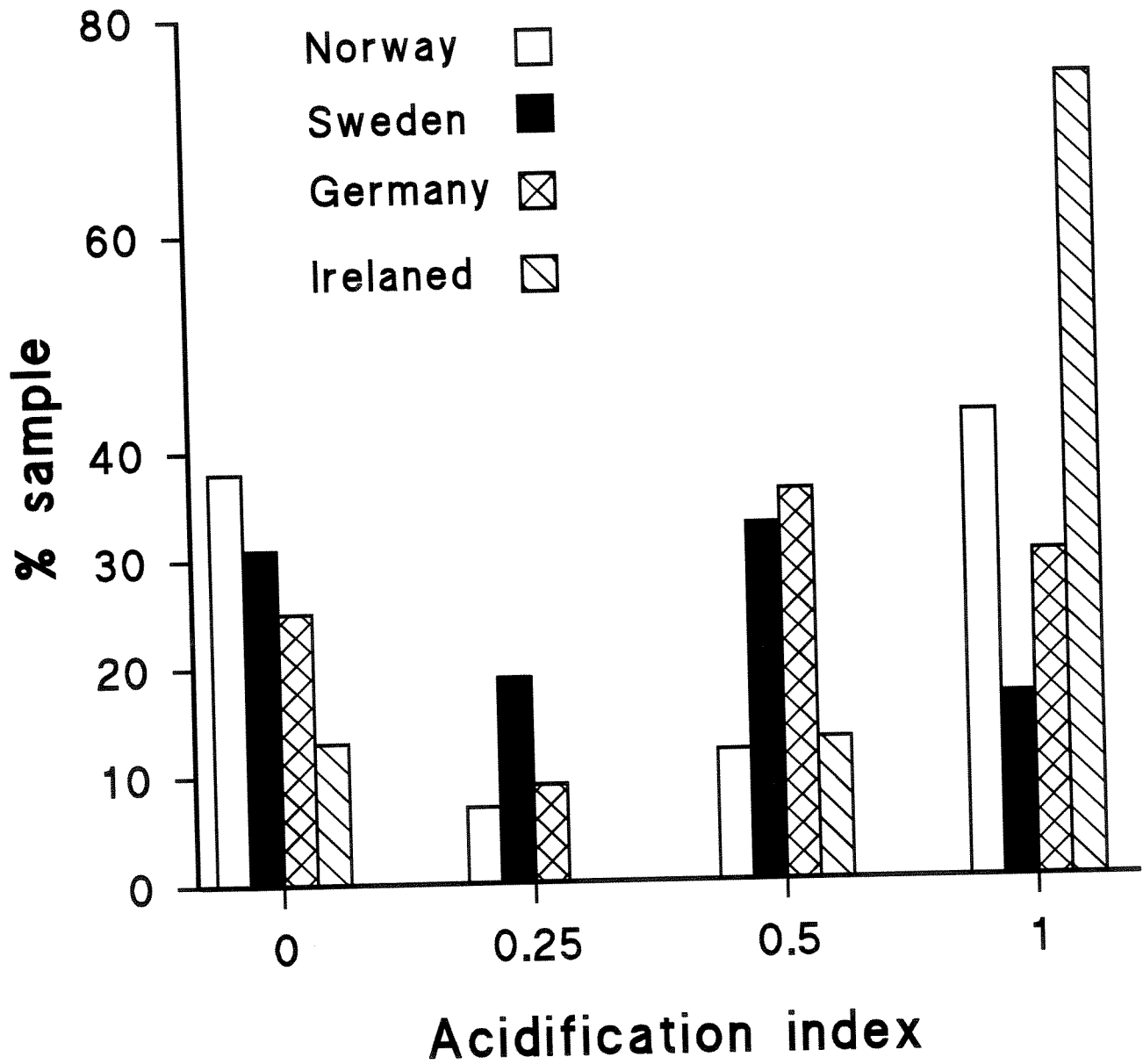


Figure 6.1.14 Distribution of samples at different acidification scores in Norway, Sweden, Germany and Ireland.

6.1.4 Observations of the acidification in Russia

The acidification scores at 238 sites in lakes and rivers from the western part of Russia are available in the database. Table 6.3.1 shows the percent of sites for the four acidification scores.

Table 6.1.4 Number of sites in percent with acidification index 0, 0.25, 0.5, and 1 in Western Russia (total number of sites), in Kola (49 sites) and in Karelia (138 sites).

Region	Index			
	0	0.25	0.5	1
Western Russia (Total)	3	3	7	87
Kola	6	4	2	88
Karelia	2	4	11	83

The observations from Russia suggest low acidification in the majority number of sites. However, strongly acidified waters exist in limited areas in Kola and Karelia. The acidification scores 0 and 0.25 are also recorded at sites in other regions of Russia, but in very low numbers.

Complete fauna lists are not in the database from Russia, but record of sensitive species is reported for stating the acidification index. The samples are in most cases from one sample trip, and time series for monitoring of specific sites/watersheds are currently lacking. Table 6.3.1. therefore only shows an overview of the general acidification of waters in Western Russia.

6.2 Long-term trends and comparison of acidification trends in water chemistry

Statistical work on long-term trends in this report is limited to interpretation of the chemical data. There were three objectives for this data investigation: (1) identify trends in chemical variables among the various surface water bodies, and describe these in terms of the major ion chemistry at each site; (2) relate these trends to trends in deposition or any other characteristics responsible for a plausible cause and effect relationship; (3) group sites by their physical and chemical characteristics and by the trends observed. The characteristics of these groups might then reveal external factors that influence the trends. In addition these site groups might be used to infer the regional extent of trends.

6.2.1 Site Characteristics

Most of the lakes included in the trend analysis are small lakes less than 1 km² in area. Acidic sites, with alkalinity less than zero are found in Norway, Sweden, Finland, Canada and the U.S. (Table 6.2.1). Alkalinity at the Germany sites is high, even if pH is less than 5.0. Except for the German sites, the remaining sites generally show alkalinity < 100 µeq/L.

The highest concentrations of SO₄²⁻ are found in Germany and the Netherlands, with mean values ranging from 150 - 650 µeq/L. The Danish site and Swedish sites have moderate high SO₄²⁻, at 100 - 300 µeq/L. The Norwegian sites have lower SO₄²⁻ concentrations than many other ICP sites and the correlation between SO₄²⁻ and pH is strong, suggesting that SO₄²⁻ loading may be responsible for the acidity (Wright et al. 1988). This correlation is not so strong among the Canadian sites. The Canadian sites are all lakes, whereas the Norwegian sites are clearwater rivers and streams. Because organic carbon concentrations is higher at the Canadian sites, DOC and SO₄²⁻ both influence the acidity of these sites.

Nitrate values tend to be low at US. and Canadian monitoring sites. For about half the U.S. and Canadian sites, NO₃⁻ values are below detection limits during most of the year, so mean values are near or below detection limit. At most of the European sites, the mean NO₃⁻ values are above detection limit values.

Table 6.2.1. Mean pH, alkalinity, $Ca^{2+}+Mg^{2+}$, SO_4^{2-} and NO_3^- , in $\mu eq/L$ for the year 1989 (sites with* are from 1990) watershed (WS) (km^2) and retention time (RT) (yr) for the sites included in the trend analyses.

Country	Area	Site Name	pH	Alk	Ca+ Mg	SO4	NO3	WS area	RT
Norway	Buskerud	Langtjern	4.5	< 0	105	124	14	4	-
	Oppland	Aurdøla	6.0	27	74	17	3	-	-
	Telemark	Storgama	4.7	< 0	44	72	9	-	-
	Aust Agder	Birkenes	4.7	< 0	66	61	2	-	-
		Tovdalselva	4.9	< 0	71	77	14	1888	-
	Rogaland	Vikedalselva	5.3	< 0	61	46	11	119	-
Sogn og Fjordane	Gaular	5.3	4	39	28	5	-	-	
	Nausta	5.6	6	41	27	4	273	-	
	Trodøla	5.4	1	38	27	4	-	-	
Sweden		Alsteraaen,Getebro	6.4	102	455	211	13	119	-
		Alsteraaen,Stroemsborg	6.3	103	377	232	19	1345	-
		Fraecksjoen	6.1	46	334	215	10	4	-
		Aneraasen,Haersvatn	4.4	< 0	128	188	13	1992	-
		Delangeraaen, Iggersund	6.7	158	274	104	10	2	-
Finland		Hirvilampi	5.0	-9	151	191	12	0.4	2.3
		Kivijaervi *	6.7	n.a.	509	77	22	494	3.3
		Maekelampi	4.9	-14	125	159	24	0.6	2.4
		Vuorlampi	5.2	4	164	203	18	0.4	0.2
Denmark		Skaerbaek	5.5	16	470	305	85	4	-
FRG	Harz	Alte Riefenbeck	7.7	986	1423	295	71	0.3	-
		Alte Riefenbeck	7.6	642	1005	355	96	3.3	-
		Alte Riefenbeck	7.6	763	858	352	103	5.7	-
		Varleybach	7.1	215	882	578	164	2.4	-
		Varleybach	7.5	632	1199	653	139	3.0	-
		Grosse Schacht	6.7	215	559	283	117	5.0	-
	Taunus	Rombach *	3.9	92	335	432	226	0.6	-
Hunsrueck	Traunbach	4.3	40	229	212	50	1.5	-	
The Netherlands		Achterste Goorven A	4.9	58	194	333	5	-	-
		Achterste Goorven B	5.5	59	172	364	7	-	-
		Achterste Goorven E	5.6	105	213	401	4	-	-
		Gerritsfles	4.3		115	192	10	-	-
		Kliplo	5.6	19	92	146	6	-	-

Country	Area	Site Name	pH	Alk	Ca+ Mg	SO4	NO3	WS area	RT
Canada	Nova Scotia	Beaverskin Lake	5.4	4	47	50	1	1	1
		Kejumkujik Lake	5.4	6	57	53	1	20	0.2
		Little Red Lake	4.4	-36	48	43	1	3.8	0.04
		Mount Tom Lake	4.7	-10	45	44	1	-	2
	Quebec	Lac Bonneville	5.1	0	64	73	4	-	2
		Lac Josselin	5.7	13	76	69	3	1	0.7
		Lac MacLeod	5.6	7	73	60	1	0.9	0.7
		Lac Veilleux	5.9	18	69	56	3	0.5	0.2
		Laflamme Lake	6.4	100	158	73	7	0.7	0.3
	Ontario	Batchawana Lake	5.9	41	166	112	19	0.9	0.3
		Little Turkey Lake	6.7	125	266	123	27	4.9	0.3
		Turkey Lake	6.7	174	318	125	24	8	0.9
		Wishart Lake	6.5	84	226	116	31	3.4	0.2
USA	Maine	Little Long Pond	5.5	11	69	76	1	238	1.2
		Tilden Pond	6.0	48	91	59	1	67	1.5
	New York Adirondack	Arbutus Lake	6.6	80	229	144	12	342	-
		Constable Lake	4.9	-9	126	140	30	966	-
		Dart Lake	5.3	-2	128	129	28	1476	-
		Heart Lake	6.3	40	149	107	3	60	-
		Rondaxe Lake	5.9	26	156	125	27	13937	-
		Moss Lake	6.4	70	207	131	26	1248	-
		Otter Lake	5.1	-9	119	136	21	321	-
	New York Catskill Mt.	East Branch Neversink	5.2	-2	124	119	27	3457	-
		High Falls Brook	6.4	127	257	133	30	715	-
	Michigan	Andrus Lake	5.7	16	114	96	3	186	0.6
		Buckeye Lake	7.5	180	233	69	4	328	0.9
		Johnson Lake	4.7	-17	98	124	5	137	0.4
	Minnesota	Cruiser Lake	6.8	119	162	53	4	119	9.0
	Wisconsin	Luna Lake	6.7	24	98	86	1	80	2.9
		Sand Lake	5.5	7	115	129	4	39	1.7
		Nichols Lake	6.0	30	95	63	6	91	0.9
	Colorado	White Dome Lake	6.1	5	54	40	8	40	-
		Seven Lakes	7.2	44	55	12	-	67	-
Summit Lake		6.8	60	76	22	5	11	-	
Upper Sunlight Lake		7.1	32	106	56	2	75	-	

6.2.2 Trend Results

Trend results on a site by site basis are presented as z-scores, where the magnitude of the index represents the strength of the trend, or how certain we are that there is a non-zero trend (Table 6.2.2.).

A large value indicates the certainty with which we can say that the trend is significantly different from zero. This should not be confused with the slope of the trend, which is not presented. Negative scores indicate decreasing trends, while positive scores indicate increasing trends. Absolute values of index magnitudes greater than 1.645 represent significant trends at the 0.10 level, these have been highlighted in bold print.

Decreases in surface water SO_4^{2-} dominated at the US. sites, although these were not accompanied by any specific trend patterns in other variables. At the Canadian sites, decreases in SO_4^{2-} were common, and were accompanied by increases in pH, which may result from decreased acid input with decreased SO_4^{2-} . Concurrent increases in alkalinity occurred in some of these sites. Decreases in surface water SO_4^{2-} concentrations were common in the European sites, among both river and lake sites. Another dominant trend in the European sites was decreasing Ca^{2+} concentration at many of the sites in the Netherlands, Norway, and Sweden. Concomitant trends in alkalinity were uncommon, except in the Netherlands, where increasing alkalinity was observed along with increases in NH_4^+ . Decreases in NO_3^- concentrations were common across the European sites, although both increases and decreases were observed in the Norwegian and Swedish sites, and only increases were observed at the sites in the Netherlands.

Trends in precipitation volume were significant only at a few sites, one Canadian deposition site, at one of the Swedish deposition monitoring sites, and at one of the US. sites (Table 6.2.3). However, although most trends in precipitation volume were not significant, trends at all but two sites were in a decreasing direction. For the European and Canadian sites, precipitation volumes were obtained from the same sites as the deposition chemistry. For the US. sites precipitation volumes were obtained at meteorological sites that were located closer to the surface water monitoring sites than were the deposition sites.

Trend tests were run on both precipitation concentration and deposition loads in precipitation for the available monitoring sites. Significant decreases in SO_4^{2-} deposition were noted only at two sites in Sweden, one in Germany and two in the U.S. Other dominant trends were increases in Ca^{2+} deposition at some Canadian and German sites, and decreases in Ca^{2+} deposition at some Dutch sites, Swedish sites, and U.S. sites. Increases in NO_3^- deposition were common in Canada, Germany, and Norway. Increases in NH_4^+ deposition were seen in Canada, The Netherlands, and in Norway, while a decrease was observed in Finland. Hydrogen ion deposition increased at the Finnish site, and decreased at one of the German sites, the Dutch site, and at one US. site.

Table 6.2.2 Z-Scores for trends in pH, alkalinity, Ca^{2+} , SO_4^{2-} , NO_3^- and NH_4^+ . Negative scores indicate decreasing trends, while positive scores indicate increasing trends. Scores greater than 1.645 or less than -1.645 (bold) are significantly different from 0 at the 0.10 significance level. (Scores >1.96 or <-1.96 are significant at the 0.05 level)

Country	Area	Site Name	ZPH	ZALK	ZCA	ZSO4	ZNO3	ZNH4
Norway	Buskerud	Langtjern	-1.62	.	-4.74	-8.59	-1.20	-0.4
	Oppland	Aurdøla	2.73	3.14	-1.05	-1.83	-1.97	.
	Telemark	Storgama	-1.94	.	-4.30	-5.52	-2.15	-0.4
	Aust Agder	Birkenes	-0.43	.	-2.35	-5.95	3.41	0.1
		Tovdalselva	1.42	0.84	-3.37	-6.03	2.85	.
	Rogaland	Vikedalselva	-0.93	-0.12	-3.04	-3.52	3.01	.
	Sogn og Fjordane	Gaular	-1.20	1.47	-2.27	-3.98	4.36	.
Nausta		0.89	1.13	-3.28	-6.38	0.19	.	
Trodøla		-4.95	-3.87	-5.33	-3.24	-0.18	.	
Sweden		Alsteraaen,Getebro	3.24	0.00	4.07	3.53	-3.06	-2.6
		Alsteraaen,Stroemsborg	1.19	.	3.83	0.91	-2.01	-0.2
		Fraecksjoen	-0.22	-1.15	-1.59	-0.68	0.00	-0.6
		Aneraasen,Haersvatn	-3.30	.	-5.29	-5.06	3.73	-1.8
		Delangeraaen, Iggersund	-4.83	2.57	-3.52	-1.52	-3.10	1.1
Finland		Hirvilampi	-2.63	.	0.74	-0.86	-0.44	.
		Kivijaervi	1.47	2.60	.	.	0.10	.
		Maekelampi	-1.20	.	0.00	-2.05	1.16	.
		Vuorlampi	0.44	.	-0.36	-0.42	-0.22	.
Denmark		Skaerbaek	-3.06	-0.15	0.56	0.42	-3.06	2.7
FRG	Harz	Alte Riefenbeck	0.53	.	-3.25	-2.60	-2.56	0.0
		Alte Riefenbeck	1.60	.	-1.05	-0.60	-2.18	0.0
		Alte Riefenbeck	1.90	.	-1.75	-1.62	-1.69	0.0
		Varleybach	1.45	.	1.05	-3.22	0.97	-1.7
		Varleybach	0.38	.	-0.91	-2.61	1.33	-1.6
		Grosse Schacht	0.73	.	0.50	-2.82	-2.04	-1.3
	Taunus	Rombach	0.00	.	-0.80	-0.80	.	0.0
	Hunsrueck	Traunbach	3.30	.	1.25	-3.53	0.82	-2.7
The Netherlands		Achterste Goorven A	4.22	4.76	-0.91	-0.28	1.72	1.6
		Achterste Goorven B	5.40	5.46	-2.55	-0.57	0.73	2.2
		Achterste Goorven E	7.13	5.34	-3.74	-4.10	0.69	3.6
		Gerritsfles	0.15	0.45	-3.18	-5.49	1.72	1.4
		Kliplo	-1.26	1.15	-1.35	2.77	1.02	4.7

Table 6.2.2 cont.

Country	Area	Site Name	ZPH	ZALK	ZCA	ZSO4	ZNO3	ZNH4
Canada	Nova Scotia	Beaverskin Lake	0.80	2.37	0.95	1.77	0.50	.
		Kejumkujik Lake	2.40	5.57	-1.64	1.57	0.00	.
		Little Red Lake	0.34	0.42	0.43	0.00	0.00	.
		Mount Tom Lake	0.37	1.54	0.00	-0.12	0.00	.
	Quebec	Lac Bonneville	-2.23	0.99	-3.27	0.10	-0.46	.
		Lac Josselin	0.77	1.35	0.60	0.36	-0.81	.
		Lac MacLeod	-0.39	1.46	-2.11	0.00	0.84	.
		Lac Veilleux	0.12	0.00	-0.89	-0.52	-0.11	.
		Laflamme Lake	3.18	-2.58	1.22	0.73	-4.90	.
	Ontario	Batchawana Lake	2.78	-0.94	-0.18	-3.12	-0.61	.
		Little Turkey Lake	3.07	0.81	2.99	-2.92	2.84	.
		Turkey Lake	3.37	4.15	3.99	-3.54	1.98	.
		Wishart Lake	3.84	1.43	1.51	-5.18	1.45	.
USA	Maine	Little Long Pond	1.86	2.20	0.14	-2.28	.	.
		Tilden Pond	-1.04	2.47	0.65	0.35	.	.
	New York Adirondack	Arbutus Lake	1.68	1.70	2.03	-0.56	2.76	.
		Constable Lake	-1.50	-2.42	0.11	-2.65	2.18	.
		Dart Lake	0.79	-1.65	0.37	-2.88	1.73	.
		Heart Lake	-0.08	-1.43	0.39	-2.08	2.52	.
		Rondaxe Lake	-0.11	-1.81	-0.54	-2.47	1.71	.
		Moss Lake	-0.46	0.87	-0.30	-2.62	1.02	.
		Otter Lake	-0.71	-2.13	-0.44	-2.40	2.40	.
	New York Catskill Mt.	East Branch Neversink	-0.14	0.00	-0.29	-2.21	2.04	.
		High Falls Brook	-1.95	1.56	0.00	-2.73	2.52	.
	Michigan	Andrus Lake	0.50	0.34	0.00	-1.43	.	.
		Buckeye Lake	1.71	1.98	0.96	-2.58	.	.
		Johnson Lake	1.08	0.11	0.88	1.04	.	.
	Minnesota	Cruiser Lake	1.92	-1.24	0.66	-1.92	.	.
	Wisconsin	Luna Lake	1.29	0.63	0.00	-1.53	.	.
		Sand Lake	1.90	0.00	-0.87	-0.57	.	.
		Nichols Lake	1.83	0.87	-0.20	-0.60	.	.
	Colorado	White Dome Lake	-0.32	-0.76	0.66	0.65	.	.
		Seven Lakes	1.83	0.22	0.77	1.20	.	.
Summit Lake		0.00	0.55	0.87	2.12	.	.	
Upper Sunlight Lake		-0.18	0.00	-0.30	-0.30	-0.32	.	

Table 6.2.3 Deposition sites for each of the surface water monitoring sites and Z-scores for the trends. Z-scores > 1.645 or < -1.645 (bold) indicate trends that are significantly different from zero at the 0.10 level. Negative z-scores indicate decreasing trends, positive z-scores indicate increasing trends. (Scores >1.96 and <-1.96 are significant at the 0.05 level).

Country	Area	Site Name	Deposition Site	Distance km	ZH+	ZCa	ZSO4	ZNO3	ZNH4+	Precip Volum
Norway	Buskerud	Langtjern	Gulsvik	0	0.82	0.37	1.06	2.59	3.21	1.46
	Oppland	Aurdøla		19.7						
	Telemark	Storgama	Treungen	0	0.99	0.18	0.48	2.23	0.84	1.11
	Aust Agder	Birkenes	Birkenes	0	0.42	-0.21	0.33	1.23	0.99	1.29
		Tovdalselva		16.92						
	Rogaland	Vikedalselva		<160km						
	Sogn og Fjordane	Gaular		<160km						
	Nausta		<160km							
	Trodøla		<160km							
Sweden		Alsteraaen,Getebro	Norra Kvill	94	-2.33	-4.15	-2.57	-0.93	-0.78	-0.16
		Alsteraaen,Stroemsborg	Aneboda	71	-2.24	-2.66	-3.01	-1.57	-1.56	-2.03
		Fraecksjoen	Svartedalen	5	-1.29	-3.56	-1.14	0.62	0.57	-0.31
		Aneraasen,Haersvatn		-						
		Delangeraaen, Iggersund	Stormyren	20	-1.85	-1.16	-1.39	-0.98	-1.39	0.34
Finland		Hirvilampi	Virolahti	23	2.01	-0.47	-0.75	0.88	-2.54	-1.11
		Kivijaervi		50						
		Mackelampi		27						
		Vuorlampi		28						
Denmark		Skaerbaek	Westerland	153	0.98	0.27	1.19	1.19	1.06	1.17
FRG	Harz	Alte Riefenbeck	Langenbrugge	120	0.21	2.93	1.87	1.76	1.23	0.34
		Alte Riefenbeck		120						
		Alte Riefenbeck		122						
		Varleybach		110						
		Varleybach		110						
		Grosse Schacht		84						
	Taunus Hunsrueck	Rombach	Deuselbach	115	-2.47	-0.11	-0.39	0.00	0.99	1.52
	Traunbach		7							
The Netherlands		Achterste Goorven A	Eibergen	113	-0.20	-3.13	-0.60	0.30	0.20	0.30
		Achterste Goorven B		113						
		Achterste Goorven E		113						
		Gerritsfles		54						
		Kliplo		16						

Table 6.2.3 cont.

Country	Area	Site Name	Deposition Site	Distance km	ZH+	ZCa	ZSO4	ZNO3	ZNH4+	Precip Volum					
Canada	Nova Scotia	Beaverskin Lake	Kejmkujik	12	1.16	-2.19	0.09	1.92	0.65	12.07					
		Kejmkujik Lake		0											
		Little Red Lake		15											
		Mount Tom Lake		7											
	Quebec	Lac Bonneville	Lac Laflamme	23	.	-1.29	.	.	.	-0.28					
		Lac Josselin		43											
		Lac MacLeod		11											
		Lac Veilleeux		37											
		Laflamme Lake		0											
	Ontario	Batchawana Lake	Turkey Lakes	2	-1.22	2.92	0.31	2.41	3.66	0.48					
		Little Turkey Lake		1											
		Turkey Lake		1											
Wishart Lake		0.5													
USA	Maine	Little Long Pond	Acadia	34	-0.22	-1.13	-0.65	-0.45	.	-1.63					
		Tilden Pond	Acadia	34						-1.63					
	New York Adirondack	Arbutus Lake	Huntington	3	0.18	-1.14	0.00	0.16	.	-1.06					
		Constable Lake	Huntington	50						-0.95					
		Dart Lake	Huntington	57						-0.95					
		Heart Lake	White face	26						-1.62	-0.12	-0.90	-1.62	.	-1.06
		Rondaxe Lake	Huntington	62						0.18	-1.14	0.00	0.16	.	-0.95
		Moss Lake	Huntington	56						-0.95					
	Otter Lake	Huntington	93	0.43											
	New York Catskill Mt.	East Branch Neversink	Stillwell Lake	79	0.58	0.33	-0.37	0.00	.	-1.35					
		High Falls Brook	Milford	79						-1.35					
	Michigan	Andrus Lake	Raco	45	-1.21	-1.03	-1.13	-0.35	.	-0.27					
		Buckeye Lake	Raco	80						-0.62					
		Johnson Lake	Raco	25						-0.99					
	Minnesota	Cruiser Lake	Fernberg	118	-1.24	-2.28	-1.97	-1.66	.	-0.32					
	Wisconsin	Luna Lake	Trout Lake	59	-2.09	-2.26	-2.51	-1.52	.	-1.75					
		Sand Lake	Trout Lake	38						-1.10					
		Nichols Lake	Trout Lake	7						-1.10					
	Colorado	White Dome Lake	Molas Pass	13	0.00	0.23	0.23	0.00	.	-1.69					
		Seven Lakes	Buffalo Pass	41	-1.62	-1.26	-0.52	-0.61	.	-0.79					
Summit Lake		Buffalo Pass	1	-1.42											
Upper Sunlight Lake		Molas Pass	17	0.00	0.23	0.23	0.00	.	0.59						

6.2.3 Correlations Between Trends and Other Factors

In order to examine some of the trends observed in the context of other events, correlations between the surface water trend statistics and other characteristics that may help to explain or to describe the observed trends are presented in Figure 6.2.1, and in Table 6.2.4. Only results for the correlations that were significant at the 0.05 level are included in the tables. Data from North America and Europe have been included in separate analyses. In evaluating these results, it should be kept in mind that correlations are descriptive statistics, and although they may be suggestive, they are not proof of cause and effect relationships.

Very different patterns for SO_4^{2-} trends were observed for the North American and European sites. Trends in SO_4^{2-} appear to be related to mean base cation concentrations, although in opposite ways on either side of the Atlantic. In North America, decreases in surface water SO_4^{2-} occur at sites with higher $\text{Ca}^{2+} + \text{Mg}^{2+}$ (Figure 6.2.1, Table 6.2.4), while in Europe the decreasing SO_4^{2-} concentrations are seen at lower $\text{Ca}^{2+} + \text{Mg}^{2+}$ sites (Figure 6.2.1, Table 6.2.4). It is unclear what drives these SO_4^{2-} trends; deposition may have decreased more in the regions where base cation concentrations are low in Europe and high in North America. Also, other mechanisms may be involved. For instance, the North American sites with increasing SO_4^{2-} concentrations are sites that either have watershed sources for SO_4^{2-} , found in Colorado (Turk et al, 1993), or are seepage sites in the Upper Midwest (Michigan and Wisconsin), which are influenced by recent changes in hydrology (Webster et al, 1993), and are thus not related to deposition inputs.

Positive correlations were found between trends in alkalinity and trends in surface water pH in the European data (Table 6.2.4). Trends in both surface water alkalinity and pH were also positively correlated with mean surface water SO_4^{2-} concentrations. In North America, no correlations were observed between trends in pH and trends in alkalinity. However, positive correlations were observed between trends in pH and trends in surface water Ca^{2+} concentrations. Alkalinity and base cations were strongly correlated. Neither of the trends in pH or base cations were correlated with mean SO_4^{2-} , but both were positively correlated with mean $\text{Ca}^{2+} + \text{Mg}^{2+}$ again suggesting that increasing trends are more common in less dilute water.

Trends in surface water SO_4^{2-} were positively correlated with surface water trends in Ca^{2+} (Table 6.2.4) for the European sites. Trends in both surface water alkalinity and Ca^{2+} were negatively correlated with trends in precipitation. These results are similar to those found for the 91 sites included in the US Monitoring Program (Newell, 1993). A plausible explanation for this is that the major ions, Ca^{2+} and SO_4^{2-} , become concentrated with the decreasing precipitation that causes drought conditions (Webster et al. 1990). Alternatively, these ions may become diluted with high levels of precipitation; both these situations can result in the observed correlations.

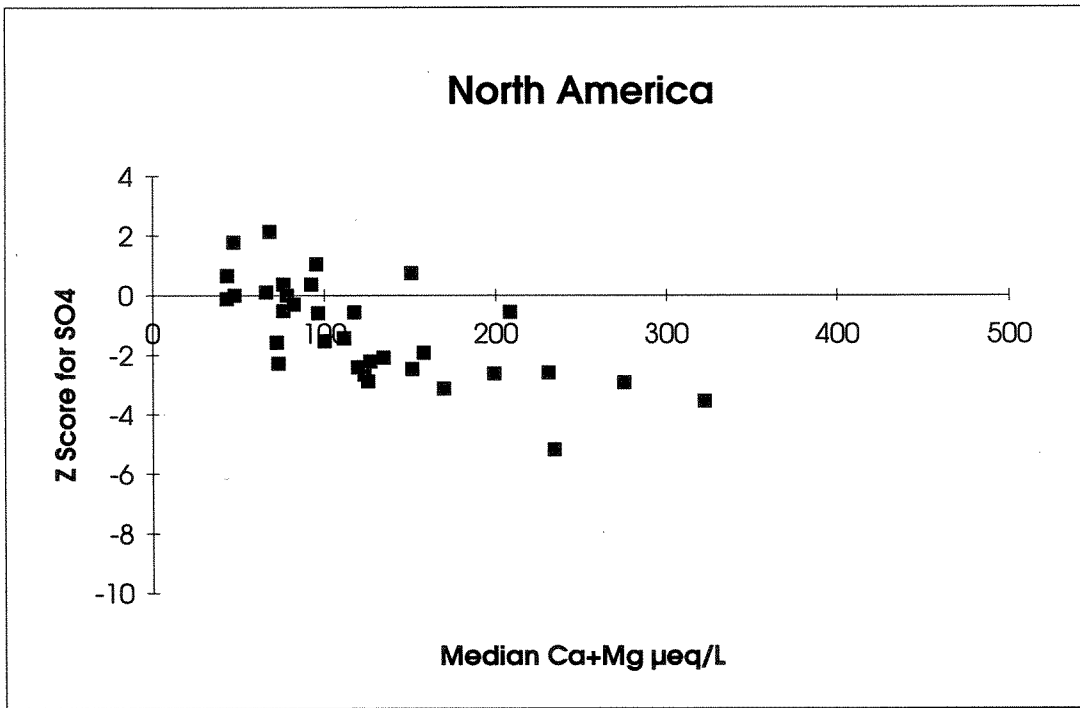
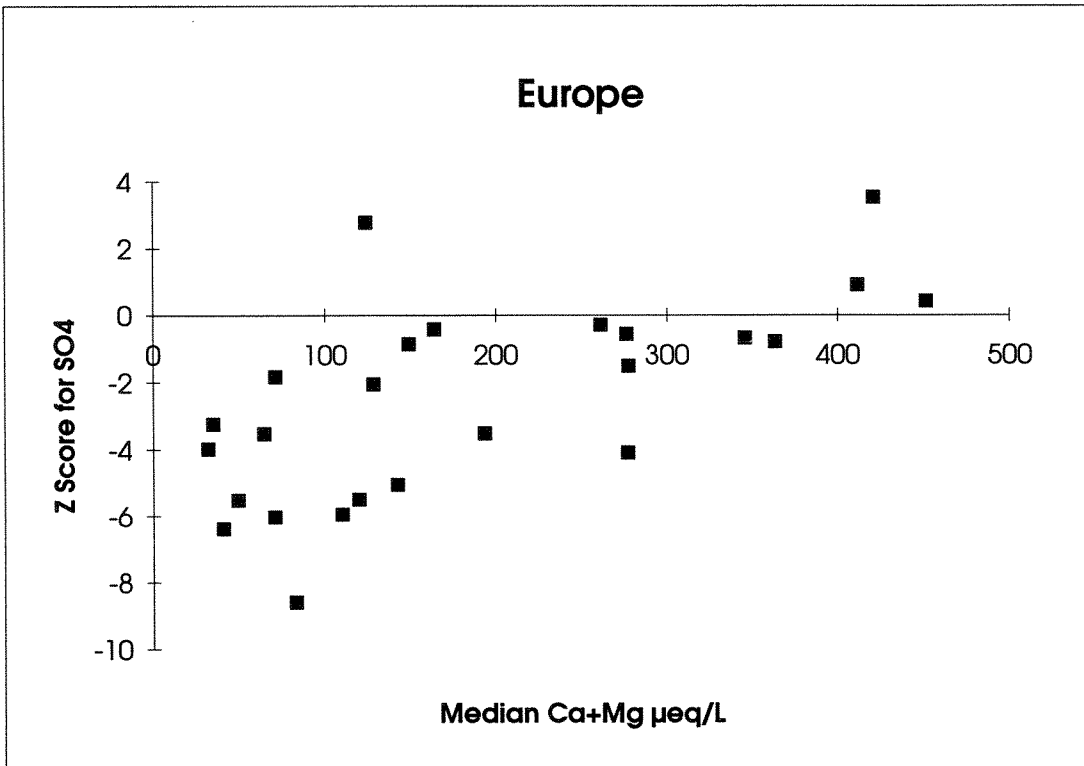


Figure 6.2.1. A plot of Z-scores for trend in surface water SO_4^{2-} in North America and Europe versus mean $\text{Ca}^{2+} + \text{Mg}^{2+}$ concentration.

Table 6.2.4 Correlations between surface water trends and between surface water trends and mean chemistry and trends in precipitation. (a) Europe, n=32, (b) North America n=35. Significance levels for the correlations are shown in parentheses. Only correlations with $p < 0.05$ are presented.

Europe

Correlation between ⇒ and ↓		Trend in surface water:				
		Alk	pH	SO ₄ ²⁻	Ca ²⁺	NO ₃ ⁻
Trend in surface water:	pH	0.723 (0.002)				
	Ca ²⁺			0.668 (0.0001)		
	NO ₃ ⁻			-0.401 (0.028)		
Mean concentration in surface water:	pH					-0.495 (0.005)
	SO ₄ ²⁻	0.619 (0.011)	0.548 (0.001)			
	Ca ²⁺ +Mg ²⁺			0.509* (0.016)		-0.357 (0.049)
Trend in Precipitation:	Volume			-0.438 (0.022)	-0.393 (0.042)	
	Concentration Ca ²⁺					-0.390 (0.044)
	Deposition NO ₃ ⁻			-0.478 (0.012)	-0.387 (0.046)	

* for sites with mean Ca+Mg < 400 µeq/L

North America

Correlation between ⇒ and ↓		Trends in surface water:				
		Alk	pH	SO ₄ ²⁻	Ca ²⁺	NO ₃ ⁻
Trend in surface water:	pH				0.569 (0.0004)	
	NO ₃ ⁻			-0.517 (0.011)		
Mean concentration in surface water:	Alk		0.558 (0.001)	-0.400 (0.019)	0.640 (0.0001)	
	pH		0.676 (0.011)	-0.593 (0.033)	0.625 (0.022)	
	SO ₄ ²⁻			-0.647 (0.0001)		0.496 (0.016)
	Ca ²⁺ +Mg ²⁺		0.523 (0.002)	-0.703 (0.0001)	0.604 (0.0002)	
	NO ₃ ⁻		0.711 (0.006)	-0.873 (0.001)	0.660 (0.014)	
Trend in Precipitation:	Volume	0.342 (0.048)				
	Concentration H ⁺		-0.616 (0.0004)	0.585 (0.001)	-0.403 (0.030)	
	Concentration SO ₄ ²⁻				0.389 (0.023)	
	Deposition SO ₄ ²⁻					0.449 (0.031)
	Deposition NO ₃ ⁻	0.380 (0.042)				

No correlations occurred between trends in SO_4^- and trends in Ca^{2+} in North America. However, correlations between trends in surface water Ca^{2+} and mean surface water alkalinity, pH, $\text{Ca}^{2+} + \text{Mg}^{2+}$, and NO_3^- occurred. Thus increases in Ca^{2+} occurred at sites with higher mean concentrations, while decreases in Ca occurred at sites with lower mean concentrations. Trends in surface water Ca^{2+} also increased with increasing trends in surface water pH. While there is no obvious explanation for the observed concurrent trends in Ca^{2+} and pH, this may also occur in drought situations, where base cations may become more concentrated, or to changes in hydraulic pathways resulting from different climatic conditions. Thus, the data for Europe and North America suggest two different trend patterns that may result from drought conditions.

In North America, surface water trends in pH were positively correlated with mean alkalinity, pH, Ca^{2+} , and NO_3^- , but not with mean SO_4^{2-} concentration or with trends in $\text{Ca}^{2+} + \text{Mg}^{2+}$, as seen in the European data. North American SO_4^{2-} trends correlated negatively with mean alkalinity, pH, SO_4^{2-} , $\text{Ca}^{2+} + \text{Mg}^{2+}$ and NO_3^- concentrations. Thus it appears that increases in SO_4^{2-} occurred at the more acidic sites, while decreases in surface water SO_4^{2-} were observed at the circum-neutral sites. Few of the surface water trends in SO_4^{2-} were increasing trends, and those sites with the highest alkalinity and $\text{Ca}^{2+} + \text{Mg}^{2+}$ exhibited the strongest SO_4^{2-} decreases, causing the observed correlations. These results suggest that increasing pH trends are likely to occur at sites where SO_4^{2-} trends are decreasing.

The sites in the ICP are monitored for surface water chemistry to assess changes that may occur as a result of changes in atmospheric deposition. Trends in surface water SO_4^{2-} , pH or NO_3^- however were not correlated with trends in deposition or precipitation concentrations of SO_4^{2-} , NO_3^- or hydrogen ion inputs. Probably the time periods for both surface water monitoring and deposition monitoring are too short to reveal the response. Also the changes in deposition may have occurred more prior to the surface water, and monitoring period. Surface waters may still be changing in response to the earlier decreases.

6.2.4 Cluster Results

Clustering analysis has been applied to group the sites according to their chemical characteristics, and according to their trend results. This two analyses give the ability to place sites into groups that exhibit similar trends, using the information about their chemical characteristics. Then one might be able to predict future trends for a given site, if something is known about its' current chemistry. In addition, grouping sites according to the trends that they exhibit, and subsequently comparing the site characteristics within groupings possessing similar trends, may allow us to understand more about the processes involved in producing the observed trends.

Clustering analysis on the mean chemistry ($\text{Ca}^{2+} + \text{Mg}^{2+}$, SO_4^{2-} , and pH) of European and North American sites resulted in the description of three chemical clusters. Concentrations of both $\text{Ca}^{2+} + \text{Mg}^{2+}$ and SO_4^{2-} defined the clusters. The ranges of these two variables differ greatly among the three clusters, reflecting the differences in base cation concentrations among the sites included in the Programme. Two of the clusters have significant mean z-score values for decreasing trends in SO_4^{2-} . Ranges of the z-scores for trends in alkalinity, $\text{Ca}^{2+} + \text{Mg}^{2+}$, and pH within the clusters are generally near zero. These clusters do not well define trend patterns. In addition, several trend patterns occur (table 6.2.6), while only a few clusters based on chemical characteristics were distinguished. It is likely that creating clusters that define trend patterns with chemical characteristics alone requires many more variables than we have available. Although more variables are included in the dataset, many have missing values, precluding the opportunity to explore clustering analysis on more variables for a large number of ICP sites.

Similar analysis on the trend results for SO_4^{2-} , Ca^{2+} , pH, and NO_3^- resulted in eight clusters (Table 6.2.6). Only the average trend result is shown; not every site within the cluster will have significant trends for all the variables as described in the cluster. Also, for cluster 8, which on the average has no significant trends, several of the sites within the cluster exhibit significant trends for some of the variables.

Decreases in surface water SO_4^{2-} concentrations are prevalent across Europe and North America are common trend characteristics in 6 of the 8 clusters. Differences between clusters result from trends in other variables, such as Ca^{2+} , pH and NO_3^- . Decreases in Ca^{2+} accompany the decreases in SO_4^{2-} in several clusters. These may result from two things; one is a decreased base cation loading that results from increased pH in runoff water, the other is decreased base cation input resulting from decreased concentrations in deposition.

Clusters also differ in the direction and significance of surface water NO_3^- concentrations. The soft, acid sensitive waters of southern Norway split into two clusters characterised either by increasing or decreasing surface water concentrations of NO_3^- . The causes of the increasing NO_3^- concentrations are not always clear. In these clusters, the trends in surface water NO_3^- do not correlate well with trends in NO_3^- deposition, although in some regions significant increases in NO_3^- deposition are observed.

Table 6.2.6 Results from cluster analysis. Trend characteristics are indicated by + for increasing and ÷ for decreasing and 0 for no trend.

Cluster	n	Trend Characteristics	Site Location
Cluster 1	7	÷ Ca ²⁺ ÷ SO ₄ ²⁻ + NO ₃ ⁻	5 Norwegian Rivers 1 Swedish River 1 Dutch Lake
Cluster 2	4	÷ Ca ²⁺ ÷ SO ₄ ²⁻ ÷ pH ÷ NO ₃ ⁻	3 Norwegian Rivers 1 Swedish River
Cluster 3	8	÷ SO ₄ ²⁻ + NO ₃ ⁻	5 Adirondack Lakes 1 Catskill Stream 1 Finnish Lake 1 German River
Cluster 4	16	÷ SO ₄ ²⁻	6 Upper Midwest Lakes 3 German Rivers (1 with 3 sites) 2 Canadian Lakes 1 Adirondack Lake 1 Maine Lake
Cluster 5	4	+ Alk + Ca ²⁺ ÷ SO ₄ ²⁻ ÷ pH ÷ NO ₃ ⁻	3 Ontario Lakes 1 German River
Cluster 6	3	+ Alk ÷ Ca ²⁺ ÷ SO ₄ ²⁻ + pH	3 sites in 1 Dutch Lake
Cluster 7	3	+ Ca ²⁺ + SO ₄ ²⁻ + pH - NO ₃ ⁻	1 Lake in Quebec 2 sites on 1 Swedish River
Cluster 8	20	0	7 US. Lakes 7 Canadian Lakes 2 Finnish Lakes 1 Dutch Lake 1 German River 1 Danish River 1 Swedish River

The cluster results group sites that exhibit similar trends. All of the Norwegian sites fall into cluster 1 and 2. Both these clusters are characterised by increasing NO_3^- , decreasing Ca^{2+} and decreasing SO_4^{2-} and in addition in cluster 2 has decreasing pH. A representative of this type of sites is Tovdalselva in southern Norway (Figure 6.2.2). Here we can see that SO_4^{2-} is decreasing while pH is constant. Cluster 3 is characterised by decrease in SO_4^{2-} and increase in NO_3^- . This cluster is not unlike cluster 1 and 2 and differs only from cluster 4 by the increase in NO_3^- . Most of the North American sites fall into cluster 3 and 4. An example of cluster 3 is the site Constable Lake in the Adirondacks, New York (Figure 6.2.3). Here NO_3^- is increasing while SO_4^{2-} is decreasing. Cluster 4 is characterised by increasing alkalinity, Ca^{2+} and pH and decreasing SO_4^{2-} . An example of this cluster is one of the Ontario lakes, Turkey Lake (Figure 6.2.4). Here SO_4^{2-} is decreasing and pH and NO_3^- are increasing. All these three sites are situated in areas where the deposition of SO_4^{2-} has decreased and deposition of NO_3^- has increased the last decade, and this is reflected in the surface water chemistry. The last example is from a Dutch pond falling in cluster 6, which is characterised by increasing pH, alkalinity and NO_3^- and decreasing SO_4^{2-} . This site is very different from the others and the trends in pH and SO_4^{2-} (Figure 6.2.5) are not related to changes in deposition, but to recovery after a severe drought in 1977 (van Dam, 1988)

However, further understanding of the observed trends may be apparent with more information about trends in specific systems. Also additional information regarding physical, chemical and hydrologic characteristics may improve both the clustering results and interpretation, and our understanding of the systems in general.

NORWAY
Station OVELV, Tovdalselva (7 1) 1980-1991

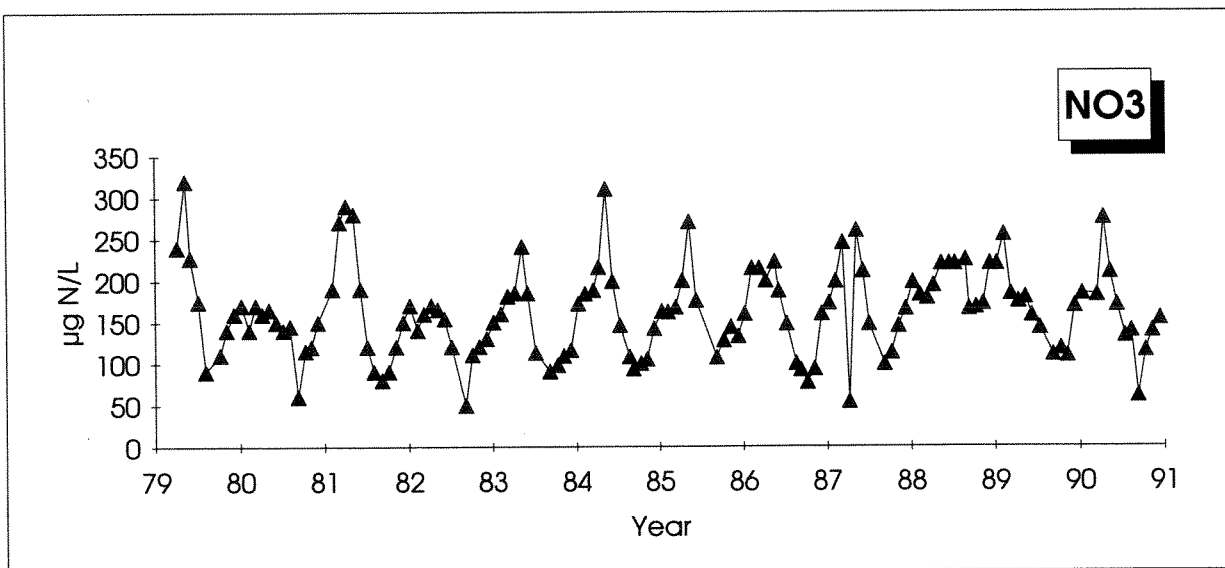
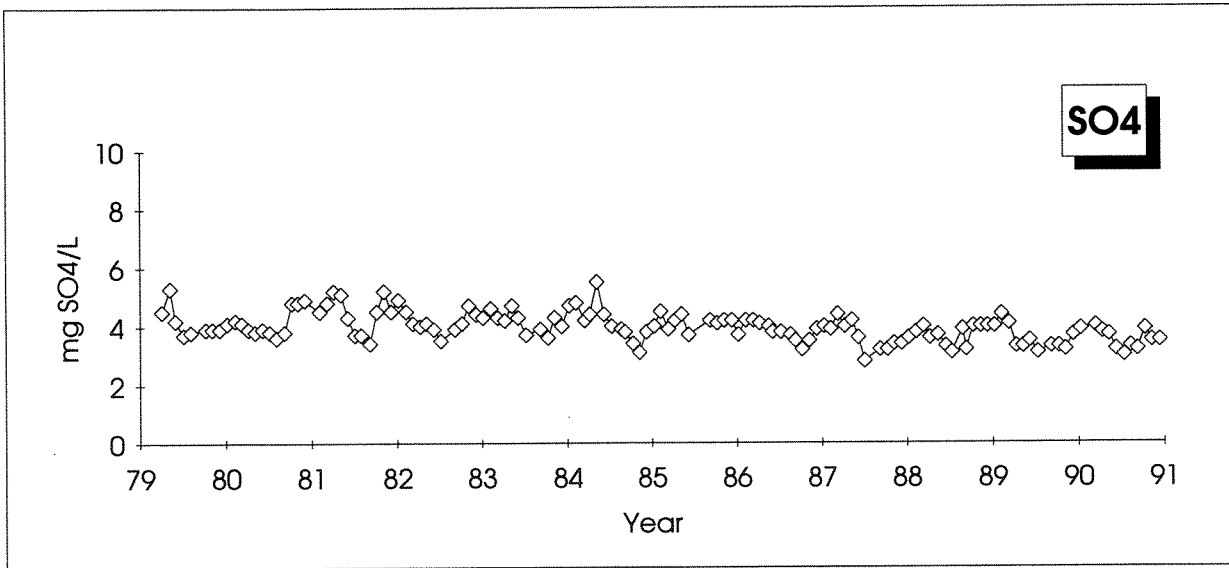
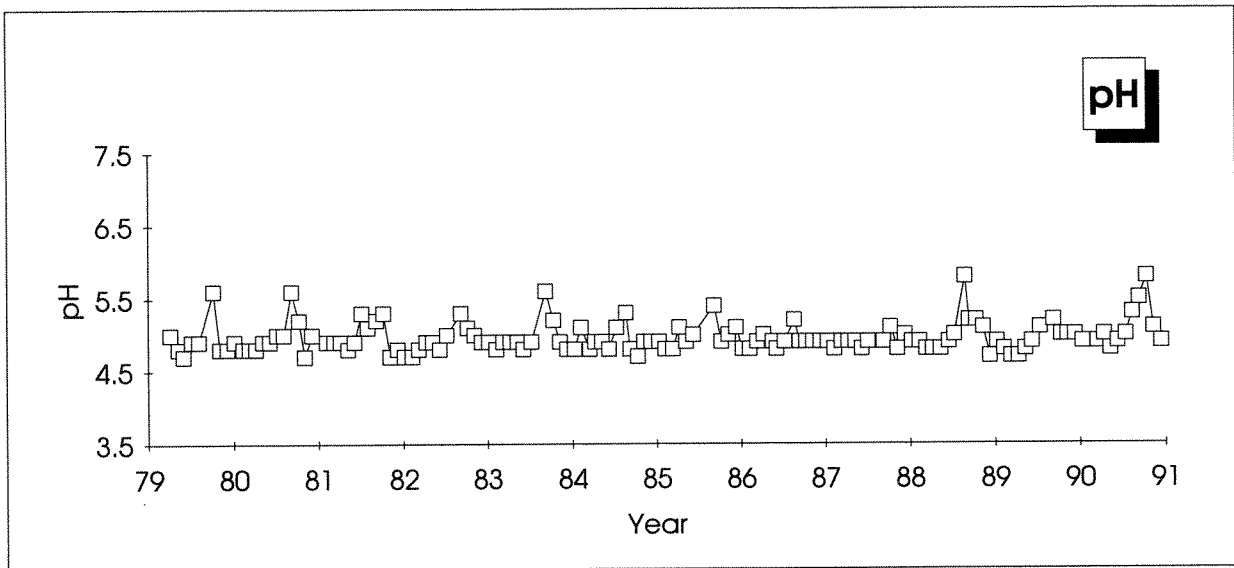


Figure 6.2.2 Long-term trends of pH, SO₄²⁻ and NO₃⁻ in Tovdalselva, Norway

USA
New York, Adirondack (Constable Lake 1A1017) 1981-1991

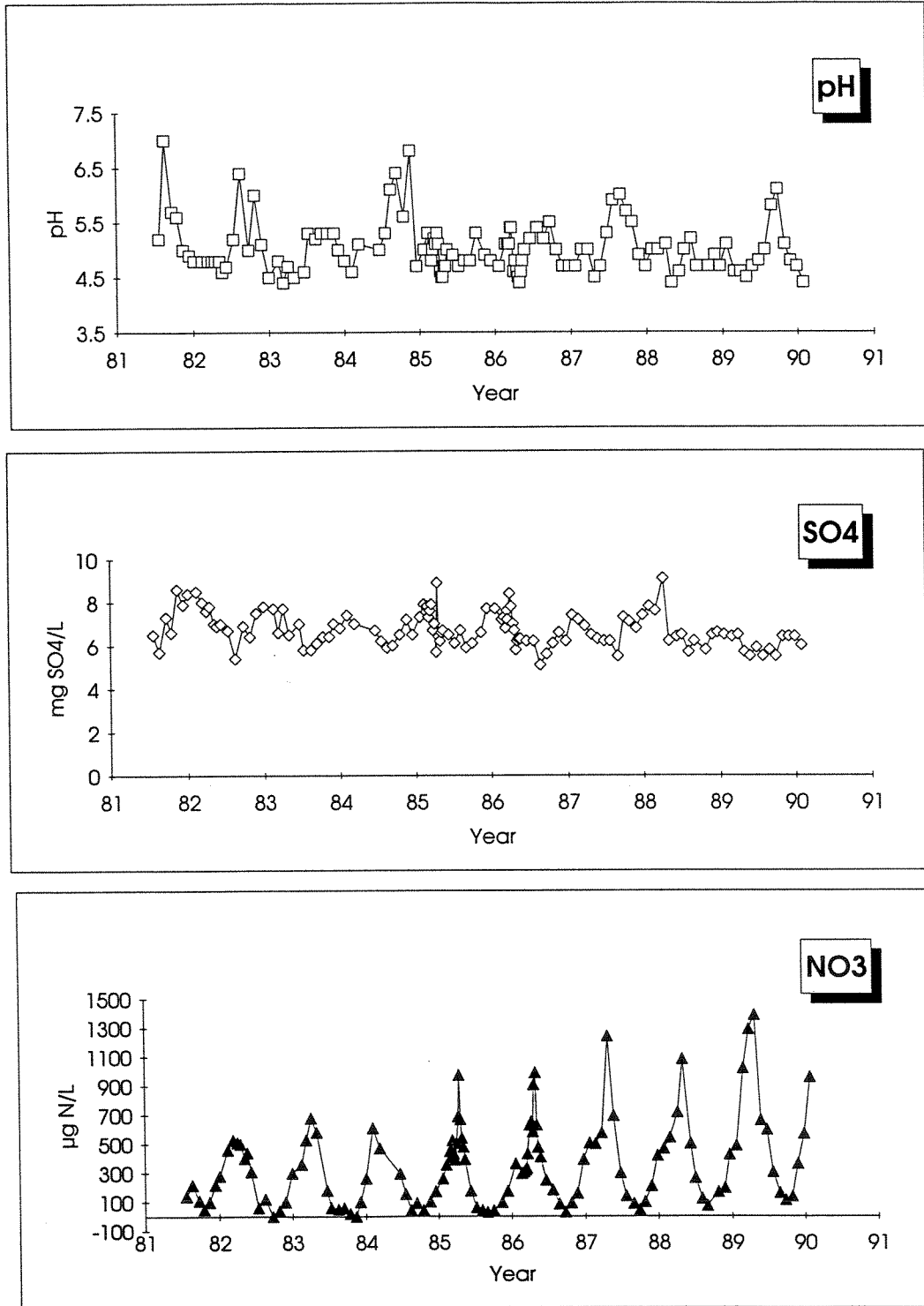


Figure 6.2.3 Long-term trends of pH, SO_4^{2-} and NO_3^- in Constable Lake, Adirondack, New York, US

CANADA
Ontario, Turkey Lake (C04) 1979-1991

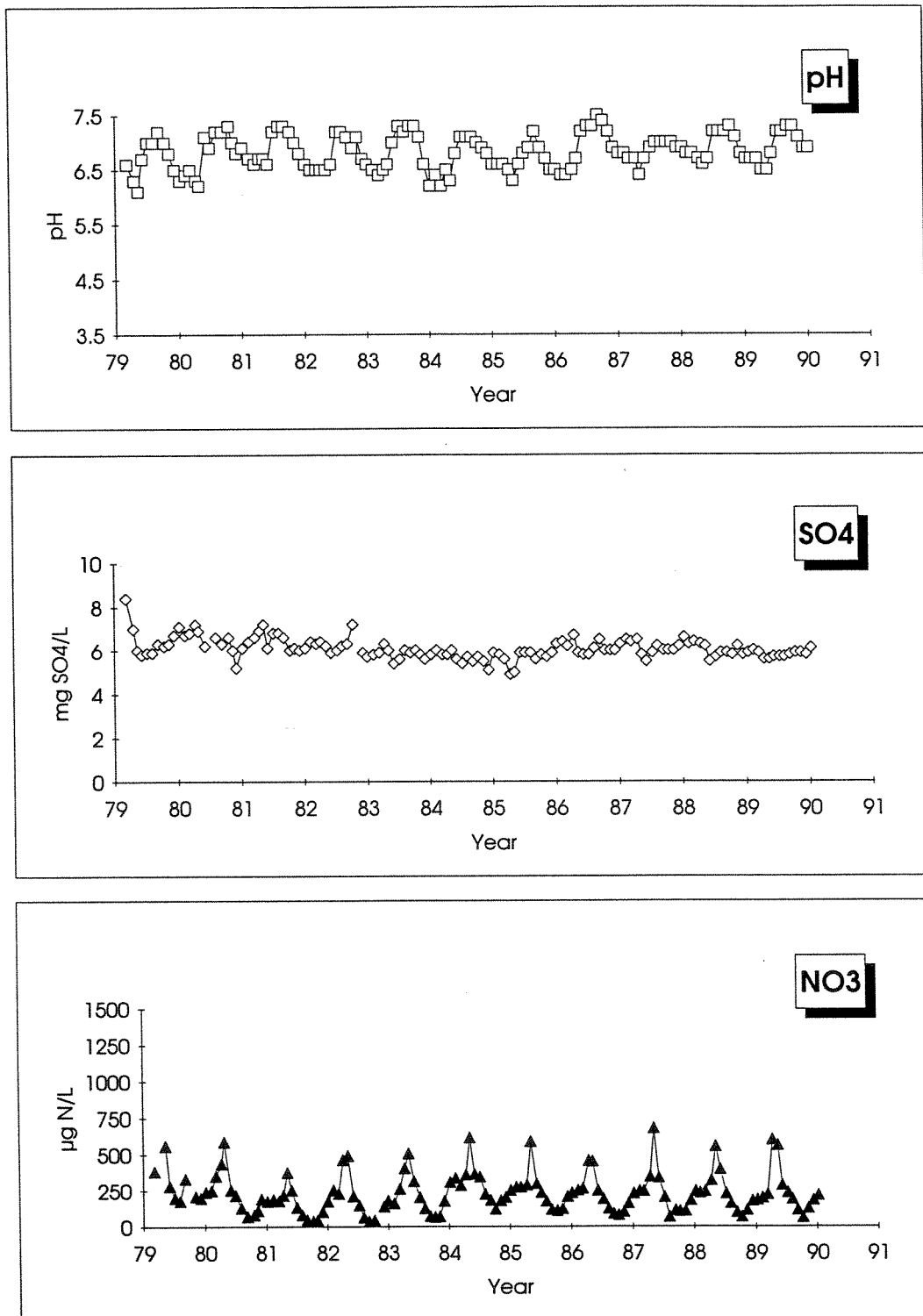


Figure 6.2.4 Long-term trends of pH, SO_4^{2-} and NO_3^- in Turkey Lake, Ontario, Canada.

THE NETHERLANDS
Station Achterste Goorven E (AGE) 1978-1991

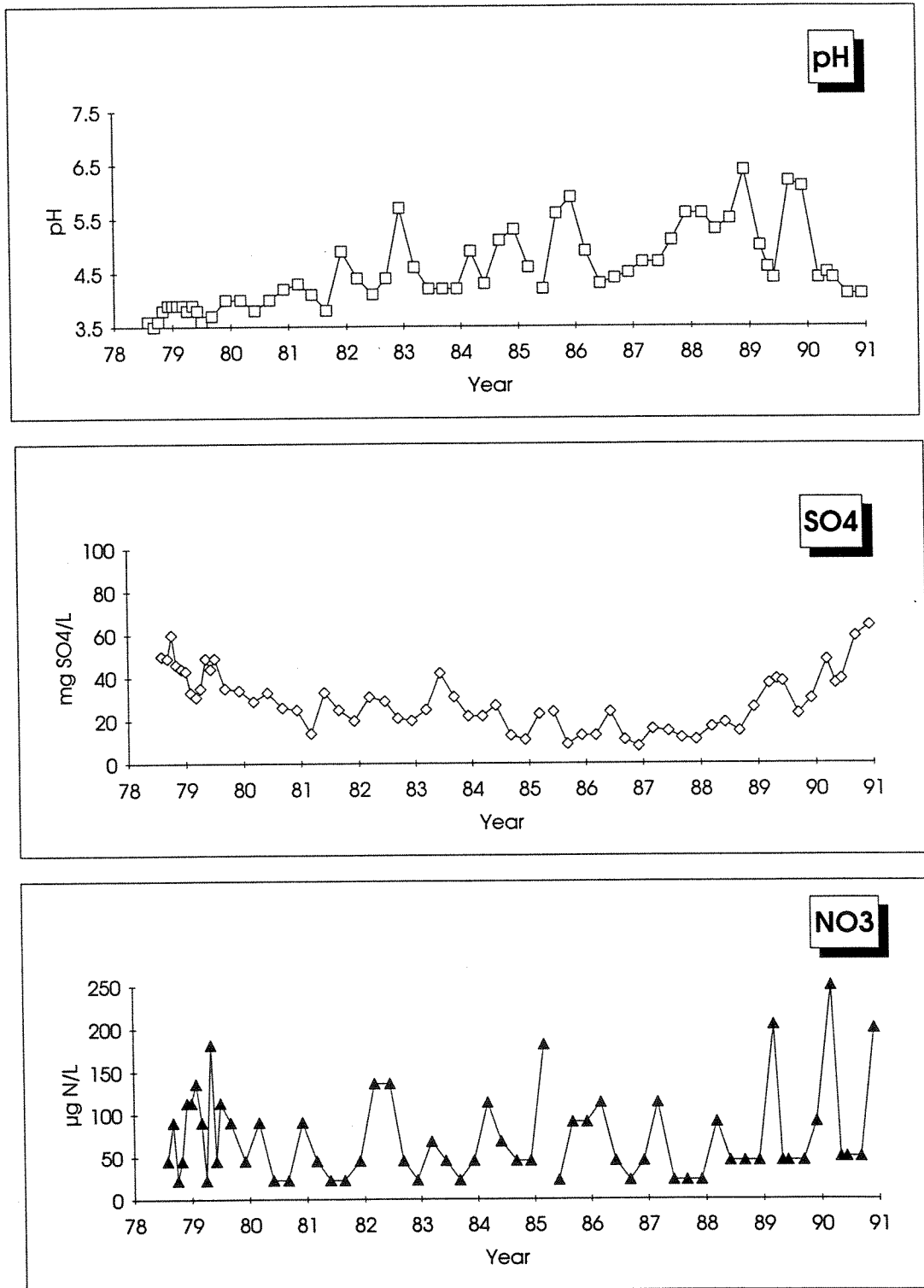


Figure 6.2.5 Long-term trends of pH, SO₄²⁻ and NO₃⁻ in Achterste Goorven, The Netherlands.

6.3 Intercalibration results

6.3.1 Report from chemical intercalibration exercises 1991-1993

The "Manual for Chemical and Biological Monitoring" (1987), states that a between-laboratory programme is necessary in a multilaboratory programme, to assure clear identification and control of the bias between analyses carried out by individual participants of the programme. Therefore, seven intercalibration exercises have been arranged by the programme centre during the years 1987 - 1993. It is reasonable to suppose that the within-laboratory variation is of minor importance, the Youden technique using sample pairs has been applied for the exercises, giving information about the between-laboratory deviations, which is most important. The following analytical variables have been tested in the last three exercises:

- 9105 - pH, conductivity, alkalinity, nitrate + nitrite, chloride, sulphate, calcium, magnesium, sodium, potassium, and total organic carbon
- 9206 - pH, conductivity, alkalinity, nitrate + nitrite, chloride, sulphate, calcium, magnesium, sodium, potassium, aluminium, and dissolved organic carbon
- 9307 - pH, conductivity, alkalinity, nitrate + nitrite, chloride, sulphate, calcium, magnesium, sodium, potassium, total aluminium, reactive and non-labile aluminium, dissolved organic carbon, and chemical oxygen demand

For these three last exercises 2, 4 and 2 samples, respectively, have been prepared from natural waters. The preparation included no treatment like adjustment of sample concentrations by addition of stoichiometric salts or mixing of natural waters from different locations. Then the sample batches were stored at the laboratory for equilibration, and filtered to remove the effect of particulate matter. The evaluation of the results of the three intercalibrations are compared in table 6.5.1.

The target accuracies outlined in the Manual for Chemical and Biological monitoring (1987) have been used as acceptance limits for the results of the intercalibration studies. These correspond to the analytical detection limit or the true value \pm a percentage (Table 6.5.1), whichever is greater. Acceptable results were obtained for conductivity, the main constituents (SO_4^{2-} , Cl^- , NO_3^- , Ca^{2+} , Mg^{2+} , Na^+ , K^+), and organic matter (dissolved organic carbon). Even with a narrower limit of $\pm 10\%$, most of the results were acceptable.

Generally, a low percentage of acceptable results was obtained for pH in samples of

circumneutrality, while 70 % of the results in the rather acid sample pair CD of intercalibration 9206 were acceptable. The "in situ" method is applied by far most of the participating laboratories. The non-equilibrium in nearly neutral samples with respect to the CO₂ pressure, leads to a rather great spread of the results from the participants, the results depending on storage and handling conditions.

Regarding the determination of alkalinity, and Al fractions, too many laboratories have reported results lying outside the acceptance limit. Some samples contained low concentrations of these analytical variables, and the low percentage of acceptable results for these variables are partially dependent on this fact. In addition, some laboratories are using other methods than those recommended in the manual, which also may lead to deviating results.

For the analytical variables NO₃⁻ + NO₂⁻, Cl⁻, SO₄²⁻, Ca²⁺, Mg²⁺, Na⁺, and K⁺, acceptable results have been obtained from nearly all laboratories, provided that the sample concentration is far away from the determination limit of the method used. The results for organic matter are more varying.

During the last three years an increasing number of laboratories have been participating in the intercalibration exercises, representing an increasing number of countries, especially in the eastern part of Europe. In some cases, the low concentration in some of the samples used in these intercalibration exercises, may be outside the routine analytical range for methods routinely used for determination in more polluted samples.

Table 6.3.1 Summary of the results from the three last intercalibration exercises carried out by the International Programme of Assessment and Monitoring of Acidification of Rivers and Lakes, during the period 1991 - 1993.

Analytical variable	Intercal. no.	Sample pair	Accept. limit	No. of results	No. of accept.	% accept.
pH	9105	AB	0.1	21	7	35
	9206	AB	0.1	23	7	30
		CD	0.1	23	16	70
	9307	AB	0.1	25	13	52
Conductivity	9105	AB	20 %	20	19	95
	9206	AB	20 %	22	21	95
		CD	20 %	22	19	86
	9307	AB	20 %	24	22	82
Alkalinity	9105	AB	20 %	20	18	90
	9206	AB	20 %	20	11	55
		CD	20 %	23	16	70
Nitrate + nitrite	9105	AB	20 %	20	19	95
	9206	AB	20 %	22	13	59
		CD	20 %	22	20	91
	9307	AB	20 %	24	17	71
Chloride	9105	AB	20 %	20	18	90
	9206	AB	20 %	22	20	91
		CD	20 %	22	18	82
	9307	AB	20 %	24	21	88
Sulfate	9105	AB	20 %	20	19	95
	9206	AB	20 %	22	20	91
		CD	20 %	22	18	82
	9307	AB	20 %	24	22	92
Calcium	9105	AB	20 %	20	19	95
	9206	AB	20 %	22	20	91
		CD	20 %	22	18	82
	9307	AB	20 %	23	19	79
Magnesium	9105	AB	20 %	20	19	95
	9206	AB	20 %	22	21	95
		CD	20 %	22	19	86
	9307	AB	20 %	23	19	79
Sodium	9105	AB	20 %	20	20	100
	9206	AB	20 %	21	18	86
		CD	20 %	21	18	86
	9307	AB	20 %	23	19	79
Potassium	9105	AB	20 %	20	19	95
	9206	AB	20 %	20	18	90
		CD	20 %	20	10	50
	9307	AB	20 %	23	20	87
Aluminium, total	9206	AB	20 %	17	7	41
		CD	20 %	17	11	65
Aluminium, reactive	9307	AB	20 %	11	8	73
	9307	AB	20 %	10	5	50
Total organic carbon	9105	AB	20 %	12	10	83
Dissolved organic carbon	9206	AB	20 %	16	11	69
		CD	20 %	16	13	81
	9307	AB	20 %	17	13	76
Chem. oxygen demand	9307	AB	20 %	7	3	43

6.3.2 Results from biological intercalibration

Nine laboratories from seven countries participated in the first intercalibration of invertebrates in 1992. However, some additional laboratories wanted to take part in the test, but were not able to send material within the scheduled time. Two of these laboratories attended in 1993. All laboratories identified a high portion (>89%) of the total number of the genus/species in the samples. In 1992 the average failure rate was 4 % with mean and maximum of 0 and 11 %, respectively. In 1993 the corresponding values were 3.4 % ,1 % and 11 %, a result practically identical with the previous year.

The results for the specific groups varied more. The laboratories identified mayflies very well, but short coming determinations were in some cases relatively high. This can be seen by looking at laboratories with no faults, but reduced number (< 100 %) of correct identified species.

The errors made in determination of stoneflies were in contrast to the mayflies, only at species level. At some laboratories the determination was carried out only to genus level in cases where species determination was expected. This was mostly seen among the leuctrids and nemourids. However, there are no known sensitive species among these stoneflies so these shortcomings will not influence on the acidification index.

Species determination of the caddiesflies was good in general. One laboratory had a low number of correct species/genus in 1992, but this was based on few animals. In 1993 only one laboratory had misidentifications, while the others had proper determinations of species/genus. Shortcoming identifications were, however, also recorded in this group.

A low number of sensitive as well as tolerant species belonging to other taxa not named above, were included in the samples. No mistake were made, either regarding genus or species on these invertebrates, but shortcoming determinations were recorded.

Compared with the fault margin of 20 % of the intercalibration of water chemistry, these results are fairly good. Further the estimations of the acidification index, based on the identifications of the laboratories and the material placed in the test-samples were identical in 21 of 22 samples. The explanation for this is that the index is based on common and well known species which had the lowest fault percent. The misidentifications usually occur among rare species or species that seldom are important for the evaluation of the acidification index. The divergence of the index in one sample was due to the record of one sensitive larvae which was "not placed in the sample". The presence of the species can probably be explained by contamination during the processing of this sample which was intended to contain invertebrates typical for a

moderately acidic site. However, the registration of the sensitive species in this case lead to the divergence as mentioned. To guard against this possibility in the future samples should be constructed so as to contain at least two species of the index level intendent.

The mean acidification index based on the delivered samples and the determinations was 0.91 and 0.93, respectively, a good agreement. However, the acidification index technique is relatively insensitive to a quality of taxonomic work compared to many other techniques of community analysis. If the biological data are to be used in other forms of analysis we recommend as a guideline that no more than 5-10% errors in taxonomic identification be tolerated.

The quality of the biological material based on the present intercalibration seems to be generally good and suitable for evaluation of the acidification. The tests included, however, only some of the species listed for the different countries. Reservations must therefore be taken regarding the quality for any part of the fauna. In future tests it will be important to cover as much as possible of the whole fauna. Only with this kind of effort can the real quality of the biological data can be stated.

7. Summary and Conclusions

The International Cooperative Programme on Assessment and Monitoring of Acidification in Rivers and Lakes has been designed to establish the degree and geographic extent of acidification of surface waters. Further, to collect information in order to evaluate dose/response relationships, and to define long-term trends and variations in aquatic chemistry and biota attributable to atmospheric pollution, particularly to acidic deposition. The countries of Europe and North America have representatives in the Programme. Data from 200 sites in 15 countries are available in the database. Approximately half of these have both biological and chemical data, the other have only chemical data. Sites included in the Programme are monitored by each country individually, although the monitoring methods used have been standardised across participants, and intercalibration studies are completed each year. The Programme Centre and the Programme database are located at the Norwegian Institute for Water Research (NIVA).

7.1 Dose/response

The effect of acidification on invertebrates varies from region to region depending on the natural water quality. The most important factor that can modify the tolerance of invertebrates to acid water is the content of humic substances, Ca^{2+} and the ionic strength. In Norway and other areas with high distribution of oligotrophic localities with low conductivity of the water, the fauna have adapted to these conditions. Naturally these localities have pH in the range 5.5 - 6.5, ANC 10-30 $\mu\text{eq/L}$, and Ca^{2+} 0.5-2 mg/L. This environment exclude many invertebrates and is near the tolerance limit for other invertebrates. Small changes in the water quality of such localities can easily change the presence of sensitive species. A pH decrease from 5.5 to 4.7 or a decrease in ANC from 20 to - 30 $\mu\text{eq/L}$ will therefore have a dramatic effect on the fauna community. Within these levels, 4 acidification levels can be defined (the Raddum index). Moderate increase of humic substances or Ca^{2+} will to some extent increase the tolerance of the sensitive taxa.

In aquatic environments with high ionic strength, rich in Ca^{2+} and with high pH, like in Central Europe, the fauna have adapted to this water quality. These regions represent the centre of the geographical distribution area of many sensitive species. Here the selection pressure from the environment on sensitive species will be lower than along the border of the distribution area (example western Norway). This explain why damages on the sensitive fauna occur at higher pH (>6) and ANC (> 50 $\mu\text{eq/L}$) in the central parts of Europe. Many species seem to require ANC > 100 $\mu\text{eq/L}$ to sustain. Due to this, a much wider range of the pH and ANC scale is used in Central Europe than in Scandinavia for stating the tolerance limits of different categories of invertebrates.

In central Europe the diversity of the fauna is much higher than in Scandinavia in unacidified localities, the diversity is usually lower in Central Europe than in Scandinavia. The damages on invertebrates is therefore relatively higher in Central than in Northern Europe at the same acidification levels.

The invertebrates taxa can generally be divided along two lines. One line is the taxa associated with high ANC, pH and conductivity (the sensitive taxa). The other line is species connected with high Al, low pH and conductivity (the tolerant taxa). Critical limit of Al and conductivity were not determined, due to their close relationship of these variables to pH and other factors. A critical limit of 1 mg Ca/L exist, however, for many sensitive species.

7.2 Long-term trends

Long-term trend analysis on water chemistry is a method that is mainly used to reveal trends in a single catchment. Long-term trend analysis on the ICP dataset is the first attempt, as far as we know, to try to discover regional patterns both in Europe and North America in long-term changes in surface water chemistry, due to changes in deposition. It has to be emphasised that the trends have been analysed for an ecologically short period of time (15 years). Therefore detected trends must be considered preliminary. In general, SO₂ emissions have decreased over the last two decades. However, much of this decrease occurred prior to 1980, while most of the datasets used here for both surface water trends and deposition do not begin until the mid 1980's. Thus, the decreasing SO₄²⁻ deposition trends documented by others, are not observed here. On the other hand, although few significant trends were observed in our analysis, many of the non-significant results had negative z-scores, indicative of decreasing trends.

The monitoring sites in the ICP programme include a wide range of watersheds; small streams, big rivers, seepage lakes and drainage lakes of variable size. In addition there is also a wide range in physical characteristics such as climate, topography, hydrology, soil type and depth and amount of acid loading. Although the sites are different in many ways, it is still possible that the same mechanisms can influence on the chemistry in all of the sites. On the other hand, even if some sites show the same trends, it does not necessarily mean that they are controlled by the same mechanisms.

However, we expect to see the results of widespread atmospheric inputs not in single sites, but in several sites having similar characteristics within a region. In the analysis here, we have characterised trends at individual sites, and tried to interpret the results from many sites both within and across several regions.

Trends in surface water chemistry that result from atmospheric acidification are expected to occur at low magnitudes. Climatic cycles such as droughts and wet periods often have a large, though short-term effect on surface water chemistry. Thus the short periods of record, often as short as 5 years for many of these sites, is too short, ecologically speaking, to document long-term changes from relatively small atmospheric inputs. Therefore, the trends reported here should be considered as preliminary results. However, these results are still important, as they outline the limitations of our understanding of these systems.

Although some of the correlation results suggest some interesting trend mechanisms, and mechanisms that can indeed mask the observation of trends due to changes in atmospheric deposition, we have not discovered any statistically significant links between the observed surface water trends, and

deposition chemistry. However, the period of record for both the surface water and deposition chemistry are quite short, so our failure to observe a statistically significant correlation is not too surprising.

In this work we have conducted long-term trend analysis on the sites of the ICP-programme using the Seasonal Kendal tau (SKT) method. The probability of a significant trend, either increasing or decreasing, was measured as a z-score. The result from this analysis showed that a major feature for the sites is a decreasing trend in SO_4^{2-} , either significant or insignificant. There are however, some sites that show increasing trends in SO_4^{2-} and these sites must obviously be controlled by some other mechanisms, such as e.g. local sources of sulphur in the watershed (Turk et al. 1993; Webster et al. 1993). The decrease in SO_4^{2-} does not necessarily result in an improvement of water quality in terms of acidification. There were no obvious patterns that connected decreases in SO_4^{2-} with increases in pH or decrease in base cations (calculated here on the basis of $\text{Ca}^{2+}+\text{Mg}^{2+}$).

Decreases in surface water SO_4^{2-} predominated at the U.S. sites, although these were not accompanied by any specific trend patterns in other variables. At the Canadian sites, decreases in SO_4^{2-} were common, and were accompanied by increases in pH, which may result from decreased acid input with decreased SO_4^{2-} . Concomitant increases in alkalinity were noted in some of these sites, but not all.

Decreases in surface water SO_4^{2-} concentrations were common among European sites, in both river and lake sites. In particular, SO_4^{2-} decreases dominated at the sites in Germany, The Netherlands, and in Norway. Decreases were noted at some of the sites in other countries, but not with the same frequency. Another dominating trend in the European sites was decreasing Ca concentration at many of the sites in the Netherlands, Norway, and Sweden. Concomitant trends in alkalinity were uncommon, except in the Netherlands, where increasing alkalinity was observed along with increases in NH_4^+ . Decreases in NO_3^- concentrations were common across the European sites, although both increases and decreases were observed in the Norwegian and Swedish sites, and only increases were observed at the sites in the Netherlands.

Further the sites were clustered according to chemistry and the trends were correlated to the chemical clusters. Too few clusters were obtained to reflect the number of trend patterns observed across the ICP sites. These clusters were defined by $\text{Ca}^{2+}+\text{Mg}^{2+}$ and SO_4^{2-} concentrations. Few variables were available for every site, and sites with missing values could not be included in the analysis. A more complete data set, including data for Al and NO_3^- , may result in a more successful analysis.

The last method to investigate the data was to cluster them according to trends. This gave 8 clusters, of these 6 had decreasing SO_4^{2-} as a main trend, and most sites fell into the group of decreasing

SO_4^{2-} and increasing NO_3^- . Nearly half of the sites in the clusters with decreasing SO_4^{2-} were in clusters characterised by increasing NO_3^- . These sites are mainly found in Norway, Sweden, the Adirondacks and Catskills regions of New York, and in Ontario, Canada.

Several of these sites are characterised by NO_3^- breakthrough. Thus, and concurrent decreases in base cations, both in surface water concentrations and in deposition, may account for our failure to observe recovery from acidification in sites with decreasing SO_4^{2-} . When NO_3^- inputs are low, surface water NO_3^- values are usually near or below detection limit values because NO_3^- is a nutrient in short supply in most terrestrial ecosystems. Surface water systems with higher NO_3^- values are often indicative of some level of N saturation within the watershed (Kelly et al. 1990). In addition, NO_3^- is a strong acid, and its presence in higher concentrations may indicate significant acid loading. Nitrate showed a range of trends from significantly increasing to significantly decreasing.

7.3 Chemical intercalibration

During the last three years an increasing number of laboratories have been participating in the intercalibration exercises, representing an increasing number of countries, especially in the eastern part of Europe. The results from the intercalibration shows that the results are generally good. In some cases, the low concentration in some of the samples used in these intercalibration exercises, may be outside the routine analytical range for methods routinely used for determination in more polluted samples.

As a main conclusion, it may be stated that it is absolutely necessary that all laboratories are using the same method to obtain comparable results. In addition, an internal quality control procedure should be routinely used to secure that the instrumental system is working satisfactorily.

7.4 Biological intercalibration

The intercalibration of the biological material was generally good and suitable for evaluation of the acidification. The tests included, however, only a part of the species listed for the different countries. Reservations must therefore be taken regarding the quality for every part of the fauna.

8. Further Work - The ICP on Waters Programme in the next three years

To achieve an understanding of the acidification of aquatic ecosystems it is crucial to link changes in water chemistry with changes in biology. Establishment of a better understanding of this relationship is a programme priority. Indeed examples in this report show that effects might be traced differently in different regions and continued work on biota will add valuable information to the Programme as well as to the other ICP's. The importance of an international programme to clarify methods and evaluate data in a uniformed manner should be obvious.

Unanswered chemical questions also remain. For example where declines in sulphate are observed at some locations in Norway, Sweden, Scotland and North American locations, and some of the catchments show improvements in corresponding water acidity, others do not, even though they are spatially near by. The differing response may be explained in terms of changes in base cation concentrations, but further evaluation is needed. The increase in nitrate concentrations, and the question of N-based acidification also requires further assessment.

Long-term data are needed to evaluate areas of improvement, areas of stable acidification and areas of further/future acidification. A comprehensive long-term database on acid sensitive surface waters is of great importance as a starting point for future research programmes. Cooperation with other monitoring programmes under the convention is a priority. As an example, when critical loads are to be tested, trend data will indicate whether specific areas are experiencing levels of acidic deposition that exceed, or are less than, the critical loads established by individual countries.

Annual task force meetings establish monitoring strategies, and the direction of future work. Continuation of the practice of having scientists visit the Programme Centre to work for shorter or longer periods of time will be of great importance for the future success of the Programme.

The ICP on Waters faces challenges along all the lines mentioned in the programme objectives. In addition further Cooperation with the other ICP's and the mapping work will be given priority by the Programme Centre.

The extent and regional aspects of acidified lakes and rivers and sensitive areas was established in the first phase of the programme. Further, regional survey data will be used according to decisions made by the Programme Task Force, to update this information and establish a thorough understanding of the representativeness of catchments in the programme relative to the area.

The long-term trends work has just started as shown in this report. The importance of long-term consistent data records are strongly stressed and indeed the established database has grown to a size where trend results are possible (min. 5 years etc.). This is a topic of general concern, and the ICP-waters will take initiatives to have a general discussion of methods and establish close relationships to the other ICP's on this issue. Long-term data records are valuable in this work and the national focal points will be asked if additional long-term data records can be made available for this kind of work.

As the basis of the work on regional aspects, dose-relationships and trends are the data submitted to enter the international database of the ICP on waters. Data quality control has been stressed as an important part of the programme. Annual intercalibration control has been established. The number of participants in the chemical intercalibration has reached 30 laboratories. Several of the laboratories delivering data to the ICP on integrated monitoring are also participating in this intercalibration work and many laboratories have indicated an interest in joining. The programme has also conducted the first known international intercalibration on benthic fauna, and this will be repeated. Further data base control covers continuity of data, inspection for outliers, and identification of gaps. Indeed call for "missing data" have shown that the submitted data records often can be extended. This kind of tedious work is important but necessary for creating a database which can be broadly used with confidence.

Annual Task Force meetings are used to refine the monitoring strategies, and to determine the priorities for the work at the Programme Centre. Continuation of the practice of having scientists visit the Programme Centre to work for shorter or longer periods of time will be of great importance to the future success of the programme. Transfer of parts of the database to research groups who may wish to take on special issues has also been proposed as a method for future work. Indeed, as the database grows, possibilities for different approaches, evaluations and modelling work are rapidly increasing. All contributions in kind as or even as cash will be appreciated.

It is our wish that the coming three years will show the importance of international Cooperation on methods and development of the data base. Further that the programme will be able to answer questions raised by the other ICP's and give valuable information to the mapping work..

9. Publications from the programme

1. Manual for Chemical and Biological Monitoring. Programme Manual. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo 1987.
2. Intercalibration 8701. pH, Ks, SO₄, Ca. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo 1987.
3. Data Report 1987 and available Data from Previous Years. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo April 1988.
4. Intercalibration 8802. pH, K₂₅, HCO₃, NO₃, SO₄, Cl, Ca, Mg, Na, K. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo August 1988.
5. Proceedings of the Workshop on Assessment and Monitoring of Acidification in Rivers and Lakes, Espoo, Finland, 3rd to 5th October 1988. Prepared by the Finnish Acidification Research Project, HAPRO, Ministry of Environment, October 1988.
6. Intercalibration 8903. Dissolved organic carbon and aluminium fractions. Prepared by the Programme Centre, Norwegian Institute for Water Research. Håvard Hovind, NIVA, Oslo April 1989.
7. Note: Some reflections about the determination of pH and alkalinity. Prepared by the Programme Centre, Norwegian Institute for Water Research. Håvard Hovind, NIVA, Oslo October 1989.
8. Data Report 1988. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo March 1990.
9. Intercalibration in the framework of an international monitoring programme. Proceedings for third annual Ecological Quality Assurance Workshop, Canada Centre for Inland Waters, Burlington Ontario. Prepared by the Programme Centre, Norwegian Institute for Water Research., NIVA, Merete Johannessen, Oslo April 1990.

10. Intercalibration 9004. pH and alkalinity. Prepared by the Programme Centre, Norwegian Institute for Water Research. Håvard Hovind, NIVA, Oslo August 1990.
11. Proceedings for the fifth Meeting of the Programme Task Force Freiburg, Germany, October 17 -19, 1989. Prepared by the Umweltbundesamt, Berlin July 1990
12. Data Report 1989. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo October 1990.
13. Intercalibration 9105. pH, K_{25} , HCO_3 , $NO_3 + NO_2$, Cl, SO_4 , Ca, Mg, Na, K and TOC. Prepared by the Programme Centre, Norwegian Institute for Water Research. Håvard Hovind, NIVA, Oslo July 1991.
14. The Three Year Report. Summary and results 1987 - 1989. Results from the International Cooperative Programme on Assessment and Monitoring of Acidification in Rivers and Lakes. Prepared by the Programme Centre in Cooperation with elected delegates. NIVA, Oslo September 1991.
15. Summary of The Three Year Report 1987 - 1989. Compiled by the Programme Centre, Norwegian Institute for Water Research. NIVA, approved by the Programme Task Force on the 26th of October 1990 in Skövde. Oslo August 1991.
16. Scientific papers presented at the Sixth Task Force meeting in Sweden 23 - 24 October 1990. Swedish Environmental Protection Agency, Sweden, September 1991.
17. Data Report 1990. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo March 1992.
18. Proceedings of an international symposium: Ecological Modifications and Criteria of Ecological Norms. Moscow 1990 (in Russian).
19. Amendment to the Manual: Assessment of regional monitoring data, diatom and fish population monitoring. NIVA, Oslo May 1992.

20. Evaluation of the International Cooperative Programme on Assessment and Monitoring of Acidification in Rivers and Lakes. Prepared by the Programme Centre with an expert panel appointed by the Programme Task Force. NIVA, Oslo 1992.
21. Intercalibration 9206. pH, K_{25} , HCO_3 , $NO_3 + NO_2$, Cl, SO_4 , Ca, Mg, Na, K, Al and DOC. Prepared by the Programme Centre, Norwegian Institute for Water Research. Håvard Hovind, NIVA, Oslo September 1992.
22. Draft Data Report 1991. Prepared by the Programme Centre, Norwegian Institute for Water Research. NIVA, Oslo August 1992.
23. Intercalibration 9307. pH, K_{25} , HCO_3 , $NO_3 + NO_2$, Cl, SO_4 , Ca, Mg, Na, K, total aluminium, reactive and non-labile aluminium, TOC and COD-Mn. Prepared by the Programme Centre, Norwegian Institute for Water Research. Håvard Hovind, NIVA, Oslo September 1993.
24. Intercalibration of Invertebrate Fauna 9301. Prepared by the Programme Centre, Zoological Institute, University of Bergen. Gunnar G. Raddum, Bergen October 1993.

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APPENDIX A

Overview of the sites in the programme and what type of data there exist for each site.

Station name	File	Chemistry code	Year	Site data	Biology year
Austria, Kaernten, Doesener See	IA-AU	DOS-K	1989-1991	x	
Austria, Kaernten, Gradenbach	IA-AU	GRA-K	1989-1991	x	
Austria, Kaernten, Hairlacher See	IA-AU	HAI-T	1988-1991	x	
Austria, Kaernten, Melniksee	IA-AU	MEL-K	1988-1991	x	
Austria, Kaernten, Wangenitzbach	IA-AU	WAN-K	1990-1991	x	
Austria, Tirol, Gossenkoelle See	IA-AU	GKS-T	1988-1991	x	
Austria, Tirol, Mutterberger See	IA-AU	MUT-T	1988-1991	x	
Austria, Tirol, Piburger Bach	IA-AU	PIB-T	1988-1990		
Austria, Tirol, Schwarzee ob Soelden	IA-AU	SOS-T	1989-1991	x	
Austria, Upper Austria, Klafferbach, Rehberg	IA-AU	KLA1-0	1989-1991	x	1989
Austria, Upper Austria, Stungbach vor Klafferbach	IA-AU	KLA2-0	1989-1991	x	1989
Austria, Upper Austria, Klafferbach, Brucke, Freundorf	IA-AU	KLA3-0	1989-1991	x	1989
Austria, Upper Austria, Ramenbach, Brucke Hinterberg	IA-AU	KLA4-0	1989-1991		
Belgium, Eupen, Getz river, Small dam	IA-BE	B03	1987-1991	x	1988
Belgium, Eupen, Vesdre river, Bellesfort	IA-BE	B02	1987-1991	x	1988
Belgium, Eupen, Vesdre, Dam	IA-BE	B01	1987-1991	x	
Belgium, Jalhay, Gileppe river, Chemin des charbonniers	IA-BE	B06	1987-1991	x	1988
Belgium, Jalhay, Helle river, Schornstein	IA-BE	B04	1987-1991	x	1988
Belgium, Jalhay, Lake Gileppe	IA-BE	B05	1987-1991	x	
Belgium, Jalhay, Louba river, Les Hes	IA-BE	B07	1987-1991	x	1988
Belgium, Jalhay, Soor river, Small dam	IA-BE	B08	1987-1991	x	1988
Belgium, Raeren, Steinbach					1990
Canada, Nova Scotia, Beaverskin Lake	IA-CA	C14	1970-1992	x	
Canada, Nova Scotia, Kejimikujik Lake	IA-CA	C13	1970-1982	x	
Canada, Nova Scotia, Little Red Lake	IA-CA	C12	1979-1992	x	
Canada, Nova Scotia, Mount Tom Lake	IA-CA	C10	1980-1992	x	
Canada, Nova Scotia, Mountain Lake	IA-CA	C11	1970-1992	x	
Canada, Ontario, Algoma Region, Batchawana Lake	IA-CA	C01	1980-1991	x	
Canada, Ontario, Algoma Region, Little Turkey Lake	IA-CA	C03	1980-1991	x	
Canada, Ontario, Algoma Region, Turkey Lake	IA-CA	C04	1980-1991	x	
Canada, Ontario, Algoma Region, Wishart Lake	IA-CA	C02	1980-1991	x	
Canada, Ontario, Lake 111	IA-CA	CA111	1987-1990		1990
Canada, Ontario, Lake 224	IA-CA	CA224	1987-1990		1990
Canada, Ontario, Lake 239	IA-CA	CA239	1987-1990		1990
Canada, Ontario, Lake 262	IA-CA	CA262	1987-1990		1990
Canada, Ontario, Lake 305	IA-CA	CA305	1987-1990		
Canada, Ontario, Lake 373	IA-CA	CA373	1987-1990		1990
Canada, Quebec, Laflamme Lake	IA-CA	C08	1981-1991	x	
Canada, Quebec, Parc de la Jacques-Cart., Lac Bonneville	IA-CA	C07	1981-1991	x	
Canada, Quebec, Reserve faunique des Laur., Lac MacCeod	IA-CA	C09	1982-1991	x	
Canada, Quebec, Reserve faunique des Laur., Lac Josselin	IA-CA	C06	1982-1991	x	
Canada, Quebec, Reserve faunique des Laur., Lac Veilleux	IA-CA	C05	1982-1991	x	
Denmark, Sestrup Sande, Skaerbaek, Station A	IA-DA	SK297A	1989-1990	x	
Denmark, Sestrup Sande, Skaerbaek, Station B	IA-DA	SK110B	1976-1991	x	
Denmark, Sestrup Sande, Skaerbaek, Station C	IA-DA	SK298C	1989-1990	x	
Denmark, Sestrup Sande, Skaerbaek, Station D	IA-DA	SK299D	1989-1991	x	
Denmark, Sestrup Sande, Skaerbaek, Station E	IA-DA	SK037E	1987-1990	x	
Denmark, Sestrup Sande, Skaerbaek, Station F	IA-DA	SK038F	1987-1991	x	
Finland, Hirvilampi	IA-SF	SF01	1978-1992	x	1988
Finland, Kivijaervi	IA-SF	SF04	1962-1992	x	
Finland, Maekilampi	IA-SF	SF03	1978-1992	x	1988
Finland, Vuorilampi	IA-SF	SF02	1978-1992	x	1988
FRG, Black Forest, Duerreychblach	IA-FRG	Due6	1987-1992	x	1985-1988
FRG, Black Forest, Goldersbach	IA-FRG	Go17	1986-1992	x	1985-1988
FRG, Black Forest, Huettenbaechle	IA-FRG	Hue2	1986-1991	x	1985-1988
FRG, Black Forest, Kaltenbach	IA-FRG	Kal4	1988-1991		
FRG, Balck Forest, Kaltenbach	IA-FRG	Kal5	1986-1989	x	1985-1988
FRG, Black Forest, Kinzigalsperre	IA-FRG	Kin3	1988-1989		
FRG, Black Forest, Kleine Kinzig	IA-FRG	Kle1	1986-1992	x	1985-1987
FRG, Black Forest, Kleine Kinzig	IA-FRG	Kle3	1987-1987		

Station name	File	Chemistry code	Year	Site data	Biology year
FRG, East Bavaria, Grosse Ohe	IA-FRG	Gro33	1979-1992	x	1985-1988
FRG, East Bavaria, Hinterer Schachtenbach	IA-FRG	Hin31	1983-1992	x	1985-1988
FRG, East Bavaria, Seebach	IA-FRG	See30	1983-1992	x	1985-1988
FRG, East Bavaria, Vorderer Schachtenbach	IA-FRG	Vor32	1983-1992	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal1	1986-1991	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal2	1986-1992	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal3	1986-1992	x	1988-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal5	1986-1987		
FRG, East Bavaria, Waldnaab	IA-FRG	Wal6	1988-1988		
FRG, East Bavaria, Waldnaab	IA-FRG	Wal8	1986-1992	x	1988-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal9	1986-1992	x	1988-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal11	1986-1988		1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal12	1986-1991	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal14	1986-1992	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal15	1987-1987		1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal16	1986-1991	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal17	1986-1987	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal18	1986-1991	x	1985-1988
FRG, East Bavaria, Waldnaab	IA-FRG	Wal15a	1987-1987		
FRG, East Bavaria, Waldnaab	IA-FRG	Wal17a	1987-1988		
FRG, East Bavaria, Waldnaab	IA-FRG	Wal17b	1988-1991		
FRG, Fichtelgebirge, Eger	IA-FRG	Ege1	1982-1992		
FRG, Fichtelgebirge, Roeslau	IA-FRG	Roe2	1982-1992		
FRG, Fichtelgebirge, Zinnbach	IA-FRG	Zin3	1983-1992		
FRG, Harz, Alte Riefensbeek	IA-FRG	AltR1	1986-1991	x	1986-1987
FRG, Harz, Alte Riefensbeek	IA-FRG	AltR2	1986-1991	x	1986-1987
FRG, Harz, Alte Riefensbeek	IA-FRG	AltR3	1986-1991	x	1986-1987
FRG, Harz, Dicke Bramke	IA-FRG	Dic	1988-1992		
FRG, Harz, Grosse Bode	IA-FRG	Gro14	1988-1992		
FRG, Harz, Grosse Soese	IA-FRG	GRoS1	1986-1991	x	1986-1987
FRG, Harz, Grosse Soese	IA-FRG	GroS2	1986-1992	x	1986-1987
FRG, Harz, Grosse Soese	IA-FRG	GroS3	1986-1991	x	1986-1987
FRG, Harz, Lange Bramke	IA-FRG	LanL1	1989-1989	x	1986-1987
FRG, Harz, Lange Bramke	IA-FRG	Lan12	1988-1991	x	1986-1987
FRG, Harz, Lange Bramke	IA-FRG	Lan13	1988-1991	x	
FRG, Harz, Mollentalbach	IA-FRG	Mol10	1986-1991	x	1986-1988
FRG, Harz, Grosse Schacht	IA-FRG	Sch9	1986-1991	x	1986-1987
FRG, Harz, Varleybach	IA-FRG	Var1	1986-1991	x	
FRG, Harz, Varleybach	IA-FRG	Var2	1986-1991	x	
FRG, Harz, Warme Bode	IA-FRG	War14		x	
FRG, Harz, Warme Bode	IA-FRG	War15	1988-1991	x	
FRG, Hunsrueck, Grafenbach	IA-FRG	Gra5	1982-1992	x	1985-1988
FRG, Hunsrueck, Schwallbach	IA-FRG	Sch10	1982-1991	x	1985-1988
FRG, Hunsrueck, Traunbach	IA-FRG	Tral	1982-1992	x	1985-1988
FRG, Hunsrueck, Traunbach	IA-FRG	Tra2	1984-1991	x	1985-1988
FRG, Kaufunger Wald, Nieste	IA-FRG	Nie1	1987-1992	x	1988-1988
FRG, Kaufunger Wald, Nieste	IA-FRG	Nie2	1987-1991	x	1988-1988
FRG, Kaufunger Wald, Nieste	IA-FRG	Nie3	1987-1992	x	1988-1988
FRG, Kaufunger Wald, Nieste	IA-FRG	Nie4	1987-1991	x	1988-1988
FRG, Kaufunger Wald, Nieste	IA-FRG	Nie5	1987-1992	x	1988-1988
FRG, Lauenburg, Garrensee	IA-FRG	Gar1	1986-1991	x	1986-1988
FRG, Lauenbrug, Pinnsee	IA-FRG	Pin3	1986-1992	x	1986-1988
FRG, Lauenburg, Ploetschersee	IA-FRG	Plo2	1986-1991	x	1986-1988
FRG, Odenwald, Schmerbach	IA-FRG	Sch1	1984-1992	x	1988-1988
FRG, Odenwald, Schmerbach	IA-FRG	Sch3	1985-1992	x	1988-1988
FRG, Odenwald, Schmerbach	IA-FRG	Sch4	1988-1991	x	1988-1988
FRG, Odenwald, Schmerbach, Hiltersklinger Teich	IA-FRG	Sch2	1985-1991	x	1988-1988
FRG, Rothaargebirge, Elberndorfer Bach	IA-FRG	Elb1	1986-1992	x	1986-1988
FRG, Rothaargebirge, Zinse	IA-FRG	Zin2	1986-1992	x	1986-1988
FRG, Taunus, Rombach	IA-FRG	Rom1	1988-1991	x	1988-1988
FRG, Taunus, Rombach	IA-FRG	Rom2	1986-1992	x	1988-1988
FRG, Taunus, Rombach	IA-FRG	Rom3	1986-1992	x	1988-1988
FRG, Taunus, Rombach	IA-FRG	Rom4	1987-1991	x	1988-1988
FRG, Taunus, Silberbach	IA-FRG	Sil5	1986-1991	x	1988-1988

Station name	File	Chemistry code	Year	Site data	Biology year
FRG, Erzgebirge, Neunzehnhain	IA-FRG	Neu8	1992-		1992-
FRG, Erzgebirge, Sosa	IA-FRG	Sos9	1992-		1992-
FRG, Erzgebirge, Große Pyra	IA-FRG	Gro2	1992-		1992-
FRG, Erzgebirge, Wolfsbach	IA-FRG	Wol1	1992-		1992-
FRG, Erzgebirge, Rote Pockau	IA-FRG	Rot3	1992-		1992-
FRG, Erzgebirge, Wilde Weißeritz	IA-FRG	Wil4	1992-		1992-
FRG, Coditzer Forst, Ettelsbach	IA-FRG	Ett6	1992-		1992-
FRG, Dahleener Heide, Heidelbach	IA-FRG	Hei7	1992-		1992-
FRG, Elbsandsteingebirge, Taubenbach	IA-FRG				
Hungary					
Ireland, Doo Lough, Inflow 1	IA-IR	DOO11	1987-1989	x	
Ireland, Doo Lough, Inflow 2	IA-IR	DOO12	1987-1989	x	
Ireland, Doo Lough, Inflow 3	IA-IR	DOO13	1987-1989	x	
Ireland, Doo Lough, Mid Lake	IA-IR	DOO10	1987-1989	x	
Ireland, Glendalough, Lake Upper, Inflow 1	IA-IR	GLE11	1987-1993	x	1987-1988
Ireland, Glendalough, Lake Upper, Inflow 2	IA-IR	GLE12	1987-1993	x	1987-1988
Ireland, Glendalough, Lake Upper, Inflow 3	IA-IR	GLE13	1987-1993	x	
Ireland, Glendalough, Lake Upper, Mid Lake	IA-IR	GLE10	1987-1993	x	
Ireland, Lough Maumwee, Inflow 1	IA-IR	MAU11	1987-1993	x	1988-1989
Ireland, Lough Maumwee, Inflow 2	IA-IR	MAU12	1987-1993	x	1988-1989
Ireland, Lough Maumwee, Mid Lake	IA-IR	MAU10	1987-1993	x	1988-1989
Ireland, Lough Naminna, Inflow 1	IA-IR	NAM11	1987-1989	x	1988-1989
Ireland, Lough Naminna, Mid Lake	IA-IR	NAM10	1987-1989	x	1988-1989
Ireland, Lough Veagh, Mid Lake	IA-IR	VEA10	1988-1993		1988-1989
Ireland, Lough Veagh, Inflow 1	IA-IR	VEA11	1988-1993		1988-1989
Ireland, Lough Veagh, Inflow 2	IA-IR	VEA12	1988-1993		1988-1989
Ireland, Lough Veagh, Inflow 3	IA-IR	VEA13	1988-1993		1988-1989
Ireland, Lough Veagh, Inflow 4	IA-IR	VEA14	1988-1993		1988-1989
Norway					
Norway, Aust-Agder, Birkenes stream, Birkenes	BIE01		1972-1991	x	
Norway, Aust-Agder, Tovdalselva, Bøen bruk	OVELV	7 1	1980-1991	x	
Norway, Buskerud, Langtjern, outflow	LAE01		1972-1991	x	
Norway, Finnmark, Dalelva, Jarfjord	DALELV	1	1988-1991	x	
Norway, Oppland, Aurdøla, Aurdalsfjorden	OVELV	90 1	1986-1991	x	
Norway, Rogaland, Oгна					1984-1988
Norway, Rogaland, Vikedalselva, Vindafjord	OVELV	32 9	1982-1991	x	1987-1988
Norway, Sogn og Fjordane, Gaular, Eldalen	OVELV	57 3	1980-1991		1985-1988
Norway, Sogn og Fjordane, Nausta, Naustdal	OVELV	34 1	1980-1991	x	1985-1988
Norway, Sogn og Fjordane, Trodøla	OVELV	34 5	1984-1991	x	1986-1988
Norway, Telemark, Storgama, Outflow	STE01		1974-1991	x	
Norway, Vest-Agder, Farsund					
Gjærvollstadvatn and Saudlandsvatn					1981-1988
Sweden					
Sweden, Alsteraan, Getebro	IA-SV	S02	1985-1991	x	
Sweden, Alsteraan, Stroemsborg	IA-SV	S03	1985-1991	x	
Sweden, Anraasen, Haersvatn	IA-SV	S04	1984-1991	x	
Sweden, Brunnsjoen	IA-SV	S08	1984-1991	x	1986-1988
Sweden, Delaangersaan, Iggersund	IA-SV	S01	1984-1991	x	
Sweden, Fiolen	IA-SV	S09	1984-1991	x	1986-1988
Sweden, Fraecksjoen	IA-SV	S11	1984-1991	x	1986-1988
Sweden, Haersvatn	IA-SV	S12	1984-1991	x	1986-1988
Sweden, Moecklasjoen	IA-SV	S07	1984-1987	x	
Sweden, Stensjoen	IA-SV	S06	1985-1991	x	1986-1988
Sweden, Storasjoen	IA-SV	S10	1984-1991	x	1986-1988
Sweden, Tvaeringen	IA-SV	S05	1984-1991	x	
The Czech Republic					

Station name	File	Chemistry code	Year	Site data	Biology year
The Netherlands, Achterste Goorven, station A	IA-NE	AGA	1958-1991	x	
The Netherlands, Achterste Goorven, station B	IA-NE	AGB	1925-1991	x	
The Netherlands, Achterste Goorven, station E	IA-NE	AGE	1975-1991	x	
The Netherlands, Gerritsfles	IA-NE	GER	1991-1991	x	
The Netherlands, Kliplo	IA-NE	KLI	1958-1991	x	
UK, N. Ireland, Blue Loch	IA-UK	UK21	1990-1992		
UK, England, Scoat Tarn	IA-UK	UK10	1988-1992		1988-1988
UK, Scotland, Loch Coire nan Arr	IA-UK	UK01	1988-1992		1988-1988
UK, Scotland, Lochnager	IA-UK	UK04	1990-1992		
UK, Scotland, Round of Glenshead	IA-UK	UK07	1988-1992		1988-1988
UK, Wales, Wales, Llyn Llagi	IA-UK	UK15	1988-1992		1988-1988
USA, Colorado, Seven Lakes	IA-US	4E2009	1985-1990	x	
USA, Colorado, Summit Lake	IA-US	4E2060	1985-1990	x	
USA, Colorado, Sunlight Lake	IA-US	4E2070	1985-1990	x	
USA, Colorado, White Dome Lake	IA-US	4E2071	1985-1990	x	
USA, Maine, Little Long Pond	IA-US	1E1132	1982-1990	x	
USA, Maine, Tilden Pond	IA-US	1E1133	1982-1990	x	
USA, Michigan, Andrus	IA-US	2B3082	1983-1990	x	
USA, Michigan, Buckeye	IA-US	2B2102	1983-1990	x	
USA, Michigan, Johnson	IA-US	2B1047	1983-1990	x	
USA, Minnesota, Cruiser	IA-US	2A2063	1983-1990	x	
USA, New York, Adirondack Mountain, Arbutus	IA-US	1A1052	1983-1990	x	
USA, New York, Adirondack Mountain, Constable	IA-US	1A1017	1982-1990	x	
USA, New York, Adirondack Mountain, Dart Lake	IA-US	1A1106	1982-1990	x	
USA, New York, Adirondack Mountain, Heart Lake	IA-US	1A1102	1982-1990	x	
USA, New York, Adirondack Mountain, Lake Rondaxe	IA-US	1A1110	1982-1990	x	
USA, New York, Adirondack Mountain, Moss Lake	IA-US	1A1109	1982-1990	x	
USA, New York, Adirondack Mountain, Otter Lake	IA-US	1A2078	1982-1990	x	
USA, New York, Catskill M., East Branc Neversink, Mid.	IA-US	143010	1983-1990	x	
USA, New York, Catskill m., High Falls Brook	IA-US	143105	1983-1990	x	
USA, Wisconsin, Luna	IA-US	2C2062	1983-1990	x	
USA, Wisconsin, Nichols	IA-US	2C1069	1983-1990	x	
USA, Wisconsin, Sand	IA-US	2C1068	1983-1990	x	
USSR, Segozero					
USSR, Ondozero					
USSR, Vygozersko-Ondskoye, Rsvr.					
USSR, Serebryanskoe Rsvr.					
USSR, Lovozero					
USSR, Umbozero					
USSR, Rugozero					
USSR, Suoyari					
USSR, Kola					

Appendix B

Location of the monitoring stations and mean values for major parameters for each station in the years 1987-1993.

Units:

pH

Ca²⁺ mg/L

Mg²⁺ mg/L

Na⁺ mg/L

K⁺ mg/L

Cl⁻ mg/L

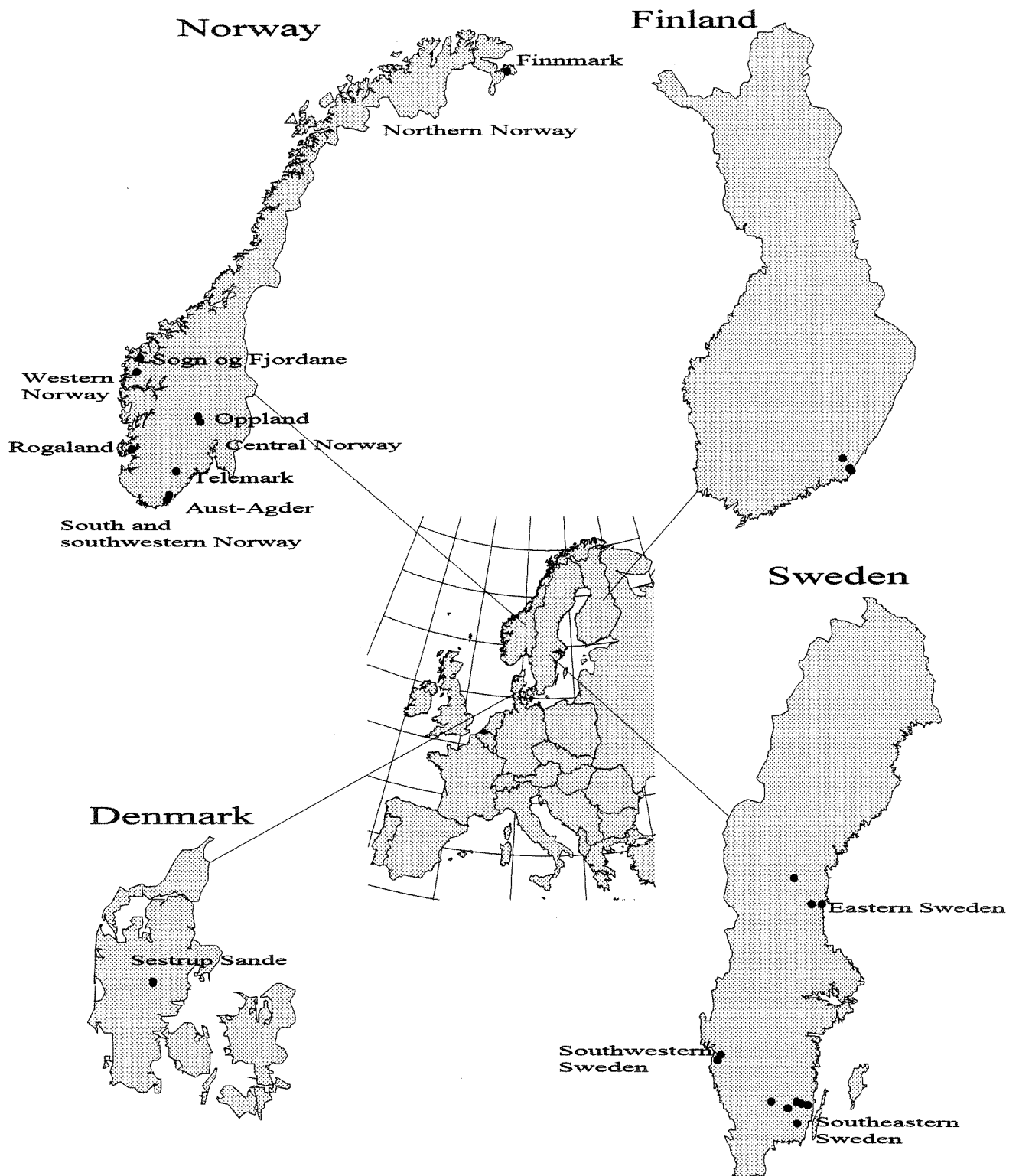
SO₄²⁻ mg/L

Alk-E µeq/L

NO₃⁻ µg N/L

NH₄⁺ µg N/L

The Nordic countries



pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
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The Nordic Countries

Norway

Central Norway; Buskerud. Oppland. Telemark

1987 LAE01	6.0	1.1	0.2	0.6	0.1	0.4	2.4	20	28	
1988 Langtjern	6.0	1.1	0.2	0.5	0.1	0.4	2.3	23	46	13
1989	6.1	1.2	0.2	0.6	0.2	0.6	2.6	27	40	
1990	6.1	1.1	0.2	0.6	0.2	0.5	2.5	24	28	
1991	6.2	1.2	0.2	0.7	0.2	0.6	2.5	31	30	
1992	6.2	1.3	0.2	0.7	0.2	0.7	2.5	32	26	
1993	6.1	1.1	0.2	0.7	0.2	0.6	2.2	30	29	
1987 OVELV 90 1	5.6	0.4	0.2	0.9	0.2	1.4	1.3	4	79	10
1988 Aurdøla	5.6	0.4	0.2	0.8	0.2	1.4	1.4	4	81	8
1989	5.4	0.4	0.2	1.6	0.2	3.1	1.3	4	71	
1990	5.3	0.4	0.2	1.6	0.2	3.1	1.3	2	88	
1991	5.6	0.4	0.1	0.9	0.2	1.4	1.1	2	97	
1992	5.5	0.4	0.3	1.8	0.3	3.6	1.2	2	74	
1993	5.4	0.6	0.3	2.4	0.3	4.9	1.4	3	104	
1987 STE01	4.9	0.9	0.3	1.5	0.3	2.4	3.8	<0	165	
1988 Storgama	4.9	0.8	0.3	1.5	0.3	2.4	3.5	<0	175	35
1989	5.0	0.9	0.3	1.6	0.3	2.7	3.7	14	199	
1990	4.9	0.9	0.3	1.7	0.3	3.2	3.6	<0	162	
1991	5.1	0.9	0.3	1.8	0.3	3.0	3.5	<0	158	
1992	5.1	1.0	0.3	1.9	0.3	3.3	3.6	<0	155	
1993	5.0	1.0	0.3	2.2	0.3	3.9	2.9	5	138	

Southern and Southwestern Norway; Aust Agder. Rogaland

1987 BIE01	5.4	0.7	0.4	2.0	0.2	3.6	2.3	<0	173	
1988 Birkenes	5.6	0.7	0.3	1.8	0.2	3.2	2.3	8	150	13
1989	5.3	0.6	0.4	2.2	0.2	4.0	2.2	<0	158	
1990	5.3	0.6	0.4	2.6	0.2	5.0	2.3	<0	198	
1991	5.4	0.6	0.3	2.2	0.2	4.0	2.2	2	156	
1992	5.6	0.6	0.3	2.1	0.2	3.9	2.0	2	136	
1993	5.4	0.7	0.4	3.2	0.2	5.8	2.1	<0	153	
1987 OVELV 7 1	5.9	0.5	0.2	1.2	0.3	2.0	1.3	11	76	
1988 Tovdalselva	5.8	0.5	0.2	1.2	0.3	2.0	1.3	13	72	17
1989	5.6	0.5	0.3	1.9	0.3	3.5	1.3	6	60	
1990	5.6	0.4	0.2	1.7	0.2	3.1	1.3	2	68	
1991	5.8	0.5	0.2	1.2	0.2	1.9	1.2	8	89	
1992	5.8	0.6	0.3	2.0	0.3	3.9	1.3	5	55	
1993	5.6	0.6	0.4	2.8	0.3	5.2	1.5	7	82	
1987 OVELV 32 9	5.6	0.4	0.2	1.3	0.2	2.1	1.3	3	57	
1988 Vikesdalselva	5.5	0.3	0.2	1.3	0.2	2.2	1.3	5	60	10
1989	5.4	0.3	0.3	1.7	0.2	3.1	1.3	1	54	12
1990	5.3	0.3	0.3	1.8	0.2	3.4	1.3	<0	61	
1991	5.5	0.3	0.2	1.4	0.2	2.4	1.1	2	64	
1992	5.5	0.4	0.3	1.9	0.2	3.6	1.3	1	55	
1993	5.5	0.4	0.3	2.5	0.2	4.4	1.3	3	64	

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
Western Norway; Sogn og Fjordane										
1987 OVELV 57 3	4.8	1.0	0.2	0.5	0.1	0.5	2.9	<0	24	22
1988 Gaular	4.7	0.9	0.2	0.5	0.1	0.5	2.7	<0	24	21
1989	4.9	1.0	0.2	0.6	0.1	0.6	2.9	<0	22	
1990	4.8	1.0	0.2	0.6	0.1	0.7	3.0		23	
1991	5.0	1.2	0.2	0.7	0.1	0.7	3.1	11	20	
1992	5.0	1.3	0.2	0.7	0.2	0.8	3.1	9	22	
1993	4.8	1.2	0.2	0.7	0.1	0.7	2.5	5	19	
1987 OVELV 34 1	4.6	0.7	0.2	0.9	0.1	1.6	3.3	<0	110	43
1988 Nausta	4.6	0.7	0.1	0.8	0.1	1.4	3.2	<0	140	54
1989	4.6	0.6	0.1	0.9	0.1	1.5	3.4	<0	123	
1990	4.5	0.6	0.1	0.9	0.1	1.5	3.2	<0	102	
1991	4.6	0.6	0.1	0.9	0.1	1.3	3.0	<0	95	
1992	4.6	0.7	0.1	1.0	0.1	1.7	3.0	<0	86	
1993	4.7	0.7	0.1	1.1	0.1	1.8	2.7	<0	95	
1987 OVELV 34 5	4.7	1.2	0.4	2.6	0.2	4.5	5.4	<0	124	61
1988 Trodøla	4.8	1.0	0.3	2.6	0.3	4.1	5.2	2	142	66
1989	4.8	1.4	0.4	3.0	0.3	5.2	5.9	38	191	
1990	4.7	1.1	0.4	3.1	0.3	5.9	5.2	<0	122	44
1991	4.7	1.1	0.3	3.1	0.2	5.1	4.9	<0	164	
1992	4.6	1.1	0.4	3.3	0.1	5.4	5.5	<0	139	
1993	4.6	1.3	0.4	3.9	0.1	7.8	4.8	<0	89	
Northern Norway										
1988 DALELELV 1	6.1	1.5	0.9	3.2	0.3	5.1	5.7	22		
1989 Dalelva	6.0	1.5	0.9	3.4	0.3	5.6	9.9	21		
1990	6.1	1.6	1.0	3.5	0.3	5.8	5.9	28		
1991	6.1	1.6	1.0	3.8	0.3	6.3	5.8	30		
1992	6.3	1.7	1.0	4.0	0.3	6.7	5.5	29		

Sverige

Eastern Sweden

1987 S01	6.8	3.6	1.1	2.8	0.8	2.3	5.0	155	187	7
1988 Delangersaan	6.8	3.5	1.1	2.8	0.7	2.3	4.7	159	102	12
1989 Iggersund	6.7	3.6	1.1	2.7	0.7	2.5	5.0	158	146	14
1990	6.8	3.7	1.2	2.5	0.8	2.7	5.7	172	100	11
1991	6.8	3.8	1.2	2.7	0.7	2.5	5.4	171	143	19
1992	6.9	3.7	1.2	2.9	0.8	2.7	5.3	177	107	9
1987 S05	6.5	2.7	0.7	1.4	0.6	1.0	2.7		26	16
1988 Tvaeringen	6.2	2.7	0.6	1.2	0.6	0.9	2.7	89	71	14
1989	6.5	2.6	0.6	1.2	0.6	0.8	2.9	96	28	14
1990	6.6	2.6	0.7	1.3	0.6	0.9	3.1	124	18	13
1991	6.5	2.8	0.7	1.4	0.5	0.9	2.8	125	26	18
1992	6.4	2.6	0.7	1.3	0.6	1.0	2.5	123	43	19
1987 S06	6.0	1.4	0.4	1.1	0.3	0.6	2.6	20	33	18
1998 Stensjoen	6.1	1.4	0.4	1.2	0.3	0.7	2.9	36	36	18
1989	6.0	1.6	0.4	1.2	0.3	0.8	2.8	39	34	17
1990	6.1	1.3	0.4	1.2	0.3	0.7	2.9	44	12	10
1991	6.0	1.5	0.4	1.3	0.3	0.8	2.5	41	22	13
1992	6.1	1.4	0.4	1.3	0.3	0.8	2.5	52	28	16

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
Southeastern Sweden										
1987 S08	5.3	4.8	1.8	4.8	0.8	6.2	17.0		116	34
1988 Brunnsjoen	5.2	4.1	1.6	4.5	0.7	5.4	13.5		62	110
1989	5.2	3.9	1.4	4.2	0.7	5.7	11.4	12	106	21
1990	5.3	5.5	2.0	4.9	0.9	6.6	19.2	24	120	48
1991	5.3	5.9	2.2	5.4	0.8	7.4	19.5	27	133	42
1992	5.3	5.5	2.1	5.5	0.9	7.6	18.1	23	137	34
1987 S09	6.1	4.2	1.4	4.4	1.5	7.6	12.4	80	94	27
1988 Fiolen	6.1	3.9	1.2	3.9	1.2	6.6	10.7	37	138	24
1989	6.2	3.8	1.2	3.7	1.4	6.8	10.3	31	195	16
1990	6.1	3.9	1.2	3.7	1.2	6.9	11.0	32	156	14
1991	6.2	3.8	1.3	4.1	1.2	7.1	11.0	40	101	13
1992	6.3	3.7	1.3	4.3	1.4	7.4	11.1	44	73	15
1987 S10	5.2	2.0	0.8	3.3	0.5	4.0	7.3		45	44
1988 Storasjoen	4.9	2.1	0.7	3.1	0.4	4.0	5.6		34	98
1989	5.1	1.9	0.6	2.7	0.5	3.8	5.9		83	41
1990	5.2	1.8	0.7	3.0	0.5	4.5	7.3		9	10
1991	5.4	1.9	0.7	3.3	0.5	4.4	6.6	12	17	14
1992	5.4	1.8	0.8	3.5	0.5	4.9	6.5	7	21	25
1987 S02	6.4	5.6	1.8	5.4	1.0	6.6	15.3	80	134	31
1988 Alsteraan.	6.4	5.4	1.5	4.7	0.8	5.5	11.4	106	69	16
1989 Getebro	6.5	5.3	1.4	4.4	0.8	5.4	10.1	102	176	22
1990	6.5	6.9	1.9	5.4	1.1	7.7	17.3	111	85	15
1991	6.6	6.8	2.0	5.8	0.9	8.5	15.8	123	114	13
1992	6.7	6.7	2.0	6.2	1.0	8.8	14.7	130	89	11
1987 S03	6.3	5.9	1.8	7.0	1.0	6.9	16.8	95	231	30
1988 Stroemsborg	6.1	5.4	1.5	5.3	0.8	5.7	12.2	91	201	43
1989	6.3	5.2	1.4	5.7	0.8	5.9	11.1	99	271	6
1990	6.3	6.4	1.9	6.9	1.1	8.4	17.7	103	178	21
1991	6.3	6.4	1.9	7.2	1.0	8.8	15.1	124	241	29
1992	6.4	6.4	2.0	8.1	1.2	9.3	16.4	116	194	28
Southwestern Sweden										
1987 S04	4.4	1.1	0.9	6.1	0.5	9.7	9.4		152	59
1988 Aneraasen.	4.5	1.0	0.9	5.3	0.4	8.1	8.9		144	93
1989 Haersvatn	4.5	1.1	1.0	5.6	0.5	9.9	8.8		165	57
1990	4.4	1.0	1.1	6.5	0.5	11.4	8.9	83	172	49
1991	4.5	1.2	1.1	7.5	0.5	12.2	8.7		172	48
1992	4.4	1.1	1.2	8.0	0.6	12.9	8.7		171	70
1987 S11	6.2	4.4	1.4	7.0	0.7	10.8	11.0	47	71	42
1988 Fracksjoen	6.0	3.9	1.3	5.9	0.6	8.4	10.4	56	140	12
1989	6.2	4.4	1.4	6.2	0.7	10.8	10.4	46	142	27
1990	5.9	4.5	1.6	7.1	0.7	12.4	11.0	51	119	38
1991	6.0	4.6	1.6	7.9	0.8	13.5	10.6	69	114	50
1992	6.0	4.5	1.6	8.4	0.8	14.1	10.3	53	129	40
1987 S12	4.4	1.1	0.9	6.3	0.4	9.9	9.5		138	61
1988 Haersvatn	4.4	0.9	0.9	5.3	0.4	8.3	9.0		163	51
1989	4.4	1.0	0.9	5.5	0.5	9.9	9.0		188	48
1990	4.4	0.9	1.1	6.0	0.5	11.5	8.9		211	57
1991	4.5	1.2	1.1	7.2	0.5	12.5	8.6		166	63
1992	4.4	1.1	1.2	7.9	0.6	13.3	8.6		183	78

pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
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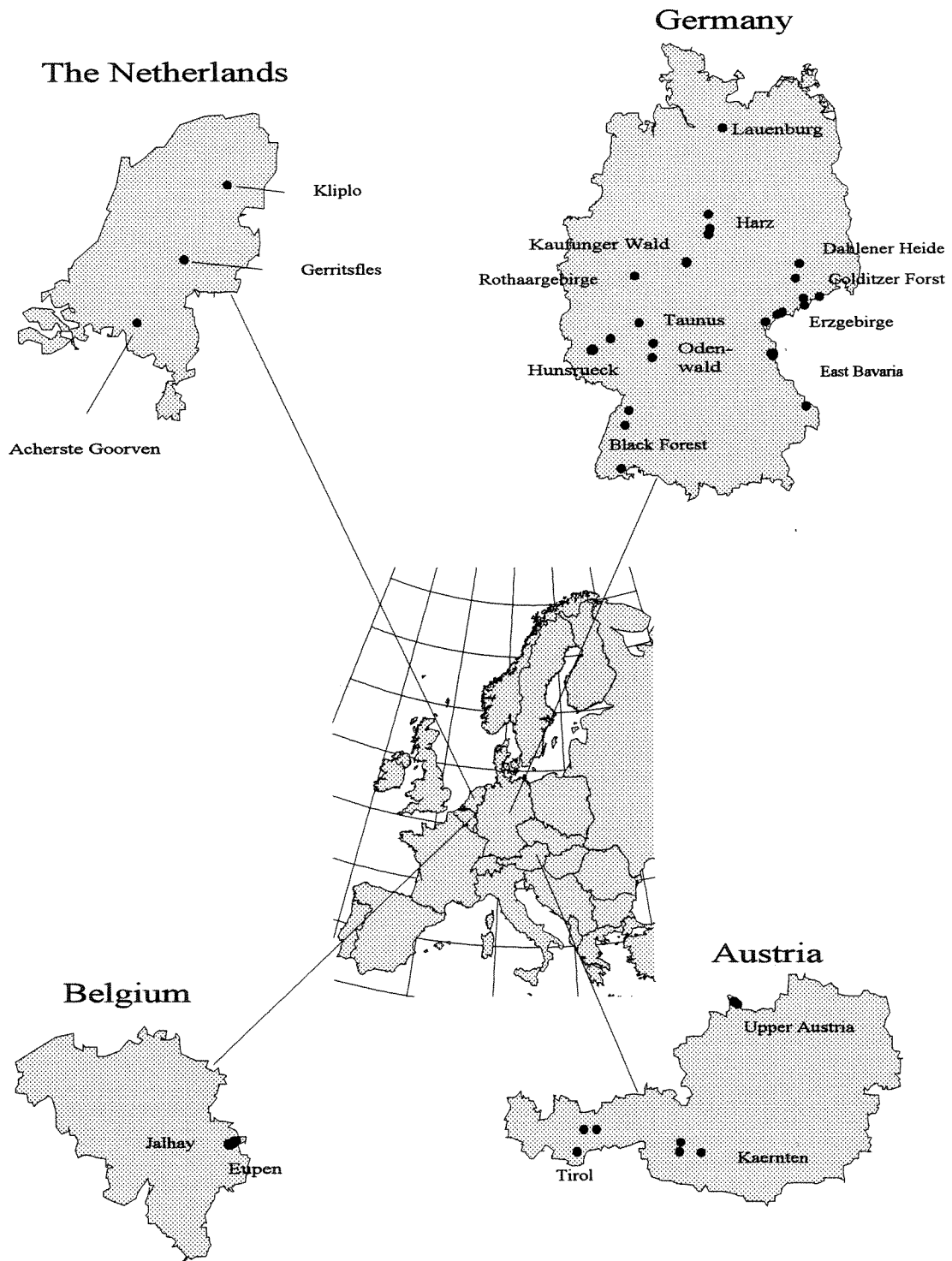
Finland

1987 SF01	5.2	2.1	0.6	1.4	0.3	1.5	9.4	-7	33	25
1988 Hirvilampi	5.1	2.3	0.8	1.4	0.4	1.3	9.8		218	
1989	5.1	2.1	0.6	1.4	0.3	1.6	9.2	-9	170	
1990	5.2	2.0	0.6	1.4	0.4	1.4	9.2	-4	34	
1991	5.3	1.9	0.5	1.3	0.3	1.3	9.1	3	20	
1992	5.3	1.9	0.6	1.5	0.4	1.6	8.8	1	23	71
1987 SF02	5.4	2.3	0.6	1.6	0.4	1.5	9.9	2	31	22
1988 Vourlampi	5.4	2.7	0.8	1.5	0.3	1.2	9.9		146	
1989	5.5	2.3	0.6	1.6	0.4	1.5	9.8	4	250	
1990	5.5	2.3	0.6	1.5	0.4	1.5	8.1	7	45	
1991	5.5	2.2	0.7	1.7	0.4	1.4	8.5	23	39	
1992	5.6	2.0	0.6	1.8	0.4	1.6	8.2	14	22	93
1987 SF03	5.0	2.1	0.4	0.9	0.4	1.4	8.1	-13	37	12
1988 Maekilampi	5.0	2.2	0.6	0.9	0.5	1.3	7.7		41	
1989	4.9	1.9	0.4	0.9	0.4	1.4	7.6	-14	330	
1990	5.1	1.9	0.4	0.9	0.6	1.3	7.4	-9	43	
1991	5.7	2.7	0.4	0.9	0.5	1.2	7.9	26	21	
1992	6.0	3.2	0.4	0.9	0.5	1.5	7.7	50	27	71
1987 SF04	7.1	6.9	1.8	2.8	1.7	4.5		211	31	16
1988 Kivijaervi	6.9					4.4		210	310	9
1990	6.9	7.1	1.8	2.7	1.8	4.5	3.7			
1991	7.0	7.2	2.0	2.3	2.4	4.6			58	
1992	6.8	7.2	1.9	2.8	1.8	4.8				

Denmark

1987 SK037E	6.5				1.2			91	2767	23
1988 Sestrup Sande.	6.4				1.3			93	2573	17
1989 Skaerbaek. SF	6.5	10.3	3.0	9.3	1.3	16.6	19.1	99	2990	17
1990	6.4	10.1	3.0	9.9	1.5	16.1	18.5	83	3170	47
1987 SK038F	6.6				1.2			92	2789	21
1988 Sestrup Sande.	6.5				1.3			96	2710	19
1989 Skaerbaek. SF	6.6	10.6	2.9	9.3	1.3	16.7	19.0	100	2988	17
1990	6.6	10.3	2.9	9.9	1.3	16.3	17.4	96	2841	19
1991	6.6	9.8	2.9	9.7	1.3	16.7	17.8	101	2827	19

Countries on the European continent



pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
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Countries on the European continent

Germany

Black Forrest

1988 Due6	5.1	1.9	0.5	1.2	1.4	2.1	4.7		1324	100
1989 Duerreychbach	5.3	1.2	0.4	0.8	1.3	1.9	3.7	39	1225	50
1990	5.1	1.3	0.4	1.1	1.3	1.9	4.0	35	1175	100
1991	5.2	1.8	0.5	1.1	1.2	1.9	3.5	40	1227	50
1992	5.2					1.9	4.0	38	1272	18
1988 Gol7	6.4	2.0	0.4	1.5	0.5	1.0	4.1		372	100
1989 Goldersbach	6.6	2.1	0.4	1.5	0.3	1.0	4.3	180	433	50
1990	6.6	2.4	0.5	1.5	1.0	1.1	4.6	91	709	100
1991	6.1	2.4	0.7	1.3	1.0	1.0	4.5	82	590	50
1992	6.2	2.1		1.1	0.1	0.9	4.3	88	487	21
1988 Kle1	6.4	2.7	0.8	2.1	1.2	3.6	4.2		890	100
1989 Kleine Kinzig	6.8	2.7	1.0	1.9	1.2	3.3	3.8	186	831	55
1990	6.7	3.0	0.8	1.8	1.2	3.4	3.6	170	850	100
1991	6.7	3.2	1.0	1.7	1.1	3.4	3.6	161	875	50
1992	6.8	3.7	1.1	1.5	1.0	3.5	3.5	188	902	16

East Bavaria

1988 Gro33	5.8	2.0	0.6	1.4	0.6	1.1	4.5		847	49
1989 Grosse Ohe	5.9	2.1	0.7	2.0	0.4	1.0	5.4		932	108
1990	6.0	1.9	0.6	1.6	0.6	2.9	6.4	102	850	45
1991	6.2	1.8	0.6	1.7	0.4	2.8	3.7	90	882	60
1992	6.1					5.0		92	881	48
1987 Hin31	5.7	1.8	0.6			1.0	4.5		1100	
1988 Hintere	6.0	1.7	0.5			1.3	4.0		992	
1989 Schachtenbach	6.1	1.8	0.5			1.0	3.9		1108	
1990	6.1	1.8	0.5			0.9	3.5	451	977	
1991	6.3	1.9	0.5			1.2	3.2	451	1000	
1992	5.8	1.9	0.5			0.7	4.1	394	1020	
1988 See30	5.9	1.6	0.6			1.3	3.8		935	
1989 Seebach	10.6	1.6	0.6			0.8	3.7		1009	
1990	6.1	1.7	0.5			0.9	3.4	507	933	
1991	6.3	1.7	0.5			1.2	3.2	497	919	
1992	5.9	1.7	0.6			0.7	3.9	431	892	
1988 Vor32	6.3	2.2	0.6			1.3	4.7		844	
1989 Vorderer	6.5	2.2	0.6			0.8	4.4		961	
1990 Sachtenbach	6.4	2.2	0.6			1.0	4.1	662	840	
1991	6.5	2.2	0.6			1.2	3.8	677	798	
1992	6.1	2.2	0.6			0.8	4.6	589	849	
1988 Wal2	5.7	5.5	2.3	2.8	0.7	2.6	7.1		1881	23
1989 Waldnaab	6.4	15.0	2.2	2.8	0.8	2.2	6.4	682	1355	28
1990	5.2	3.4	2.0	2.6	0.8	2.6	5.4	112	1508	35
1992	5.6	4.9	1.9	2.4	0.9	1.9	9.5	138	1394	28

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1988 Wal3	5.3	6.9	2.1	4.1	0.9	2.3	11.9		1279	41
1989 Waldnaab	5.3	8.1	2.0	3.2	0.8	2.1	14.0	482	1030	29
1990	5.9	3.6	2.0	3.7	1.0	2.8	9.6	235	1472	24
1992	4.5	3.3	1.8	3.3	0.8	2.2	18.5	13	1223	36
1988 Wal8	5.1	7.0	1.4	3.9	1.2	2.2	17.4		941	46
1989 Waldnaab	4.3	12.1	1.8	3.7	0.6	2.4	19.9	282	805	25
1990	4.7	4.0	1.4	3.9	0.9	2.8	16.6	21	975	28
1992	4.3	3.2	1.5	3.7	0.9	2.4	30.7		971	23
1987 Wal9	5.1	4.7	1.6	3.9	1.0	3.4	18.8	73	6650	409
1988 Waldnaab	5.4	8.5	2.4	3.8	0.9	3.1	16.4		2408	73
1989	4.9	15.1	2.0	3.8	0.9	2.8	19.3	397	1020	51
1990	4.9	3.9	1.6	4.3	1.1	3.5	16.3	40	1400	35
1992	4.4	5.1	1.7	3.9	0.9	2.9	26.0	24	1465	30
1988 Wal14	6.9	15.9	5.1			20.2	18.7		3679	208
1989 Waldnaab	6.4	17.2	5.1	14.5	2.6	20.0	26.8	1756	3920	357
1990	5.3	9.0	5.0	10.4	2.0	15.3	17.5	313	4450	310
1992	6.7	8.7	5.1	12.0	3.2	20.0	26.8	353	4562	278
Fichtelgebirge										
1987 Ege1	5.8	2.6	1.1			7.0	6.1	106	868	
1988 Eger	5.6	2.8	1.1			7.2	6.8	126	783	
1989	5.9	2.5	1.1			8.2	5.2	152	750	
1990	6.3	2.3	1.2			9.1	4.4	92	708	
1991	6.0	2.5	1.2			7.9	5.3	53	796	
1992	5.7	2.6	1.2			8.0	4.8	50	818	
1987 Zin3	4.2	4.7	1.1			13.1	23.1	79	1805	
1988 Zinnbach	4.3	4.4	1.0			2.4	24.5	92	1642	
1989	4.0	4.4	1.0			2.6	24.8	87	1491	
1990	4.0	4.2	1.0			2.7	24.2	49	1592	
1991	4.1	4.2	1.1			2.2	26.2	40	1438	
1992	4.0	4.4	1.0			2.2	22.0	63	1398	
Harz										
1988 Dic	6.4		5.4				21.5	88	3269	89
1989	6.4		4.6			5.1	19.0	102	3191	4
1990	6.5	7.9	4.9	2.7	1.0	5.0	26.9	133	2731	34
1991	6.0	4.6	3.7	2.4	0.8	4.1	23.4	97	1764	18
1992	6.3	6.0	4.6	2.7	0.9	4.5	26.9	125	1941	28
1988 Gro14	5.1		0.7				2.2	63	1907	15
1989 Grosse Bode	5.4	3.2	0.8	3.1	0.6	4.1	5.4	62	992	5
1990	6.0	2.6	0.6	3.5	0.9	2.9	9.8	105	971	121
1991	5.4	1.6	0.8	3.2	0.8	2.5	9.3	58	901	25
1992	5.5	2.5	0.9	3.4	0.9	2.9	10.6	61	977	26
1987 Sch9	6.6	6.1	2.7	2.6	0.6	2.9	17.6	139	1839	18
1988 Grosse Schacht	6.5		2.3				8.0	127	2731	5
1989	6.8	6.6	2.8	2.8	0.7	3.6	13.6	215	1644	10
1990	7.0	6.6	2.9	3.1	0.7	3.8	18.4	220	1572	75
1992	6.9	4.8	2.9	3.0	0.6	3.6	16.9	185	1277	27

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1988 GroS2	5.6		2.4			2.5	12.7		1393	
1989 Grosse Soese	6.5	5.0	2.2	2.3	0.6	3.3	12.4	123	1060	14
1990	6.4	4.5	2.2	2.5	1.0	3.4	14.9	145	1025	25
1991	6.2	3.5	2.0	2.5	0.5	3.8	13.1	82	1014	18
1992	6.3	3.5	2.2	2.6	0.5	3.5	13.8	111	804	29
Hunsrueck										
1987 Gra5	4.3	4.8	2.1	3.2	1.5	8.6	32.6		2727	58
1988 Grafenbach	4.3	5.2	2.0	4.1	0.9	6.3	28.7		559	20
1989	4.3	5.4	2.5	3.8	0.8	6.3	28.8		797	29
1990	4.5	7.1	2.7	4.3	1.0	8.2	29.4	40	1409	17
1991	4.3	6.8	2.9	4.4	1.1	8.1	28.1	40	2114	27
1992	4.3	6.7	3.0	4.1	1.3	6.2	29.7	40	1584	48
1988 Tra1	4.3	2.3	1.4	5.8	0.9	10.1	12.6		763	39
1989 Traunbach	4.6	2.6	1.5	5.0	0.5	10.2	10.8		615	15
1990	4.5	2.1	1.5	4.5	0.4	9.0	10.2	40	696	15
1991	4.3	2.3	1.6	5.0	0.4	10.3	10.1	45	739	17
1992	4.2	2.3	1.8	4.8	0.6	9.4	11.9	40	879	9
Kaufanger Wald										
1988 Nie1	4.1	5.6	3.0	2.6	0.7	5.0	31.3		850	105
1989 Nieste	4.2	6.0	2.8	3.0	1.4	6.9	36.0		2350	100
1990	4.5	5.2	2.5	2.9	1.0	4.8	28.3	92	1547	107
1991	4.4	4.7	2.3	2.9	1.0	5.0	29.4	92	487	100
1992	4.3	4.1	1.9	2.8	1.1	6.5	22.3	87	575	121
1988 Nie3	6.4	11.7	5.1	3.9	1.4	5.8	35.3		1392	109
1989 Nieste	5.8	11.0	4.6	3.8	1.5	6.9	35.5		1680	100
1990	6.0	10.6	4.4	3.5	1.3	4.9	31.4	174	1477	100
1991	6.3	13.3	5.1	3.9	1.6	5.3	37.4	253	1040	100
1992	6.1	11.9	4.3	3.3	1.7	5.8	30.6	249	1211	120
1987 Nie5	3.7	8.3	3.2	1.9	0.9	10.0	44.5		1295	55
1988 Nieste	4.3	9.4	3.3	4.4	1.1	8.5	31.7		1048	144
1989	4.5	9.1	3.2	3.8	1.0	7.7	36.1		1647	132
1990	4.4	9.1	3.7	4.0	1.0	6.6	32.7	92	1363	100
1991	4.7	8.1	2.6	4.4	3.6	8.8	37.6	92	660	118
1992	4.5	8.1	2.6	4.3	1.4	8.5	29.1	88	856	163
Lauenburg										
1987 Pin3	4.5	1.5	0.5	3.5	0.7	8.8	10.5	37	42	174
1988 Pinnsee	4.9	1.7	0.6	3.5	0.7	9.4	9.7	30	53	235
1989	4.7	1.6	0.6	3.7	0.7	8.0	7.9		38	264
1992	4.9	2.4	0.6	4.0	0.8		6.5		38	270
Odenwald										
1987 Sch1	4.4	6.0	2.5	1.9	1.5	6.0	30.0		550	100
1988 Schmerbach	4.2	4.6	2.4	2.2	1.3	4.9	30.9		642	110
1989	4.4	4.3	2.3	2.0	2.0	4.0	22.5		698	100
1990	3.9	4.3	2.3	1.9	1.7	4.5	27.5	92	852	100
1991	4.6	5.7	2.3	2.6	2.0	6.0	22.7	92	627	188
1992	4.7	4.5	1.9	2.3	2.1	5.8	22.7	81	735	166

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1987 Sch3	4.1	4.0	2.0	1.6	1.4	5.0	26.0		1030	100
1988 Schmerbach	3.9	3.1	2.0	2.0	1.3	5.9	30.6		847	102
1989	3.9	4.1	2.5	1.7	1.2	6.0	21.2		1313	115
1990	3.2	3.7	2.7	2.1	1.3	5.8	23.7		1550	113
1991	3.9	4.1	2.7	2.1	1.3	4.7	11.1	92	1650	100
1992	4.2	3.2	2.2	1.8	1.9	7.0	18.5	80	1739	126
Rothaargebirge										
1988 Zin2	6.3	3.6	2.9	2.5	0.9	3.3	14.2		1135	24
1989 Zinsee	6.2	3.9	3.1	3.5	1.0	2.3	16.1		914	23
1990	6.7	3.7	2.7	2.8	1.0	2.6	13.4	198	772	27
1991	6.1	3.9	2.8	2.8	1.0	2.3	14.0		1010	34
1992	6.1	3.6	2.4	2.3	1.0	2.8	12.1	138	880	21
1988 Elb1	8.3	3.5	3.4	2.7	0.8	3.6	15.1		1125	545
1989 Elbendorfer Bach	6.5	4.0	3.6	2.9	1.0	2.5	16.8		906	22
1990	6.8	3.9	3.2	3.0	1.0	2.9	14.4	264	911	21
1991	6.1	3.8	3.5	2.7	1.0	2.7	15.0		1162	33
1992	6.3	3.8	3.0	2.4	1.0	3.0	13.7	144	1009	22
Taunus										
1987 Rom2	4.5	4.5	2.1	4.1	0.4	11.0	20.7		2547	10
1988 Rombach	4.6	5.5	2.1	16.4	0.6	35.2	20.7		1345	107
1989	4.0	4.1	1.9	6.6	0.4	13.0	20.8		3162	103
1990	4.3	4.5	1.7	20.0	0.5	33.9	18.8	92	3062	124
1991	4.9	4.8	1.8	13.4	0.5	27.8	22.7	92	2497	100
1992	5.0	4.5	1.8	5.8	0.5	10.9	17.7	87	2321	111
1988 Rom3	6.4	6.7	2.8	4.2	0.5	6.6	14.0		1753	102
1989 Rombach	6.0	5.1	2.6	2.9	0.6	15.9	12.7		3346	102
1990	6.1	5.4	2.4	2.7	0.6	7.8	10.3	181	1966	114
1991	7.0	7.4	2.3	3.3	0.6	7.6	15.4	355	1975	103
1992	6.6	6.2	2.0	2.9	0.6	7.4	11.2	389	1803	133

The Netherlands

1987 AGA	4.9	2.2	1.1	5.7	1.3	12.0	13.0	14	40	2040
1988 Achterste Goorven /	4.9	2.4	1.1	5.6	1.4	12.3	15.0	18	79	2486
1989	5.4	2.0	1.1	6.6	1.5	14.1	16.0	58	68	3050
1990	5.3	3.0	1.8	8.5	1.6	15.7	24.2	20	132	3854
1991	4.8	4.2	2.1	8.6	1.4	15.9	28.0	15	90	4140
1992	4.6	4.3	1.8	7.2	2.4	13.6	24.8	14	78	2420
1987 AGB	4.9	2.0	1.2	5.8	1.3	12.0	13.5	5	45	1690
1988 Achterste Goorven E	5.1	2.0	1.1	5.4	1.5	12.1	14.0	23	40	2622
1989	5.6	1.6	1.1	6.7	1.7	14.0	17.5	59	96	4079
1990	5.1	3.6	2.1	8.3	1.9	15.5	33.8	22	86	5113
1991	4.6	5.1	2.4	8.3	1.4	15.0	38.6	0	150	4920
1992	4.5	4.0	1.9	7.2	1.3	13.7	27.6	10	83	2625
1987 AGE	4.5	2.0	1.2	5.7	1.3	11.6	14.0	0	57	1437
1988 Achterste Goorven E	5.0	1.9	1.1	5.4	1.6	12.1	13.5	23	45	2545
1989	5.7	2.1	1.3	6.9	1.8	14.1	19.3	105	57	4545
1990	5.3	3.5	2.1	8.4	1.8	15.3	33.4	56	86	5377
1991	4.4	6.5	2.8	8.3	1.4	14.8	49.4	0	120	5420
1992	4.0	4.9	2.0	7.1	1.1	12.6	33.9	1	90	2160

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1987 GER	4.5	0.9	0.6	4.1	1.0	7.9	9.8	0	119	1515
1988 Gerritsfles	4.3	1.0	0.6	3.1	0.8	6.5	8.5	0	96	855
1989	4.3	1.2	0.7	4.3	0.8	7.9	9.2	0	134	813
1990	4.5	1.4	0.8	5.5	1.0	10.4	10.2	0	112	1386
1991	4.4	1.9	0.9	5.7	0.8	10.3	10.2	0	170	886
1992	4.4	1.5	0.8	5.0	1.0	9.2	9.1	3	143	800
1987 KLI	5.5	1.1	0.7	5.2	1.5	9.9	5.5	26	45	1006
1988 Kliplo	5.5	1.0	0.6	4.3	1.4	9.8	4.3	19	34	892
1989	5.7	0.9	0.6	6.3	2.1	11.8	7.0	19	79	1317
1990	4.8	1.5	1.0	7.3	2.1	14.5	9.8	5	68	1742
1991	5.0	1.7	1.0	8.7	2.0	15.8	8.6	1	90	1430
1992	5.3	1.9	0.9	7.8	3.7	15.3	8.4	57	80	1325

Belgium

Eupen

1988 B01	4.4	2.5	1.0	3.0	1.1	3.3	11.8		654	137
1989 Vesdre. Dam	4.4	2.9	0.9	3.1	1.0	3.9	13.6		680	122
1990	4.4	2.9	1.0	3.9	1.0	4.3	14.5		571	79
1991	4.5	3.2	0.9	3.8	1.0	6.3	14.8		626	92
1988 B02	4.6	4.0	1.1	3.1	1.0	3.2	14.4		724	71
1989 Vesdre River	4.7	4.1	1.3	3.0	1.0	4.6	16.3		691	81
1990 Bellesfort	4.7	4.2	1.3	4.0	1.0	4.7	17.6		553	31
1991	4.8	4.3	1.3	3.2	1.0	5.5	17.8		621	16
1988 B03	4.4	3.0	1.0	3.4	1.0	3.8	11.1		362	67
1989 Getz River. Dam	4.5	3.3	1.4	4.0	1.0	5.3	14.7		458	77
1990	4.5	3.2	1.0	5.2	1.2	5.4	14.6		372	30
1991	4.5	3.5	1.0	4.2	1.1	6.6	14.1		353	13

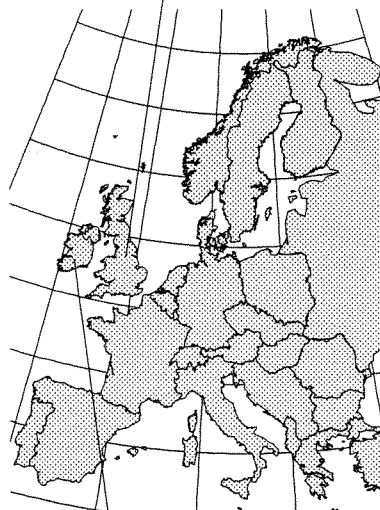
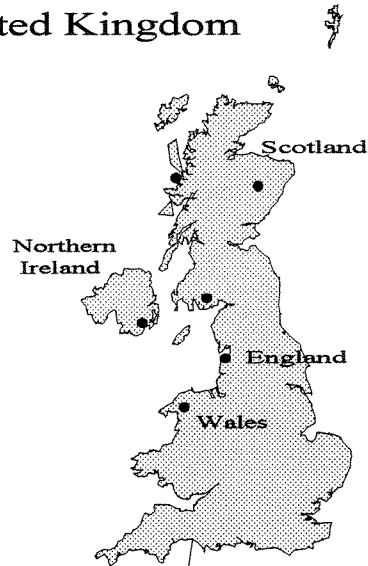
Jalhay

1988 B04	4.2	2.2	1.0	2.5	1.0	2.8	11.2		475	73
1989 Helle River	4.3	2.8	1.3	2.6	1.6	3.3	14.1		571	62
1990 Schornstein	4.2	2.3	1.0	3.4	1.3	3.5	13.4		466	26
1991	4.3	2.9	1.2	3.1	1.0	5.8	14.1		511	20
1988 B05	4.8	4.3	1.2	6.8	1.1	9.0	12.8		1034	130
1989 Lake Gileppe	4.8	4.2	1.4	5.2	1.0	7.1	14.6		1014	95
1990	4.7	3.9	1.4	5.8	1.0	6.1	15.7		884	61
1991	4.7	4.4	1.3	5.7	1.0	9.2	15.3		931	60
1988 B06	4.6	3.9	1.3	6.7	1.0	9.2	12.0		624	90
1989 Gileppe River.	4.7	3.7	1.3	7.6	1.1	9.4	15.4		673	53
1990 Chemin des charb.	4.6	3.5	1.3	7.6	1.0	8.3	15.5		598	35
1991	4.8	4.2	1.4	8.0	1.0	14.0	15.8		653	24
1988 B07	6.3	5.9	2.0	7.7	1.0	9.8	14.3		1022	77
1989 Les Hes	6.4	6.2	1.9	7.2	1.1	8.8	16.3		1037	69
1990	6.4	6.4	2.0	8.2	1.1	9.0	16.8		1022	27
1991	6.3	6.6	1.9	8.5	1.1	13.1	15.6		2796	21
1988 B08	4.1	3.1	1.3	2.4	1.3	3.0	13.1		603	106
1989 Soor River. Dam	4.3	3.2	1.1	2.5	1.1	3.1	16.0		702	74
1990	4.3	3.0	1.1	3.6	1.0	3.7	16.1		629	36
1991	4.3	3.3	1.1	2.9	1.0	5.6	16.3		685	31

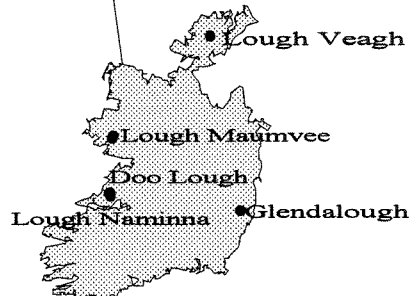
	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
Austria										
Kaernten										
1989 DOS-K	6.3	2.8	0.1	0.3	0.3	0.1	3.2	26	287	13
1990 Dosener See	6.6	2.5	0.2	0.5	0.1	0.1	2.0	37	359	15
1991	7.4	2.0	1.2	0.3	0.3	0.1	1.6	38	324	2
1988 GRA-K	7.5	8.0	1.9	2.6	0.8	0.2	14.8	278	277	8
1990 Gradenbach	7.5	13.6	3.2	1.2	1.2	0.5	15.7	552	387	5
1991	7.5	20.2	4.4	1.1	1.3	0.2	37.0	1	303	11
1989 HAI-T	5.2	1.5	0.1	0.1	0.2	0.1	1.0	1	300	3
1990 Hairlacher See	5.3	0.4	0.1	0.1	0.1	0.1	0.9	6	289	4
1991	5.4	0.6	0.1	0.1	0.2	0.1	0.9	-3	314	6
1990 MEL-K	7.5	9.0	1.3	0.8	1.7	0.6	5.1	464	299	18
1991 Melniksee	7.4	6.6	1.8	0.3	1.3	0.1	2.5	342	204	4
1988 WAN-K	7.3	5.1	1.0	0.9	1.1	0.2	8.3	233	271	10
1989 Wangenitzbach	7.1	6.7	1.2	0.9	1.3	0.8	11.0	265	438	12
1990	7.4	11.4	2.8	1.2	1.3	0.5	13.6	490	332	6
1991	7.3	6.6	2.1	1.3	1.3	0.2	10.0	306	334	14
Tirol										
1989 GKS-T	7.2	4.2	0.2	0.4	0.2	0.7	2.5	92	301	4
1990 Gossenkoelle See	6.9	3.0	0.2	0.3	0.2	0.3	2.4	96	299	4
1991	6.9	2.9	0.2	0.4	0.2	0.1	2.6	86	335	3
1989 MUT-T	5.1	1.6	0.3	0.2	0.2	0.5	1.4	8	194	17
1990 Mutterberger See	5.2	0.5	0.1	0.2	0.2	1.1	0.2	3	183	6
1991	5.4	0.4	0.1	0.1	0.2	0.1	1.1	-4	196	9
1989 PIB-T	7.5	8.1	1.9	1.8	0.8	0.3	6.4	316	467	13
1990 Piburger Bach	7.4	8.2	1.8	1.7	0.9	0.6	6.3	325	527	7
1989 SOS-T	5.4	2.1	0.2	0.3	0.1	0.2	2.6	24	150	5
1990 Schwarzee	5.7	0.9	0.1	0.3	0.1	0.2	2.5	9	106	7
1991 ob Soelden	5.5	1.1	0.2	0.3	0.1	0.2	2.8	2	143	21
Upperaustria										
1989 KLA1-0	6.5	3.7	0.7	1.3	0.8	0.8	3.7	48	1123	22
1990 Klafferbach,	6.4	1.9	0.6	1.4	0.7	1.3	3.6	41	1125	10
1991 Rehberg	6.4	3.3	0.7	1.6	0.8	1.0	3.7	58	1133	11
1989 KLA2-0	7.0	5.1	1.0	1.8	0.6	1.2	4.3	115	1267	13
1990 Stingbach	6.9	3.5	0.9	2.0	0.5	1.1	4.2	127	1180	14
1991 vor Klafferbach	6.9	5.5	1.1	1.9	0.5	1.0	5.0	134	1283	9
1989 KLA3-0	6.7	4.0	0.8	1.6	0.6	1.4	3.4	67	1215	19
1990 Klafferbach Brucke	6.6	2.5	0.7	1.6	1.0	1.0	3.5	81	1177	17
1991	6.8	3.8	0.8	1.6	0.7	0.9	3.7	86	1207	5

UK and Ireland

United Kingdom



Ireland



pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
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UK and Ireland

UK

Scotland

1988 UK01	6.4	0.8	0.6	4.5	0.4	6.8	1.8	75	66
1989 Loch Coire nan Arr	6.4	1.0	1.0	6.6	0.4	12.1	2.1	63	37
1990	6.3	0.8	0.8	5.7	0.4	10.1	2.0	50	31
1991	6.5	0.8	0.7	4.8	0.3	8.3	2.1	67	48
1992	6.4	0.7	0.6	4.7	0.3	8.4	1.7	52	27
1993	6.1	1.1	1.3	9.1	0.5	18.3	2.5	28	42

1988 UK04	5.4	0.5	0.3	1.7	0.4	1.8	2.6	2	245
1989 Lochnagar	5.4	0.7	0.5	2.5	0.4	3.8	3.2	3	143
1990	5.3	0.6	0.4	2.0	0.4	2.9	2.8	0	153
1991	5.4	0.5	0.4	1.9	0.3	2.9	2.8	-1	196
1992	5.6	0.6	0.4	2.1	0.3	3.0	2.8	9	135
1993	5.0	0.5	0.4	2.6	0.1	4.5	2.8	-12	294

1988 UK07	4.9	0.7	0.5	3.3	0.4	5.3	2.5	-21	60
1989 Round of Glensheac	4.9	0.8	0.6	4.8	0.4	8.8	3.2	-20	62
1990	4.8	0.7	0.6	5.1	0.4	9.5	3.3	-21	56
1991	5.0	0.7	0.7	4.8	0.4	8.4	3.8	-20	95
1992	4.9	0.6	0.4	3.7	0.3	6.3	3.2	-20	70
1993	4.9	0.7	0.6	4.8	0.4	8.7	3.5	-17	84

England

1988 UK10	4.8	0.6	0.5	3.4	0.5	5.6	2.3	-21	261
1989 Scoat Tarn	5.0	0.7	0.5	3.8	0.4	6.6	2.8	-9	265
1990	4.9	0.8	0.7	5.1	0.4	9.4	3.2	-17	269
1991	4.8	0.8	0.7	4.5	0.3	8.1	3.2	-26	414
1992	5.0	0.6	0.6	3.5	0.3	6.1	2.9	-14	265
1993	5.0	0.7	0.7	4.4	0.3	8.6	2.9	-15	308

Wales

1988 UK15	5.3	0.9	0.4	2.8	0.4	4.6	2.3	52	75
1989 Llyn Lagi	5.4	1.1	0.6	4.1	0.5	7.3	2.9	60	190
1990	5.2	1.5	0.8	5.5	0.4	10.7	3.2	49	176
1991	5.3	1.2	0.6	4.5	0.2	8.2	3.4	41	97
1992	5.2	1.0	0.5	3.7	0.2	6.4	2.9	36	184
1993	5.1	1.0	0.6	4.4	0.2	8.5	2.8	-6	170

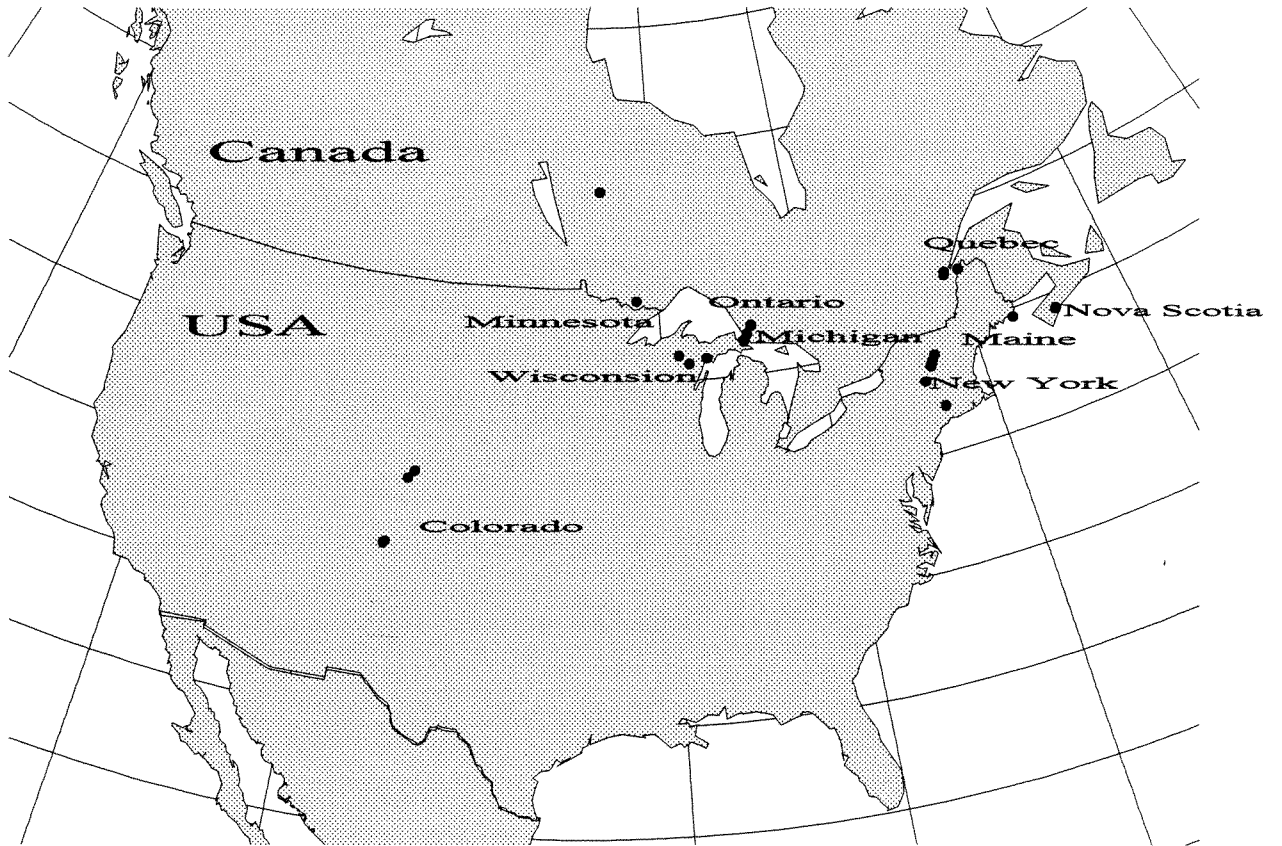
North Ireland

1990 UK21	4.7	0.9	0.8	6.5	0.4	11.4	4.9	-41	299
1991 Blue Loch	4.7	2.6	0.7	6.0	0.4	9.7	5.4	-48	373
1992	4.7	0.7	0.6	5.3	0.4	8.6	4.5	-36	280
1993	4.7	0.9	0.9	7.0	0.5	12.6	4.6	-38	280

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
Ireland										
1987 DOO10	6.7	3.5	2.0	11.7		21.8	2.5	208	111	27
1988 Doo Lough	6.4	3.8	2.4	13.4		23.9	1.5	194	84	19
1989 Mid Lake	6.3	3.1	1.9	11.8		21.3	1.8	222	82	38
1987 DOO11	5.4	1.8	1.5	9.9		19.7	1.0	144	127	15
1988 Doo Lough	5.3	2.3	2.4	16.2		27.8	2.5	126	73	52
1989 Inflow 1	5.5	2.6	2.2	13.0		24.0	1.8	232	122	51
1987 DOO12	5.0	1.8	1.6	11.1		19.0	2.4	142	84	30
1988 Doo Lough	4.9	2.3	2.5	16.6		30.3	1.6	120	61	64
1989 Inflow 2	4.9	2.0	2.2	14.2		27.1	2.0	138	131	147
1987 DOO13	6.1	2.8	1.8	9.4		17.7	0.8	204	109	26
1988 Doo Lough	5.9	3.0	2.8	15.8		28.0	1.7	210	64	54
1989 Inflow 3	5.6	2.5	2.2	12.0		22.1	1.9	216	114	72
1987 GLE10	5.9	1.8	0.8	3.6		8.5	3.0	124	406	23
1988 Glendalough	5.9	2.0	0.8	4.2		6.8	2.3	124	310	25
1989 Mid Lake	5.7	1.9	0.8	3.9		6.8	2.9	136	249	21
1990	5.1							120	280	20
1991	7.0	1.6	1.0	5.8	3.0			80	360	20
1992	6.2	1.6	0.7	5.4	0.8	6.6	3.8		210	23
1993	6.2							80	200	20
1987 GLE11	6.2	2.8	0.9	3.5		7.6	3.8	172	172	7
1988 Glendalough	6.2	3.1	0.9	4.0		6.2	3.9	168	152	14
1989 Inflow 1	5.9	2.4	0.9	3.6		6.3	4.1	172	132	10
1990	5.8							140	200	10
1991	6.5	2.8	1.0	4.8	0.8			140	240	5
1992	6.7	2.6	0.7	4.8	0.4	5.0	4.5		110	8
1993	6.6							120	220	5
1987 GLE12	6.6	4.4	1.4	4.1		8.6	6.6	230	561	10
1988 Glendalough	6.2	4.8	1.2	4.3		6.8	5.0	222	365	11
1989 Inflow 2	6.3	3.9	1.3	4.7		8.4	5.4	238	184	9
1990	6.3							260	230	10
1991	6.3	4.0	1.5	0.8	0.9			180	240	5
1992	7.0	5.2	1.4	6.4	0.5	7.3	8.2		200	5
1993	7.1							240	220	5
1987 GLE13	4.9	1.6	1.0	4.8		10.3	2.0	108	1034	7
1988 Glendalough	5.0	1.9	0.9	5.0		8.5	2.1	104	893	14
1989 Inflow 3	4.8	1.0	0.9	5.0		9.3	2.5	88	761	7
1990	4.4							100	760	30
1991	3.8	1.1	1.0	5.7	0.8			40	700	5
1992	5.2	0.8	0.8	7.1	0.3	9.4	3.4		390	8
1993	5.1							60	460	1
1987 MAU10	6.3	1.5	1.0	9.6		13.9	1.5	124	64	6
1988 Lough Maumwee	6.0	2.1	2.0	13.2		21.9	1.2	116	31	14
1989 Mid Lake	5.9	1.6	1.4	9.7		17.3	1.2	142	38	15
1991	5.0							100	60	15
1992	6.4	1.4	0.9	6.1	0.2	12.9	2.6	140	20	15
1993	6.4	1.4	0.6	6.1	0.5	10.0	2.8	160	20	25

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1987 MAU11	6.4	3.0	1.3	7.9		13.7	1.4	212	82	17
1988 Lough Mumwee	6.1	4.8	2.3	13.9		20.2	1.1	244	105	13
1989 Inflow 1	6.0	3.3	1.6	9.7		17.3	2.0	250	92	15
1991	5.5							220	100	10
1992	7.2	5.1	1.5	9.4	0.2	14.9	4.6	320	140	15
1993	6.8	7.3	1.4	9.5	1.1	14.9	3.6	380	100	10
1987 MAU12	6.3	2.1	1.4	9.0		15.8	0.7	164	36	14
1988 Lough Maumwee	6.1	2.7	2.3	15.7		23.4	1.1	166	49	25
1989 Inflow 2	5.9	2.1	1.7	11.1		20.1	1.6	190	56	28
1991	5.5							180	60	20
1992	6.9	4.6	1.4	7.2		15.9	3.1	280	40	15
1993	6.7	5.0	1.4	9.3	0.5	16.1	3.0	360	100	65
1987 NAM10	6.2	2.0	1.7	10.1		19.8	1.7	144	52	10
1988 Loch Naminna	6.1	1.9	1.6	11.4		20.5	2.3	116	37	11
1989	5.8	1.7	1.7	10.9		19.9	1.7	138	54	18
1987 NAM11	5.8	2.2	1.8	10.0		18.9	1.6	174	43	9
1988 Loch Naminna	5.7	3.0	3.1	15.4		29.2	0.8	164	31	19
1989	5.5	3.5	2.6	12.8		23.2	2.0	226	29	24
1988 VEA10	5.9	1.7	1.5	10.5		19.0	1.0	126	43	27
1989 Loch Veagh	5.8	1.3	1.4	9.4		17.2	1.0	126	38	23
1991 Mid Lake	5.4	1.4	1.4	10.4	4.3			100	60	20
1992	6.2	1.1	0.9	11.0	0.3	13.7	2.8	60	60	20
1993	6.0	1.6	1.3	9.3	0.9	16.2	2.8	160	40	60
1988 VEA11	6.1	2.7	2.6	18.9		23.0	1.3	168	45	16
1989 Loch Veagh	6.2	2.0	1.7	10.6		17.7	1.6	224	51	18
1991 Inflow 1	6.2	1.3	1.3	10.9	7.9			140	20	10
1992	6.8	2.3	1.6	14.3	0.3			100	140	10
1993	6.4	1.9	1.4	10.7	0.3	15.9	2.8	220	40	20
1988 VEA12	5.8	2.6	2.6	16.6		20.8	2.1	158	52	15
1989 Loch Veagh	5.7	1.9	1.9	10.8		21.1	1.7	146	69	14
1991 Inflow2	5.3	1.3	1.2	9.7	0.8			120	40	10
1992	6.8	1.4	1.2	12.8	0.3	15.6	3.6	80	140	10
1993	6.1	1.1	1.0	7.8	0.6	13.0	2.5	140	20	15
1989 Loch Veagh	6.2	2.8	1.9	10.6		19.7	1.5	240	66	15
1991 Inflow 3	5.6	2.2	1.5	10.4	1.4			180	60	5
1992	6.9	3.1	1.5	13.4	0.4	14.0	4.9	200	180	20
1993	6.4	2.6	1.4	7.9	0.6	15.5	3.0	220	40	15
1988 VEA14	6.6	4.3	3.2	13.8		24.1	0.9	292	173	14
1989 Loch Veagh	6.5	4.3	2.8	11.1		21.4	1.3	350	173	14
1991 Inflow 4	5.9	2.9	2.1	12.7	1.2			220	120	5
1992	7.4	4.5	2.8	14.3	0.5	17.7	4.0	400	260	10
1993	6.7	4.1	2.3	10.1	0.6	17.0	3.3	300	180	20

North America



pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
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North America

Canada

Nova Scotia

1987 C10	4.7	0.3	0.3	2.3	0.3	3.7	1.8	-20	10	
1988 Mount Tom Lake	4.7	0.3	0.3	2.3	0.2	3.5	1.5	-10	10	
1989	4.7	0.3	0.3	2.2	0.3	3.1	2.1	-10	10	
1990	4.7	0.3	0.3	2.3	0.3	3	2	-15	35	
1991	4.6	0.2	0.3	2.2	0.3	2.9	1.6	-20	10	
1992	4.8	0.3	0.5	2.2	0.3	3.3	1.3	-10	10	
1987 C11	5.3	0.4	0.3	2.5	0.2	4.1	2.2	0	10	15
1988 Mountain Lake	5.2	0.4	0.3	2.5	0.2	3.5	2.3	3	13	6
1989	5.3	0.4	0.3	2.3	0.2	3.5	2.3	0	10	6
1990	5.3	0.4	0.3	2.5	0.2	3.3	2.3	5	20	30
1991	5.2	0.4	0.3	2.4	0.2	3.2	2.5	0	10	11
1992	5.3	0.4	0.4	2.4	0.2	3.6	1.9	-10	30	
										54
1987 C12	4.4	0.4	0.4	2.8	0.3	3.8	2.8	-50	10	
1988 Little Red lake	4.4	0.3	0.4	2.4	0.2	3.4	2.1	-40	17	64
1989	4.4	0.4	0.4	2.8	0.3	3.4	2.3	-35	10	20
1990	4.4	0.3	0.4	2.7	0.3	3.2	2	-45	35	24
1991	4.4	0.3	0.4	2.7	0.4	3.5	1.9	-30	10	10
1992	4.5	0.4	0.4	2.4	0.2	3.7	1.6	-30	10	
										55
1987 C13	5.2	0.6	0.4	2.8	0.3	4.7	2.3	-5	10	54
1988 Kejimkujik Lake	4.9	0.7	0.5	3	0.3	4.2	2.4	0	10	
1989	5.2	0.6	0.4	2.8	0.3	4.1	2.7	5	10	31
1990	5.3	0.6	0.4	2.9	0.3	3.9	2.4	13	10	43
1991	5.1	0.6	0.4	2.8	0.3	4.3	2.4	5	10	32
1992	5.1	0.6	0.4	2.8	0.3	4.6	1.4	-10	10	46
										40
1987 C14	5.6	0.4	0.3	2.5	0.2	4.1	2.1	8	10	51
1988 Beaverskin Lake	5.5	0.4	0.4	2.4	0.2	3.9	2.2	10	10	45
1989	5.4	0.4	0.3	2.4	0.2	3.5	2.4	0	10	50
1990	5.4	0.3	0.3	2.5	0.2	3.4	2.4	5	10	52
1991	5.8	1.6	0.6	4.2	0.4	4.9	2.8	75	10	
1992	5.6	0.4	0.4	2.5	0.2	4	1.7	10	10	3379
										2877
Quebec										
1987 C05	6.1	1.3	0.2	0.4	0.2	0.3	3.4	25	35	19
1988 Lac Veilleux	6.1	1.2	0.2	0.3	0.2	0.3	3.3	23	35	11
1989	5.9	1.0	0.2	0.4	0.1	0.2	3.1	18	45	
1990	5.9	1.2	0.2	0.4	0.2	0.3	3.1	12	57	16
1991	6.1	1.2	0.2	0.4	0.2	0.3	3.0	23	45	14
1987 C06	5.9	1.2	0.2	0.4	0.2	0.3	3.9	11	35	27
1988 Lac Josselin	5.9	1.2	0.2	0.4	0.2	0.3	3.6	12	35	14
1989	5.8	1.2	0.2	0.3	0.1	0.2	3.4	13	40	9
1990	5.5	1.3	0.2	0.4	0.2	0.3	3.9	6	100	
1991	5.9	1.2	0.2	0.4	0.1	0.2	3.3	13	35	31
										17

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1987 C07	5.2	0.9	0.2	0.4	0.1	0.3	4.0	0	40	
1988 Lac Bonneville	5.2	0.9	0.3	0.4	0.1	0.3	4.1	0	58	
1989	5.1	0.9	0.2	0.3	0.1	0.3	4.3	0	55	
1990	4.9	0.9	0.3	0.4	0.2	0.3	4.3	-8	85	
1991	5.3	1.0	0.2	0.4	0.1	0.3	3.7	6	53	
1987 C08	6.7	2.2	0.5	0.9	0.1	0.3	4.3	94	59	
1988 Laflamme Lake	6.6	2.3	0.5	0.9	0.2	0.4	4.4	101	73	
1989	6.5	2.4	0.5	0.9	0.2	0.4	4.3	100	99	
1990	6.2	2.4	0.5	1.0	0.2	0.5	4.2	92	108	
1991	6.4	2.3	0.5	1.0	0.2	0.4	3.7	102	88	
1987 C09	5.3	1.0	0.3	0.4	0.1	0.4	4.0	5	10	
1988 Lac MacLeod	5.4	1.0	0.3	0.4	0.1	0.3	4.0	8	10	
1989	5.8	1.0	0.3	0.4	0.1	0.3	4.1	7	15	
1990	5.2	1.1	0.2	0.4	0.2	0.4	3.8	5	53	
1991	5.6	1.0	0.2	0.4	0.1	0.3	3.3	10	15	
Ontario. Algoma region										
1987 C01	6.1	2.8	0.4	0.5	0.2	0.3	5.4	47	220	
1988 Batchwana Lake	5.9	2.7	0.4	0.5	0.2	0.3	5.6	35	296	
1989	6.0	2.6	0.4	0.5	0.2	0.3	5.4	42	267	
1990	6.0	2.7	0.4	0.5	0.2	0.4	5.3	45	271	
1991	5.9	2.5	0.4	0.5	0.2	0.3	5.4	37	244	
1987 C02	6.7	4.2	0.5	0.6	0.2	0.3	5.5	106	412	
1988 Wishart Lake	6.5	3.9	0.5	0.5	0.2	0.3	5.7	86	439	
1989	6.6	3.8	0.4	0.5	0.2	0.3	5.6	84	436	
1990	6.6	3.9	0.5	0.5	0.2	0.3	5.4	92	432	
1991	6.5	3.8	0.4	0.6	0.2	0.3	5.5	80	441	
1987 C03	6.9	5.0	0.5	0.6	0.2	0.3	5.9	147	344	
1988 Little Turkey Lake	6.6	4.8	0.5	0.6	0.2	0.3	6.0	129	389	
1989	6.7	4.6	0.5	0.5	0.2	0.3	5.9	125	378	
1990	6.7	4.7	0.5	0.6	0.2	0.3	5.7	133	368	
1991	6.7	4.6	0.5	0.6	0.2	0.3	5.8	125	391	
1987 C04	6.9	5.8	0.5	0.6	0.2	0.3	6.1	186	293	
1988 Turkey Lake	6.7	5.6	0.5	0.6	0.2	0.3	6.0	173	367	
1989	6.8	5.5	0.5	0.6	0.2	0.3	6.0	174	338	
1990	6.8	5.6	0.5	0.6	0.2	0.3	5.8	180	343	
1991	6.8	5.6	0.5	0.6	0.2	0.3	5.8	174	331	
Ontario										
1987 Ca111	6.4	6.2	0.9	0.9	0.5	0.3	2.6	772	135	
1988 Lake 111	6.4	5.2	0.9	0.7	0.6	0.3	2.6	881	94	
1989	6.4	5.9	0.9	0.8	0.6	0.3	2.5	888	46	
1987 CA224	6.5	2.1	0.5	0.8	0.4	0.2	3.5	87	17	
1988 Lake 224	6.5	2.0	0.5	0.7	0.4	0.2	3.5	91	19	
1989	6.6	2.0	0.5	0.7	0.4	0.2	3.4	97	15	
1987 CA239	6.7	3.4	1.0	1.4	0.8	0.5	5.3	154	52	
1988 Lake 239	6.6	3.4	1.0	1.3	0.8	0.5	5.3	158	48	
1989	6.6	3.2	1.0	1.3	0.8	0.4	5.2	164	46	

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1987 CA305	6.7	2.7	0.7	1.4	0.4	0.3	3.9	140	20	
1988 Lake 305	6.7	2.7	0.7	1.3	0.4	0.3	3.8	145	15	
1989	6.8	2.6	0.7	1.2	0.4	0.2	3.8	142	23	

USA

Maine

1987 1E1132	5.9	0.9	0.4	2.3	0.3	3.2	3.5	11	3	
1988 Little Long Pond	5.7	0.9	0.4	2.4	0.3	3.1	3.6	13		
1989	5.5	0.8	0.3	2.3	0.3	2.8	3.7	11	10	
1990	5.6	0.9	0.3	2.1	0.3	2.8	3.7	13	13	
1987 1E1133	6.4	1.3	0.4	2.3	0.2	2.8	2.6	48	31	
1988 Tilden Pond	6.3	1.3	0.4	2.3	0.3	2.7	2.8	49	21	
1989	6.0	1.2	0.4	2.3	0.3	2.5	2.8	48	10	
1990	6.1	1.2	0.4	2.2	0.3	2.5	2.9	27		

New York, Adirondack Mt.

1987 1A1017	5.0	1.9	0.3	0.6	0.4	0.3	6.7	-13	475	183
1988 Constable Lake	5.2	2.2	0.4	0.8	0.5	0.4	6.7	-4	451	119
1989	4.9	2.0	0.3	0.7	0.4	0.4	6.7	-9	483	129
1990	5.0	2.0	0.3	0.6	0.5	0.4	6.0	-2	690	132
1987 1A1052	6.5	3.2	0.6	0.9	0.4	0.4	6.6	70	178	89
1988 Arbutus Lake	6.6	3.5	0.6	1.0	0.4	0.4	6.6	79	165	91
1989	6.6	3.6	0.6	1.0	0.4	0.4	6.9	80	198	116
1990	6.5	3.2	0.6	0.9	0.4	0.4	6.3	69	236	179
1987 1A1102	6.3	2.4	0.3	0.6	0.2	0.3	5.1	29	225	125
1988 Heart Lake	6.3	2.3	0.3	0.6	0.2	0.3	4.8	32	148	120
1989	6.3	2.4	0.3	0.6	0.2	0.3	5.1	40	185	153
1990	6.4	2.3	0.3	0.5	0.2	0.3	4.5	40	223	126
1987 1A1106	5.4	2.0	0.3	0.7	0.4	0.4	6.0	-5	385	94
1988 Dart Lake	5.5	2.1	0.4	0.7	0.4	0.4	6.0	-2	475	148
1989	5.3	2.0	0.3	0.7	0.4	0.4	6.2	-2	449	120
1990	5.2	2.0	0.3	0.6	0.4	0.4	5.6	1	604	161
1987 1A1109	6.4	2.9	0.6	1.0	0.5	0.4	6.2	59	427	104
1988 Moss Lake	6.6	3.2	0.6	1.1	0.5	0.5	6.3	69	464	93
1989	6.3	3.2	0.6	1.0	0.5	0.5	6.3	70	410	96
1990	6.2	2.8	0.5	0.9	0.5	0.5	5.8	56	591	98
1987 1A1110	6.2	2.3	0.5	0.8	0.4	0.4	5.9	23	350	232
1988 Rondaxe Lake	6.2	2.4	0.4	0.9	0.5	0.4	5.9	36	405	135
1989	5.9	2.4	0.4	0.8	0.5	0.4	6.0	26	427	169
1990	5.9	2.3	0.4	0.8	0.5	0.4	5.5	27	532	141
1987 1A2078	5.6	1.7	0.4	0.7	0.3	0.4	6.2	20	263	57
1988 Otter Lake	5.6	1.8	0.4	0.8	0.3	0.4	6.2	1	265	48
1989	5.1	1.7	0.4	0.8	0.3	0.4	6.5	-9	345	101
1990	5.4	1.7	0.4	0.8	0.3	0.4	5.7	5	497	100

New York, Catskill Mt.

1987 143010	5.1	1.7	0.6	0.4	0.4	0.5	5.7	-5	389	
1988 East Branch Neversir	5.2	1.5	0.6	0.4	0.3	0.6	5.9	-5	305	
1989	5.2	1.5	0.6	0.4	0.3	0.7	5.7	-2	438	

	pH	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	Cl mg/L	SO4 mg/L	ALK-E ueq/L	NO3N µg N/L	NH4N µg N/L
1987 143105	6.2	4.3	0.5	0.3	0.3	0.5	6.1	92	753	
1988 High Falls Brook	6.7	4.9	0.7	0.4	0.3	0.5	6.6	127	419	
1989	6.4	4.2	0.6	0.3	0.3	0.6	6.4	107	487	
Michigan										
1988 2B1047	4.8	1.3	0.3	0.1	0.3	0.4	5.8	-13	55	
1989 Johnson Lake	4.9	1.4	0.4	0.1	0.2	0.2	5.9	-17	74	81
1987 2B2102	7.0	3.2	0.9	0.5	0.5	0.2	3.3	167		213
1988 Buckeye Lake	7.3	3.2	0.9	0.5	0.5	0.4	3.5	172	53	183
1989	7.6	3.3	0.8	0.5	0.5	0.3	3.3	180	71	235
1987 2B3082	5.6	1.4	0.5	0.5	0.4	0.3	4.7	14	18	150
1988 Andrus Lake	6.2	1.4	0.5	0.5	0.4	0.3	4.8	19	41	161
1989	5.7	1.6	0.4	0.5	0.3	0.3	4.6	16	22	168
Minnesota										
1987 2A2063	6.6	1.9	0.7	0.7	0.5	0.3	2.9	121	11	
1988 Cruiser Lake	6.9	1.9	0.7	0.7	0.7	0.2	2.7	112	30	25
1989	6.8	2.0	0.7	0.7	0.5	0.3	2.6	119	70	209
1990		1.2	0.4	0.1	0.2	0.2	5.7		16	102
Wisconsin										
1987 2C1068	5.2	1.6	0.5	0.5	0.5	0.5	7.0	-6	108	249
1988 Sand Lake	5.4	1.6	0.5	0.6	0.6	0.7	6.7	-2	52	170
1989	5.5	1.5	0.5	0.6	0.5	0.8	6.2	-7	57	294
1987 2C1069	5.8	1.3	0.4	0.3	0.4	0.2	2.8	30	35	151
1988 Nichols Lake	5.9	1.3	0.4	0.3	0.4	0.4	3.0	24	78	150
1989	6.0	1.3	0.4	0.3	0.4	0.3	3.0	30	100	220
1987 2C2062	6.1	1.2	0.5	0.4	0.5	0.4	4.1	20	31	93
1988 Luna Lake	6.1	1.2	0.4	0.5	0.7	0.6	4.7	16	8	11
1989	6.2	1.3	0.4	0.5	0.7	0.5	4.1	24	17	247
Colorado										
1987 4E2009	6.9	0.7	0.2	0.5	0.2	0.1	0.5	43		61
1988 Seven Lakes	6.9	0.7	0.2	0.4	0.2	0.1	0.6	42	29	36
1989	7.2	0.8	0.2	0.4	0.2	0.1	0.6	44		22
1990	6.5	0.7	0.2	0.4	0.1	0.1	0.6	43		44
1987 4E2060	7.3	0.9	0.3	0.4	0.1	0.1	0.9	49	92	
1988 Summit Lake	7.4	1.2	0.3	0.4	0.1	0.1	1.1	60	230	
1989	6.8	1.0	0.3	0.4	0.2	0.1	1.0	60	75	
1990	7.1	1.0	0.2	0.4	0.2	0.2	1.0	61	58	
1987 4E2070	6.4	1.1	0.2	0.2	0.1	0.1	1.5	29	83	
1989 Upper Sunlight Lake	7.1	1.4	0.3	0.2	0.1	0.1	1.9	32	96	
1990	7.1	1.7	0.2	0.2	0.1	0.1	2.7	46	29	
1987 4E2071	6.3	0.5	0.1	0.1	0.1	0.1	1.2	5	57	
1988 White Dome Lake	6.1	0.9	0.1	0.1	0.1	0.1	2.0	5	115	
1989	6.1	0.8	0.2	0.1	0.1	0.1	1.9	5	126	
1990	6.2	0.9	0.2	0.1	0.1	0.1	2.4	10	99	

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