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KIRSTI KORKKA-NIEMI

Cumulative geological, regional and site-specific factors affecting groundwater quality in domestic wells in Finland

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Kirsti Korkka-Niemi

Cumulative geological, regional and site-specific factors affecting groundwater quality in domestic wells in Finland

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Preface

Groundwater is playing an increasingly important role in domestic water supplies in Finland. Close to 65 % of the population use groundwater as a source of water for drinking and other domestic purposes, approximately 75 % of these via public water supplies, and the rest, or 16 % of the total population, directly from private wells. This latter category is equivalent to approximately 310,000 households (Korkka-Niemi *et al.* 1993), in addition to whom, most of the estimated 300,000 summer cottages in Finland have private water supplies (Etelämäki 1999). Thus a substantial number of people, who may not be counted as groundwater users, do in fact depend on groundwater for their domestic water for at least some parts of the year. Since the average water consumption in Finnish households is approximately 120–155 litres per capita per day (Etelämäki 1999), it can be estimated that more than 400 million litres of groundwater may be used daily in Finland.

Most of the groundwater used for domestic purposes throughout the country is of high quality as measured by various standards (Kujala-Räty *et al.* 1998), but there are some areas where the groundwater is somewhat sub-standard in quality or is not available in sufficient amounts (Saviranta and Vikman 1990). This affects especially people using private wells. It was estimated in 1990 that approximately 180 000 households, i.e. possibly up to half a million people, had problems of either quality or quantity with their water supply (Anon. 1990).

As part of a long-term strategy for sustainable water supplies and sanitation in Finland, the following general objectives were set for domestic and drinking water (Saviranta and Vikman 1990):

a) The entire population should be supplied with high quality potable water at reasonable cost, and

b) The environmental risks threatening water supplies should be eliminated.

One of the greatest obstacles to the implementation of the above-mentioned long-term strategy and development of the sustainable use of groundwater resources is our present poor understanding of the actual problems on a household scale (Anon. 1990) and our lack of knowledge of the systematic regional distribution of these problems.

The present research is an attempt to provide means of determining groundwater quality and ascertaining the reasons for problems in this respect on the scale of an individual household well while using a large-scale database based on a regional survey of the cumulative geological, regional and site-specific factors that affect groundwater.

List of commonly used symbols and abbreviations

DL	= Detection limit
EC	= Electrical conductivity
FC	= Faecal coliform bacteria
FS	= Faecal streptococci
$KMnO_4$	= $KMnO_4$ consumption
N	= Number of wells
Q1	= Lower quartile (25 %)
Q3	= Upper quartile (75 %)
TC	= Total coliform bacteria
PCA	= Principal Component Analysis

Cumulative geological, regional and site-specific factors affecting groundwater quality in domestic wells in Finland

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Groundwater quality in domestic wells and the reasons for problems in this respect are described on the scale of an individual household well using an extensive database of 1421 wells. The water quality in these wells is compared with that reported in the late 1950's in order to assess long-term changes, and a comparison is made between the dug wells (N = 1096) and the bedrock wells (N = 325). The possibility of seasonal changes is assessed by comparing analyses of water taken from the same 423 wells at three seasons of the year. Only 37.2 % of the wells fulfilled all the hygienic and technical requirements and recommendations for drinking water. Statistical evaluation of the water quality analyses and background data obtained from questionnaires and geological maps points to five water quality factors contributing to the conclusions reached in the survey: salinity, redox, pH, pollution and contamination. These correspond to a combination of specific geological, regional and site-specific factors which together are manifested as cumulative effects operating at particular locations. All the layers of factors are represented to various extents in each well. Well owners can modify the site-specific factors and ameliorate their effects by keeping their wells in a good state of repair, addressing problems of insufficient aeration and eliminating any nearby sources of pollution. Such measures can affect the microbiological quality of the well water, the amounts of nitrogen compounds contained in it, its turbidity, $KMnO_4$ consumption and in part its Al, Fe and Mn concentrations, but it is not possible to influence the state of oxidation in the aquifer (affecting turbidity, colour, $KMnO_4$ consumption, Fe and Mn, SO_4 and NO_3) or any of the extra-regional factors such as the chemical composition of the soil and bedrock or present or relict marine influence (F, Al, SO_4 ,Cl, Na, K, Mn, Fe, alkalinity, pH and total hardness). The seasonal variation in quality variables in individual wells is likely to be greater than the seasonal variation in the aquifers, which emphasises the vulnerability of the wells.

Keywords: groundwater, potable water, drinking water, wells, water quality, water analysis, Finland

1 Introduction and purpose of the research

1.1 Previous groundwater surveys

The first studies of the chemical quality of groundwater in Finland were conducted at the beginning of the last century, when the potable water available from the few existing groundwater intakes was analysed (Lahermo and Parviainen 1979). With the increasing use of groundwater, there has been a growth in the amount of research dealing with occurrence of groundwater and the quality of drinking water in Finland.

There have been a few nation-wide systematic groundwater surveys, two of which are particularly relevant to the present research. The first was carried out more than 40 years ago, in 1958, and its results were published by Wäre (1959, 1960, 1961a, 1961b, 1967). This survey was based exclusively on samples of water from private wells, and involved systematic sampling of over 2700 springs, wells and other water sources, targeting the rural population that was using groundwater for domestic purposes at that time, in that each sampling point represented 1000 inhabitants of rural districts. It is assumed that the data adequately represented the potable water quality in Finnish rural areas. Consequently, the Wäre survey provides a useful comparison with the present results and is used in the present connection to study long-term changes in well water in rural areas. That survey was not designed to provide a regional overview of groundwater quality, however, nor did it allow any analysis to be made of regional or broader factors affecting groundwater quality. Furthermore, if the data from the Wäre survey are to be taken as indicators of groundwater quality, it has to be assumed that the water samples taken from the individual wells faithfully represented the groundwater quality at the locations concerned.

The second nation-wide survey was performed approximately 20 years ago, between 1972 and 1982, by the Geological Survey of Finland (GSF) (Lahermo *et al.* 1990). This was designed to provide a regional overview of groundwater quality rather than a picture of the quality of the water as consumed at the household level. The GSF survey also used well water, presumably on the assumption that it was representative of regional groundwater properties. The sampling grid was relatively even and broad and it is believed that the data adequately represented the hydrogeochemical characteristics of groundwater in Finland. It is assumed in the present research that the hydrogeochemical mapping of groundwater as established by this survey was valid and that a new data set would be likely to improve very little on this broad picture. The survey was not designed to provide a household-scale discriminant analysis however, and nor does it allow any analysis to be made of local or subregional factors affecting groundwater quality. Furthermore, since the density of household wells is much greater in southern Finland than in the north, the northern parts are over-represented and the data do not describe in a statistically acceptable way the quality of the water which Finnish people were drinking in those years. Finally, the sampling was extended over several years and may have taken place at different times of the year, e.g. the spring, summer and autumn, and since these facts have not been elucidated, it must be concluded that the data set incorporates seasonal variations and is not discriminant for that factor. The database does, however, provide a second useful comparative basis for assessing the present results.

Groundwater has also been studied with respect to its chemical composition and microbiological quality in other connections. These previous works are reviewed in the appropriate sections of the thesis, but the data sets resulting from them are described briefly here.

Several summary reports on the occurrence and chemical quality of groundwater in various geological environments have been written on the basis of material gathered by private consultants. Natukka (1960, 1963) described the quality of the groundwater in Finnish eskers based on investigations carried out in 1941–1961, including sampling of 403 municipal wells, while Laakso (1966) described the quality and yield of 1108 wells drilled into bedrock in the period 1951–1963. The quality of bedrock wells was compared with that of dug wells in a national investigation into private wells (Wäre 1960) and/or with the quality of the groundwater in Finnish eskers (Natukka 1960, 1963).

Lahermo (1970, 1971), when studying the geochemistry of groundwater and surface water in Finnish Lapland and in the coastal region of south-eastern Finland, considered the influence of geological factors, notably the mineral composition and grain-size distribution of Quaternary deposits and the composition of the bedrock on the chemical content of natural water. Hyyppä (1973) and Hartikainen (1976) also discussed geological factors affecting groundwater quality.

Pönkkä (1981) described glaciofluvial aquifers in the southern part of Finland and analysed the relationship between the chemical composition of the groundwater in 635 municipal wells and the mode of formation, structure and location of the aquifers. He also discussed the effect of pollution and bedrock type on groundwater quality.

Rönkä (1983) investigated about 700 wells drilled into the Precambrian crystalline bedrock of Finland, describing the chemical composition of groundwater from 400 bedrock wells located mainly in Central Finland, while Soveri (1985), examining input-output balances of water quality in unsaturated and saturated soil zones at 54 groundwater observation stations maintained by the National Board of Waters (nowadays the Finnish Environment Institute), pointed to cyclic changes in groundwater quality.

The assessment made by Mäkelä (1990) of the possibility of using co-operative bedrock wells to supply water to sparsely populated rural areas in Central and Eastern Finland involved the analysis of both the yield and water quality of 27 drilled wells. Meanwhile Backman (1993) studied changes in groundwater quality over the period 1969–1990 on the basis of samples from three springs, five dug wells and one esker pond located in two municipalities in southern Finland. Rajala (1995) examined the effect of geological factors on the occurrence, discharge and water quality of 430 springs in Central Finland, and Korhonen *et al.* (1996) analysed the microbiological, physicochemical and radiochemical quality of the water in 150 rural wells in Finland.

The quality of drinking water supplied by public water works (from 1514 wells) in 1996 was examined on the basis of analyses extracted from the national water quality register by Kujala-Räty *et al.* (1998). These analyses applied to raw water, treated water and water distributed to consumers, and the results were compared with quality standards and target values established by the Ministry of Social Affairs and Health.

Karro (1999), in his study of temporal changes in groundwater chemistry in southern and western Finland, examined long-term monitoring data collected from four municipal groundwater intakes during the years 1956–1995. He also studied water quality in 23 bedrock wells in the Vaasa region.

The short and long-term effects of geology and human activities on the amount and quality of groundwater in 50 monitoring areas designated by the GSF in different parts of Finland were examined by Backman *et al.* (1999) on the basis of water samples taken from springs, dug wells, drilled bedrock wells and observation tubes 4–6 times a year.

The Finnish Environmental Institute has 53 permanent stations where the chemical composition of the groundwater has been observed since 1975 (Soveri *et al.* 2001), and the Geological Survey of Finland collected and analysed more than one thousand water samples from wells all over the Finland in 1999 (Tarvainen *et al.* 2001). Also some local well water studies have been carried out (e.g. Sipilä 2000) in some municipalities of Finland.

1.2 Wells in Finnish aquifers and factors affecting groundwater quality

The Precambrian bedrock of Finland, which is composed of an Archaean basement and Proterozoic complexes, is covered by a relatively thin layer of glacial and postglacial sediments, mainly till, marine or lacustrine sediments and glaciofluvial or littoral gravels and sands varying in thickness from a few metres to some tens of metres (Lahermo *et al.* 1990). According to Okko (1964), the median thickness of unconsolidated sediments in Finland is 6.7 metres.

A Finnish survey to classify potential groundwater areas in 1988–1996 (Britschgi and Gustafsson 1996) placed the total number of these at 7141 and the total estimated yield at 5.83 million m^3d^{-1}. Water supplies which serve 10 or more dwellings use 0.71 million m^3d^{-1}, which is 12 % of this estimated water yield. The average size of one groundwater basin is 2.2 km^2 and the average yield about 1000 m^3d^{-1}. Finnish aquifers are mainly unconfined, but in many cases where the topographically lower parts of an aquifer are covered by fine-grained postglacial marine or lacustrine sediments confined or even artesian groundwater exists.

The groundwater used for municipal water supplies is mainly taken from porous gravel and sand aquifers, i.e. glaciofluvial eskers and related esker deltas and ice-marginal end moraine complexes (the most extensive of which are the Salpausselkä end moraines). On the other hand, private wells in

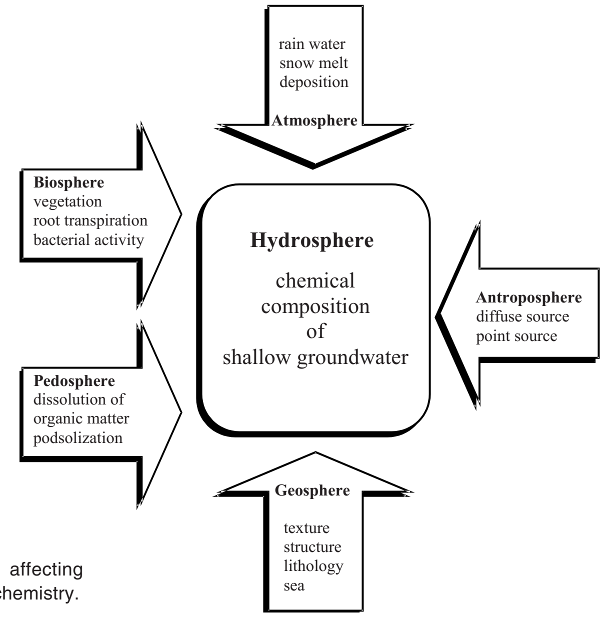


Fig. 1. Main factors affecting shallow groundwater chemistry.

sparsely populated rural areas have usually been dug into glacial till deposits, minor beach terraces composed of gravel and sand, or washed and redeposited till, or else they have been drilled into fractured bedrock. These fractured bedrock aquifers generally have a yield large enough only to supply water to a single household, but there are some places where a group of bedrock wells have been linked together to provide water for a whole rural municipality (Lahermo 1971, Rönkä 1983, Mäkelä 1990, Lahermo *et al.* 1990, Breilin 2000). Littoral deposits on the sides of drumlins have also shown potential as aquifers for utilisation in municipal water supplies (Salmi *et al.* 1990).

Technically, water supplies in the rural areas in Finland are based on natural and captured springs, dug wells or less frequently tube wells discharging water from Quaternary deposits, but wells drilled into bedrock are also common. The springs have either been preserved in their natural state or lined with a shallow wooden casing or concrete ring. The rate of outflow is generally 1800–3600 litres per hour. Dug wells represent the most common type, however. In earlier times these were lined with wood or stones, but nowadays they are mainly built with concrete rings. The wells are usually shallow (3–5 m) and 80–120 cm in diameter, exploiting the topmost portion of available groundwater. Artesian tube wells have sometimes been constructed in confined parts of sand and gravel aquifers. Wells drilled into the bedrock are generally 40–80 m deep and 110 mm in diameter, having an average yield of 500–2000 litres per hour (Laakso 1966, Lahermo *et al.* 1990, Korkka-Niemi *et al.* 1993).

The composition of the groundwater depends on a number of geochemical and biogeochemical processes that transform the initial atmospheric water from a recharge area into groundwater with given chemical characteristics at a given position in a given aquifer. Previous studies in Finland have shown that the main factors contributing to the chemical composition of groundwater, as shown schematically in Fig. 1, are of atmospheric, geological, marine and anthropogenic origin.

Atmospheric factors refer to the contribution of the atmosphere to the composition of the groundwater mainly through the composition and amount of rainwater. These factors have a dominant impact in shallow or perched aquifers throughout Finland (Lahermo *et al.* 1990), where the water is a composite result of rain, meltwater from snow and wet and dry atmospheric deposition. Rain and snow contain a wide range of dissolved atmospheric gases, particularly carbon dioxide. In areas close to the sea, the rainfall will also include a variable amount of sea salt components. Surface runoff and point source pollution may also affect the composition of shallow groundwater (Drever 1988, Mather 1997). The av-

Table 1. Average composition of precipitation at rainwater collection stations in Finland (P) in 1990 (Järvinen and Vänni 1992). Medians are calculated from nine measurements for each station. The average groundwater composition in Finland (G) is from the groundwater database of the Geological Survey of Finland (Lahermo *et al.* 1990). Medians express the range of medians calculated for sand and gravel aquifers, till aquifers, clay covered aquifers and bedrock wells. EC = electrical conductivity.

	pH	EC	SO_4	Cl	Na	K	Ca	Mg	NO_3
Units		mS m^{-1}	mg l^{-1}	mg l^{-1}	mg l^{-1}	mg l^{-1}	mg l^{-1}	mg l^{-1}	mg l^{-1}
P (medians)	4.4-5.3	1.3-3.3	0.9-3.6	0.3-1.6	0.17-0.97	0.10-0.45	0.08-0.75	0.04-0.14	0.4-2.8
G (medians)	6.2-6.6	6.1-32.7	5.0-23.0	2.4-13.4	2.5-11.7	1.0-5.1	4.4-27.0	1.4-8.9	0.7-13.6

erage rainwater composition in Finland in 1990, representing the situation during first year of the present research, is shown in Table 1.

Atmospheric factors can be extended to include contributions from the biosphere and pedosphere, as shallow groundwater are greatly affected by biogeochemical (pedological) processes in the soil, as described by Borggaard (1997) and as shown to occur in Finnish groundwater by Lahermo (1975) and Hyyppä and Penttinen (1993). As the vegetation can have a significant effect on rainfall composition, though evapotranspiration, the type of vegetation cover is also a factor (Drever 1988, Mather 1997).

Geological factors refer to the contribution of the geosphere to groundwater composition, mainly through the effect of chemical water-rock interactions in aquifers (Lahermo 1970, Hyyppä 1973, Pönkkä 1981 and Rönkä 1983). In addition to the geochemical environment of the aquifers, the hydraulic factors controlling groundwater flow may be of importance for the chemical composition of the water (Mather 1997). Numerous previous studies (Pönkkä 1981, Soveri 1985, Hatva 1989, Lahermo *et al.* 1990, Backman 1993) have shown that, the chemical composition of the groundwater in many Finnish aquifers is largely a function of residence time. A long residence time will allow more chemical interactions to take place between the water and the solid particles in the aquifers. It seems that the geological structure and texture of an aquifer and the depth and flow regime of the groundwater have a greater impact on water quality than the lithological and geochemical compositions of the aquifer itself. In general, water that follows shallow pathways interacts with more weathered and less reactive minerals than water that is moving along deeper pathways (Mather 1997).

The geological history also affects groundwater quality. During the late phases of the Weichselian deglaciation, the Baltic Sea transgressed over parts of present-day southern Finland, which were at that time isostatically depressed. The Litorina Sea, the most saline phase of the Baltic Sea, covered a wide coastal zone in the mid-Holocene (Eronen and Haila 1990). These events provided the third factor in the composition of the groundwater, the marine influence. As shown by Lahermo *et al.* (1990) and later by Karro (1999), relict seawater trapped in bedrock fissures and fractures and saline pore water in marine clay and silt sediments can be detected from increased concentrations of dissolved components in the groundwater, particularly SO_4, Cl and Na. Direct intrusion of present seawater into aquifers seems to be rare in Finland nowadays (Lahermo *et al.* 1990).

Anthropogenic factors refer to human impact on groundwater composition, mainly through urban and industrial development and modern agricultural practices. By definition, all types of chemical loading from unnatural sources may be termed pollution. Impacts of this kind are commonly divided into two broad types on geometrical grounds: point source pollution, where the pollutants are derived from a discrete, usually identifiable source such as a factory or waste disposal site, and diffuse pollution, where the input is widely distributed, as in the case of acid rain (Mather 1997). Many Finnish authors have documented the effects of point or diffuse pollution in terms of increases in electrolyte concentrations in the groundwater (e.g. Pönkkä 1981, Lahermo 1984, Lahermo *et al.* 1990, Kujala-Räty *et al.* 1998). The 'contaminated soil site survey and remediation project' (Puolanne *et al.* 1994) has identified over 20 000 possibly contaminated sites in Finland, more than 2000 of which are in classified groundwater areas (Britschgi and Gustafsson 1996) and 1600 are located close to residential areas. Most cases are related to garages, scrap yards, waste disposal plants, sawmills or wood impregnation plants (Puolanne *et al.* 1994). These are typi-

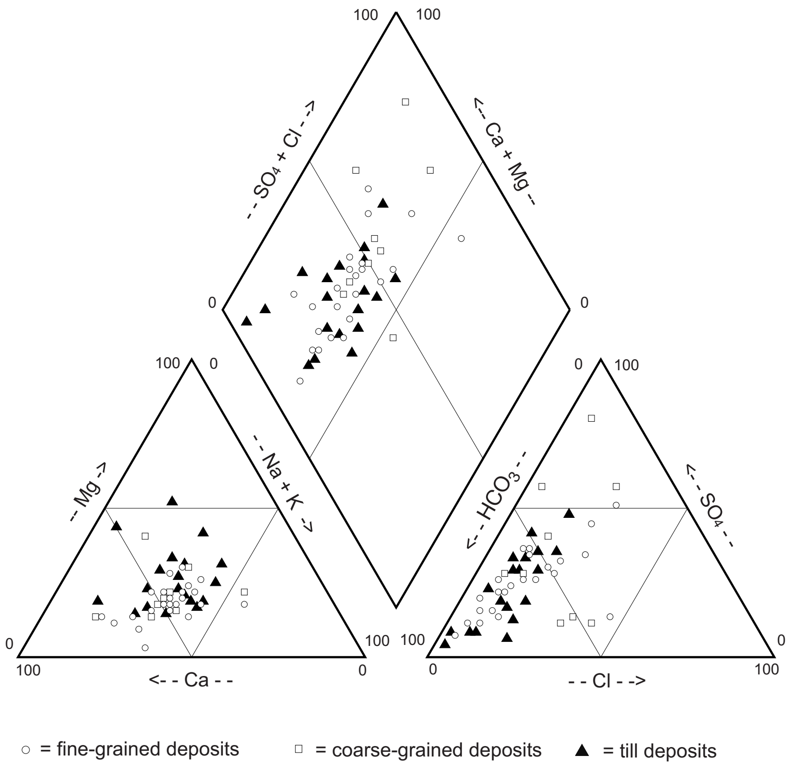


Fig. 2. General characteristics of the groundwater in Finland (Soveri *et al.* 2001).

cal point sources affecting groundwater quality. Road salting (de-icing and dust binding) gives rise to numerous point sources of groundwater pollution, the effects of which can be seen in increased chloride concentrations in aquifers traversed by roads that are frequently salted (Nystén 1998). There is also evidence of well water acidification, in that the monitoring studies of both Korkka-Niemi (1990) and Backman (1993) point to elevated electrical conductivity values and lowered pH and alkalinity on account of acid loading.

The lack of properly organised waste water treatment has caused faecal contamination of water in the past, especially in the rural areas of Finland, and still does so to some extent (Korhonen *et al.* 1996, Heinonen-Tanski *et al.* 1999). Many reviews, e.g. in Macler and Merkle (2000), indicate that significant faecal contamination of groundwater wells is a common phenomenon in many countries, leading to the conclusion that a better understanding is needed of the factors controlling microbial contamination of groundwater, which is also true as far as Finnish conditions are concerned. According to Macler and Merkle (2000), the background data required should cover the site-specific hydrogeological properties that affect vulnerability to contamination and details of the existence and types of contamination sources in the subsurface.

All these factors contribute to the general characteristics of the groundwater in Finland. It can be concluded from the GSF survey of 1972–1982 (Lahermo *et al.* 1990) and the monitoring programmes carried out at the groundwater stations of the Finnish Environment Institute (Soveri *et al.* 2001) and the Geological Survey of Finland (Backman *et al.* 1999) that the groundwater in Finland is usually of a very soft or soft Ca-HCO_3 type (Table 1 and Fig. 2), reflecting the predominant geological characteristics of the granitic bedrock. The influence of marine relict salts or present seawater intrusion into the aquifers can alter the water type to Na-Cl-dominant in coastal areas, however, and Ca-SO_4 type water (Fig. 2) exists in rare occurrences restricted to areas with a sulphide mineral potential, whereas magnesium-rich waters

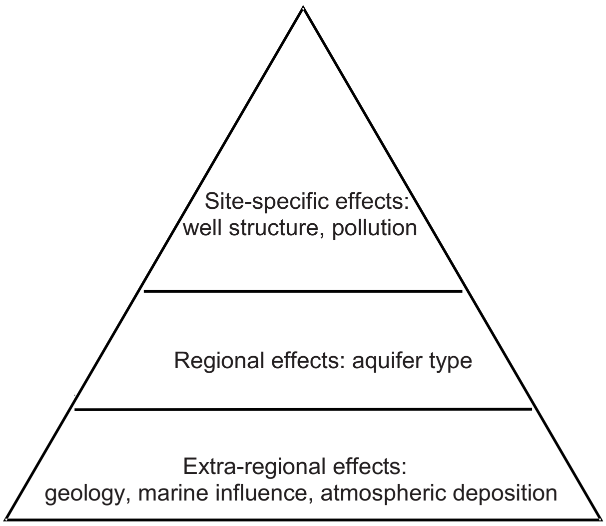


Fig. 3. Cumulative factors affecting groundwater quality on various scales.

are very seldom encountered (Lahermo *et al.* 1990). The most harmful aesthetic and technical features of water quality are comparatively high dissolved organic carbon and carbon dioxide concentrations together with low pH values and high iron and manganese (Wäre 1960, Natukka 1963, Pönkkä 1981, Lahermo 1984, Hatva 1989, Mäkinen 1989).

1.3 Cumulative effects on groundwater quality

All these atmospheric, geological, marine and anthropogenic factors operate on various scales and affect individual wells differently according to the preponderance of one factor over another. The influence of each factor is also dependent on all the others. The (litho-)geological factor, for instance, may be overwhelmed by marine influences or atmospheric factors in shallow aquifers with high transmissivity, the degree of this influence possibly being dependent on the type of aquifer and still more so on local factors such as agricultural practices and soil processes, which may be preponderant. Furthermore, the quality of the water as sampled in a well may be dependent even more on highly site-specific factors such as imperfect sealing of the well rings or surface water infiltration into the well collar (Lahermo *et al.*1990, Korkka-Niemi *et al.* 1993), or even on contamination from local septic facilities.

The numerous factors that may affect groundwater quality can be integrated into a concept of cumulative effect, as defined in the present study. This refers to the integrated, additive or balancing effects of the multiple factors that may affect well water quality at any given location. The concept is shown schematically in Fig. 3 as a series of three 'layers' of factors, as discussed above:

1) Extra-regional effects
2) Regional effects
3) Site-specific effects

The horizontal axis of the 'layers' is representative of the extent of the influence of the factor. The lowest layer, geological effects, for instance, is depicted as wide, reflecting its broad, extra-regional character. As the spatial extent of each layer decreases upwards, the diagram assumes the form of a pyramid, progressing from broad extra-regional effects at the base to highly site-specific factors at the top. The vertical axis reflects the cumulative action, from the base of the pyramid to the apex of the triangle, ultimately applying to the household level.

The extra-regional or geological effects are rooted in the geological factors as discussed above, but include marine influence as well, and are therefore dependent on the nature and composition of the rock and surficial sediment, and also on the deglacial history of the aquifer. These factors are regarded as extra-regional because they apply over large areas regardless of watersheds or municipal borders. In order to detect geological effects, a database on wells should include the following measurements or observations: whether the wells are rooted in bedrock or in surficial sediments, the nature and the composition of the rock and sediments, the distance from the present seashore, the geological history of the site, and other geological determinants that may confer a particular 'natural' geochemical signature on the water. Finally, because the effect of geological factors is by definition very broad, the set of samples used for evaluating it has to be regional and must be statistically valid on that scale.

Regional or hydraulic effects stem from factors related to the residence time of the water and the time available for reacting with the solid particles of the aquifer, and on the proximity of the aquifer system to the surface, and are therefore dependent largely on the type and nature of the aquifer and the hydraulic flow conditions. Aquifer systems in Finland are much smaller in extent than the geological factors discussed above, and can be considered regional to local in scale. They are thus shown as the middle layer of the pyramid. In order for a data set to be discriminant at that level, the constraints are that it should include observations or measurements aimed at elucidating the aquifer type, such as shallow or deep, or an aquifer located in sand and gravel or in till or an aquifer being clay covered. This means that one should attempt to determine or deduce the nature of the aquifer in which each well that one sets out to study is located.

Finally, water may still differ in quality from one well to another even if the geological contexts are similar and the hydraulic or aquifer type effects are comparable. This, it is believed, will be due mainly to local or highly site-specific factors. Furthermore, considerations as to the type of well (spring, dug well lined with stones or concrete rings), its 'age', the configuration of the water intake and the existence of any waste water disposal installations, animal stables, septic tanks, etc. in its immediate vicinity of the well will have to be added to the data set.

Intermediate between the regional and site-specific factors are 'land-use' effects such as agricultural practices or local industrial influences. These are included in the site-specific factors in the pyramid of Fig. 3, although they could just as well be included in the middle layer. The rationale for their grouping with the site-specific factors is that they share with the latter the distinction of being exogenous, as opposed to the endogenous effects of aquifer type and geological context.

1.4 Aims of the research and organisation of the thesis

The present research aims to provide a discriminant nation-wide survey of groundwater quality of domestic wells on a household scale. Its main purpose is to describe the cumulative effects of extra-regional, regional and site-specific factors on well water quality. The design of the data set is discussed more specifically in Chapter 2.

Since previous studies have pointed to seasonal variations in groundwater quality (Mälkki 1977), the present data set was acquired in the shortest possible time, within a single season, although sampling at some sites was extended through other seasons in order to allow a rough analysis to be made of the seasonal effects. Finally, a comparison with the Wäre survey provides some understanding of long-term trends in groundwater quality in Finland.

The research protocol is described and discussed in Chapter 2, after which the nature of the data set and the process by which the wells were selected, sampled and surveyed by means of a questionnaire and the examination of maps are presented. Finally, the methods of statistical analysis and the reasons for selecting them are presented.

The main results of the survey are presented in Chapter 3, where the seasonality and long-term changes in groundwater quality are also discussed. The correlation coefficients and principal component analysis leading to recognition of the factors which are the most significant in determining groundwater quality are discussed in Section 4.1, the microbial indicators of water quality are discussed separately in Section 4.2 and the well water samples are grouped by clustering analysis in Section 4.3. Finally, Section 4.4 discusses the verification of the findings by means of models of the cumulative effects as suggested here. The major findings are summarised in Chapter 5.

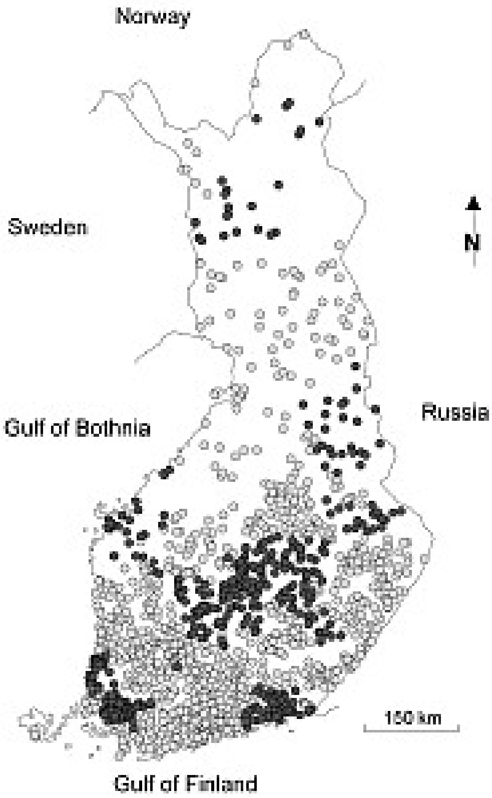


Fig. 4. Map of Finland showing the distribution of the wells studied. Black dots represent the wells included in the additional sampling undertaken to investigate seasonal effects on water quality.

2 Material and methods

2.1 Design of the data set

2.1.1 Sampling strategy

The aim was to acquire a data set that would be broad and regional, thus ensuring that it would be statistically valid and significant, and focussed on a set of quality variables closer to those required for potable water standards than for hydrogeochemical analysis. The data were gathered so as to represent each scale of environmental variables affecting the water quality in individual wells, and included chemical water analyses, site-specific factors obtained from a questionnaire and geological variables analysed by desktop study.

Data on 1421 private wells selected systematically to represent the quality of drinking water in rural areas throughout Finland were collected in 1990 (Fig. 4). The universe for the survey comprised all the registered 360 000 households which had not been connected to municipal water supplies. Every hundredth household was selected, and after elimination of false cases the number was reduced to about 3000. These were then systematically assigned to two groups, A and B, having 1500 wells in each. The water supplies in the selected households, those in group A, were checked, and all households not using their own well were discarded and replaced with the geographically nearest household in group B.

As seen in Fig. 4, the distribution of sample wells is not even but reflects the variation in the density of the rural population in Finland. This means that the data are unlikely to be strictly representative either geologically or geographically, but merely constitute a cross-section of existing wells in Finland. Some samples were taken through three seasons in order to allow rough analyses of the seasonal effects, the additional samples being collected from six areas. The spatial distribution is explained by Korkka-Niemi *et al.*(1993).

The raw data emerging from the survey have been published earlier in the form of tables by Korkka-Niemi and Sipilä (1993). The geological data have been collected to complete this study and have not been published before.

2.1.2 Sampling procedure

Samples were taken once from each of the 1421 wells in autumn 1990, with additional samples taken from 427 wells in spring and summer 1991. The sampling was carried out by the municipal health authorities following a uniform procedure. Samples were typically taken from taps (Fig. 5A), after removing any filters and allowing the water to run until it reached a constant temperature. Well samples were lifted with a clean sampler (Ruttner-type, bailer-type or plastic bucket), or with a sampling pump (Fig. 5B), from the same depth as the water taken for daily use. Bottled unfiltered and unpreserved samples were stored in a dark, cool place and transported to the laboratory on the same day. The samples were normally analysed the next day, and in any case within 72 hours of sampling.

Fig. 5. Sampling from a water tap (A) and directly from a well (B).

2.1.3 Analytical procedure

The water samples were analysed for total coliform bacteria, thermotolerant (= faecal) coliform bacteria, faecal streptococci (= enterococci), electrical conductivity (EC), pH, alkalinity, total hardness, colour, $KMnO_4$ consumption, turbidity (FTU), SO_4, Cl, F, NO_3, NO_2, NH_4, Na, K, Fe, Mn and Al to describe their potability. In addition to the above bacterial analyses, the number of *Escherichia coli* bacteria (*E.coli*) was determined in each sample in which faecal coliform bacteria were identified, i.e. 145 samples in autumn 1990 and 107 in summer 1991. The number of confirmed enterococci among the presumptive faecal streptococci was also determined.

The samples were analysed at two water laboratories in Helsinki, those of Oy Vesi-Hydro Ab and Suunnittelukeskus Oy, according to the standards and detection limits listed below (Table 2). The laboratories made comparative tests before and during the analyses. The samples were in general not filtered, the one exception being the subsamples prepared for aluminium analysis, which were passed through a 0.2 µm membrane filter.

Table 2. Analytical methods used on the well water samples.

Variable	Method/Standard	Technique	Unit	Detection limit
Total coliform bacteria	SFS 3016	Membrane filtration (LES Endo agar)	Number / 100 ml	1
Faecal coliform bacteria	SFS 4088	Membrane filtration (mFC agar)	Number / 100 ml	1
Escherichia coli	SFS 4088	Membrane filtration*	Number / 100 ml	1
Faecal streptococci	SFS 3014	Membrane filtration (mEnterococcus agar)	Number / 100 ml	1
Confirmed faecal streptococci	SFS3014	Membrane filtration**	Number/100 ml	1
Electrical conductivity (EC)	SFS 3022	Conductometry	mS m^{-1}	0.1
pH	SFS 3021	pH-electrode		0.01
Alkalinity	VAT	Indicator titrimetry	mmol l^{-1}	0.01
Total hardness	SFS 3003	Titrimetry	mmol l^{-1}	0.01
Colour number	SFS 3035		Pt mg l^{-1}	5
Turbidity	SFS 3024	Nephelometry	FTU	0.01
$KMnO_4$ consumption	SFS 3036	Titrimetry	mg l^{-1}	1
Sulphate, SO_4	VAT	Turbidimetry	mg l^{-1}	1
Chloride, Cl	ES	Titrimetry	mg l^{-1}	1
Fluoride, F	SFS 3027	Ion-selective electrode	mg l^{-1}	0.1
Nitrate, NO_3		AKEA AutoAnalyser	mg l^{-1}	0.1
Nitrite, NO_2	SFS 3029	Spectrophotometry	mg l^{-1}	0.01
Ammonium, NH_4	SFS 3032	Photometry	mg l^{-1}	0.1
Sodium, Na	SFS 2017	AAS	mg l^{-1}	0.02
Potassium, K	SFS 3017	AAS	mg l^{-1}	0.05
Iron, Fe	SFS 3044,3047	AAS	mg l^{-1}	0.05
Manganese, Mn	SFS 3044,3048	AAS	mg l^{-1}	0.02
Aluminium, Al	SFS 5736	AAS	mg l^{-1}	0.01

AAS = Atomic absorption spectroscopy, flame atomisation
ES = Elintarviketutkijain seura ry (Anon. 1962)
VAT = Vesianalyysitoimikunnan mietintö (Anon. 1968)
AKEA = Automaattinen kemiallinen analysaattori

* *E. coli* were counted by isolating typical blue colonies growing on mFC plates for confirmation. The isolation medium was tryptone (0.5 %), yeast extract (0.3 %) agar incubated for 24 h at 37 ºC. Strains giving negative results in the oxidase test and producing gas from mannitol at 44.5 ºC in 24 h in lauryl tryptose mannitol tryptophane broth (SFS 4088, 1988) as well as a positive indole reaction were regarded as presumptive *E. coli*.

**Confirmed faecal streptococci were determined by subculturing typical colonies from mEnterococcus plates on bile aesculin azide agar and incubating at 44 ºC for 44 h. Strains giving negative results in the catalase test and showing aesculin hydrolysis were regarded as confirmed faecal streptococci.

2.1.4 Geological data

In most cases no documents are available on the construction of the wells, which means that it is difficult to describe the 3D geology and hydrogeology of the aquifers. There are two ways of filling this gap, either by means of grain-size analysis, as used by Wäre (1960), or by inferring the hydrogeological environments of the individual wells from geological maps. The latter approach was employed here.

The geological environments of the wells were described in 1998 using geological and topographic maps at the most detailed scale available (1:20 000–1:400 000). The rock type, surficial sediments and depositional environment were recorded from maps for all the wells studied.

Maps of the Precambrian bedrock were used to relate the environments of the wells to major lithochemical provinces, 'bedrock type' being taken to refer to the main rock type at the well site. The rocks were classified into 11 petrological groups after Korsman *et al.* (1997).

1) rapakivi granite
2) S-type granite (microcline granite)
3) other granites
4) the TTG series (tonalite, trodhjemite, granodiorite, quartz diorite, granite gneiss, gneiss granite)
5) the gabbro group (diorite, gabbro, diabase)
6) the gneiss and schist group (mica gneiss, vein gneiss, gneiss, mica schist, greywacke, phyllite, migmatite)
7) black schist
8) quartzite
9) clay/sandstone
10) mafic volcanic rocks (greenstone, uralite porphyrite), and
11) intermediate and felsic volcanic rocks (amphibolite, tuffite, leptite, hornblende gneiss, potassium feldspar gneiss, potassium feldspar schist).

The data item 'surficial sediment' was used to describe the grain size distribution and sorting of the sediment in which the well had been dug, the categories being: clay, silt, sand and gravel or till. This information was deduced from maps on which field observations and geotechnical or engineering geology classifications of grain size distribution had been marked.

The category 'depositional environment' referred to the physical environment in which the sediment had been deposited. The clays and silts were taken as having been deposited during marine or lacustrine phases of the Baltic Sea (the Baltic Ice Lake, Yoldia Sea, Ancylus Lake or Litorina Sea), or else as being fluvial in origin, while sand and gravel deposits were recorded as representing glaciofluvial, alluvial or littoral deposits or parts of ice-marginal end moraine complexes. Some wells in northern Finland were located in outwash valley deposits. Moraines were classified into three groups according to their morphology and depositional environment: cover moraines, morainic landforms (hummocky moraines and drumlins) and ice-marginal end moraine complexes. The 'depositional environment' was used to explain the aquifer characteristics.

The shortest distance from each well to the sea was measured on the basis of the x,y coordinates of the wells and on map of scale 1:1 000 000, by means of the ARC-Info program.

2.1.5 Questionnaire data

The site-specific factors that could have affected well water quality were analysed on the basis of observations gathered on each well and its surrounding. For this purpose, a questionnaire was filled in by the sampler, providing information on matters such as the location and construction of the well, the amount of water used, water treatment, environmental condition of the well and subjective water quality observations. The questionnaire included the following sections:

I) Identification: Municipality, owner of the well, telephone number and address of the household.
II) Well: Code of the well, map sheet, estimated yield, location of well (yard/arable land/forest/other), type of well and lining material (stone-lined well/concrete ring well/bedrock well/spring/other), diameter of the well, depth of the well, depth of the water table, age of the well, year of last cleaning of the well, estimated access of surface runoff to the well (yes/no), estimated condition of the wells construction (good/moderate/poor) and amount of water (good/occasionally insufficient/constant lack of water).
III) Sample: sampling device (well/tap), representativeness of the sample (good/moderate/

poor), observed colour of the sample (colourless/coloured) and turbidity of the sample (clear/turbid).

IV) Water treatment and water pipes: Any water treatment applied, age of the water pipe, diameter and material of the pipe and the estimated condition of the pipe (good/moderate/poor).

V) Sources of pollution: Possible sources of pollution (cowshed/piggery/outdoor toilets/roads etc.), distance from the well to any pollution source and estimated probability of the pollutant penetrating into the well (yes/no/possibly).

VI) Sewage system: municipal sewage system/septic tank/discharge into the ground/discharge into a water body.

VII) Observations on water quality and other information given by the well owner.

VIII) Name of the sampler, telephone number and date of the sampling.

A map with the location of the well and some photographs of the well were normally attached to the completed questionnaire.

2.1.6 Limitations of the material

The accuracy of the survey may be affected by the samples having been collected by different people. The same sampling procedure was employed throughout, however, to reduce the variations in sampling and all the people involved were trained in taking water samples. The variable time required for transporting the samples to the laboratory (0–3 days) may have interfered with certain measurements to a minor extent (e.g. pH, alkalinity, turbidity and colour). In some cases turbidity or microbial growth was so pronounced that it made the analysis of bacteria impossible. These 19 results were discarded from the list of bacterial analyses.

The samples were collected from a tap when possible (88 %), but if there was no water pipe the sample had to be taken directly from the well. The most common pipe material was plastic, but iron, copper and galvanised steel pipes were also encountered. The influence of the water pipes on water quality was studied by taking simultaneous samples from the well and tap in the case of 100 households and comparing the results of the analyses. It was observed that zinc and copper dissolved into the water from the pipe material and that some changes in iron, manganese, oxygen and carbon dioxide concentrations could also be detected (Korkka-Niemi *et al.* 1993), but it should be noted that in this additional study the water has not been running before sampling, so that the influence of the pipes would have been more pronounced than under normal conditions. In order to minimise the effect of the pipes, the water was run before sampling in the main round.

The selection of ion analyses performed here is not adequate for geochemical or hydrogeological characterisation, as not all the major ions were analysed. Similarly, the measurement of total hardness rather than calcium and magnesium concentrations, choice based on potability criteria, does not allow geochemical interpretations to be made. Only statistical analysis of the data was possible. For the same reason, the ionic balance error of the samples could not be calculated (see Appelo and Postma 1994).

2.2 Data processing and statistical analysis

2.2.1 Research procedure

All 1421 wells were used to describe the general water quality in Finnish wells and to identify the existing problems. The water quality in these wells was compared with that reported by Wäre in the late 1950's in order to assess long-term changes, and a comparison was also made between the dug wells (1096 wells) and bedrock wells (325 wells). Possible seasonal changes were observed by comparing analyses from the same 423 wells at three seasons of the year. The water quality analyses and background data from the questionnaires and geological maps were used to identify the factors or variables that served best as determinants of groundwater quality. A schematic diagram of the research procedure and the statistical analyses used is presented in Fig. 6.

2.2.2 Guidelines for drinking water quality

The well water quality variables were compared with the quality standards for potable water set by the Finnish health authorities (Ministry of Social

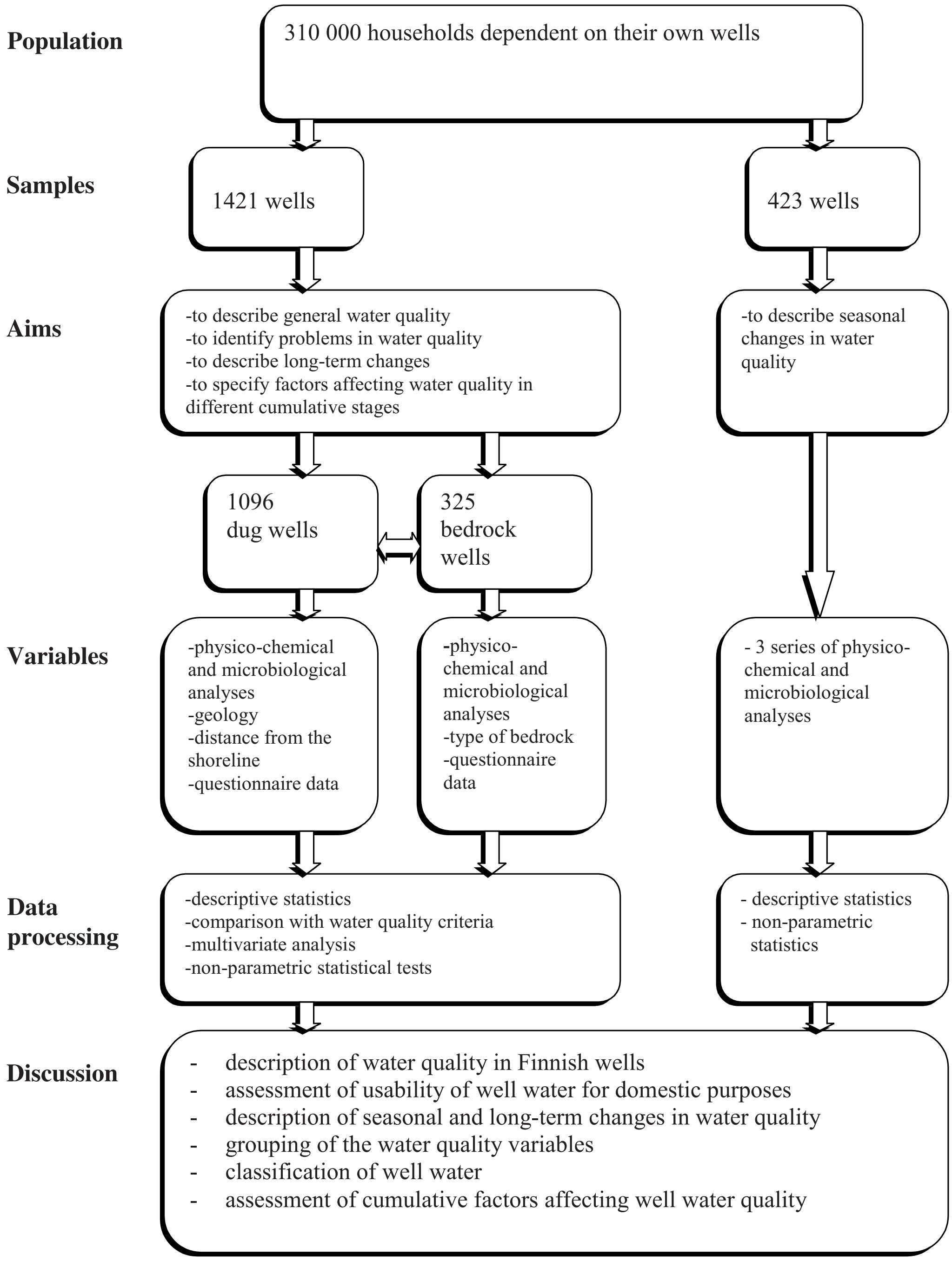


Fig. 6. Schematic diagram of the research procedure.

Affairs and Health 1994). The national standards for the variables analysed here, where such standards have been defined, are shown in Tables 3 and 4. New drinking water quality standards were laid down in the year 2000 for water supplied from water intake plants, but no new standards were yet issued for potable water taken from wells serving a single household (Anon. 2000). The Finnish legislation concerning drinking water quality follows the Directive of the European Economic Community relating to the quality of water intended for human consumption (80/778/EEC). The provisions on the quality of drinking water are divided into two groups: health-related quality standards (Table 3) and technical-aesthetic standards (Table 4).

2.2.3 Graphical and statistical methods

All the data were stored in a SAS database. The results were examined graphically and statistically using EXCEL 97, SAS 6.12, SPSS 8.0, STATVIEW 98 and STATISTICA 5.0 and 6.0 software.

The distributions of the variables measured were first studied graphically and using frequency distribution parameters, in order to describe them and to select proper statistical tests for their further analysis, as suggested by Reimann and Filzmoser (1999). The frequency distributions of the main properties of the water samples from the wells were illustrated by means of box plots and examined using selected distribution parameters such as the median, arithmetic and geometric means, standard deviation, kurtosis and skewness. The data were tested for normality and log-normality visually (histograms) and graphically (probability-probability plots). The spatial distributions of well water quality and quality problems were presented as thematic maps (MapInfo Professional 5.0).

All concentration values in the 1990 data that fell below the analytical detection limit (DL) were replaced by half of the DL value, as suggested by Frapportti (1994), Chapman (1996) and Reiman and Filzmoser (1999). Histograms were used to identify water quality defects. Cumulative frequency plots for 21 variables in the groundwater

Table 3. Health-related quality standards for water intended for human consumption (Ministry of Social Affairs and Health 1994).

Variable	Maximum density	Maximum acceptable concentration (mg l^{-1})
Microbiological standards		
E.coli (presumptive)	less than 1/100 ml	
Total coliform bacteria	less than 100/100 ml	
Chemical standards		
F		1.5
NO_3		25
NO_2		0.1

Table 4. Technical-aesthetic standards for water quality (organoleptic and physico-chemical variables, and variables concerning substances undesirable in excessive amounts)(Ministry of Social Affairs and Health 1994).

Variable	Maximum acceptable concentration (mg l^{-1})	Recommended level
Al	0.2	
NH_4	0.5	
Cl	100	
$KMnO_4$ consumption	20	
Mn	0.2	
Fe	0.5	
SO_4	250	
Na	150	
pH		6.0–9.5
Turbidity (FTU)		< 5
Colour number		< 20

from Quaternary deposits and bedrock were presented separately to compare the distributions within the groups.

Non-parametric statistical tests were used because of the non-normal distribution of most of the variables (see Section 3.2). The non-parametric Mann-Whitney U-test for two unrelated or independent populations and the non-parametric Median Test (Rock 1988 and Ranta *et al.* 1991) were used to examine the differences in water quality between the wells which seemed to be penetrated by surface water and the wells without any visible surface water effect (Chapter 3). The Mann-Whitney U-test is a non-parametric version of the two groups unpaired *t*-test and is a powerful test for unrelated medians (Rock 1988), as it analyses the equality of the medians of the populations from which two samples were drawn (Swan *et al.* 1995). The median test is a crude test for unrelated medians (Rock 1988) that is used to verify the results of the Mann-Whitney U-test (calculated as Wilcoxon scores).

For selected data concerning seasonal changes in water chemistry, values below the DL were replaced by the DL, because data values at the DL were not distinguished from those below it in the spring and summer samples. This approach can be used instead of taking half of the detection limit (Frapporti 1994, Chapman 1996). The non-parametric Wilcoxon signed-ranks test with Bonferroni correction (Ranta *et al.* 1991, Rock 1988) was used to confirm any seasonal variation in water quality. This is a non-parametric equivalent to the paired *t*-test (Chapter 3).

The methods of water analyses used in 1958 (Wäre 1960) and in 1990 were based on the same principles, but the accuracy had improved in the meantime. Since changes were also made in the way the results were presented, it was also necessary to make the results from 1958 comparable with those from 1990. The main statistical parameters and cumulative distribution curves (Chapter 3) were used to describe the changes, and the distributions for the two years were compared using the non-parametric Mann-Whitney U-test for two unrelated populations (Rock 1988, Ranta *et al.* 1991).

Frequency tables, scatter plots and box plots were used to interpret the microbiological indicators of water quality (Chapter 4) and to describe the extra-regional (geology, atmospheric deposition, distance from the seashore), regional (hydrogeology, seasonality) and site-specific factors (Chapter 3). The correlations between variables (Chapter 4) were presented in the form of non-parametric Spearman Rank correlation coefficients on the basis of the raw data (1421 wells).

R-mode Principal Component Analysis (PCA) has been employed in hydrogeochemical studies to describe the processes controlling the hydrochemistry (e.g. Briz-Kishore and Murali 1992, Frapporti 1994, Jayakumar and Siraz 1997, Join *et al.* 1997). PCA is essentially a descriptive method, the objective being to extract components representing the information contained in the data. PCA reduces multivariate data to fewer dimensions (Rock 1988, Ranta *et al.* 1991, Join *et al.* 1997), thus transforming the original set of variables into a new set of uncorrelated principal components which are at right angles to each other (Rock 1988). The method provides several features that allow interpretation of the data sets. By examining the factor loadings, communalities and eigenvalues, the variables related to specific geochemical or environmental processes can be identified and the importance of elements can be evaluated in terms of the total data set and factor by factor (Jayakumar and Siraz 1997). PCA was performed on the data for all 1421 wells (Chapter 4). Logarithmic transformation was performed to obtain multinormality in the data and in some cases to compress the numerical range, and the factor scores for the individual wells were used to examine the causative factors lying behind water quality.

Cluster analysis represents an attempt to reorganise the data set into homogeneous groups, with no use made of prior knowledge. In the present case, k-means clustering was used for the whole body of data (1421 wells), after log-transformation and standardisation, in order to classify the wells into groups (Chapter 4), and the clusters were then examined in order to determine the variables which divide well water samples into groups and cross-tabulation used to describe the wells and their environments inside the clusters.

3 Results and comparisons

3.1 Properties of the well water in 1990

The general characteristics of the well water are described in Tables 5–7 and in the box plots in Fig. 7. The water is slightly acid, with pH in the range (*Q1–Q3*) 6.3 to 7.1. In autumn the pH was less than 6 in 10 % of the wells. Alkalinity ranged (*Q1–Q3*) from 0.43 to 1.5 mmol l^{-1} and total hardness between 0.72 and 0.97 mmol l^{-1}. Fluoride exceeded the detection limit (0.1 mg l^{-1}) in about half of the wells, whereas ammonium and nitrite could be detected only in about 10 %, and usually at very low concentrations. Nitrate concentrations were generally within the range 0.5 to 15 mg l^{-1}, but isolated high values were also encountered. Electrical conductivity and chloride and sulphate content were usually low, whereas the colour value, $KMnO_4$ consumption and turbidity were high, with the first two of these exceeding the detection limits in almost half of the wells. Iron exceeded the DL in almost all the wells, manganese in less than 30 % and aluminium in a half. Total coliform bacteria were recorded in over 60 % of the wells in 1990, thermotolerant coliforms being identified in 10 % of the water samples and faecal streptococci in one fifth.

3.2 Statistical distribution of variables

Examination of the histograms and distribution statistics (Table 5) suggests that the variables can be divided into four main types according to their distribution. They are type I: normally or log-normally distributed variables, type II: variables with high kurtosis and a skewed distribution, type III: variables with many samples below the detection limits, and type IV: indicator variables (Fig. 8).

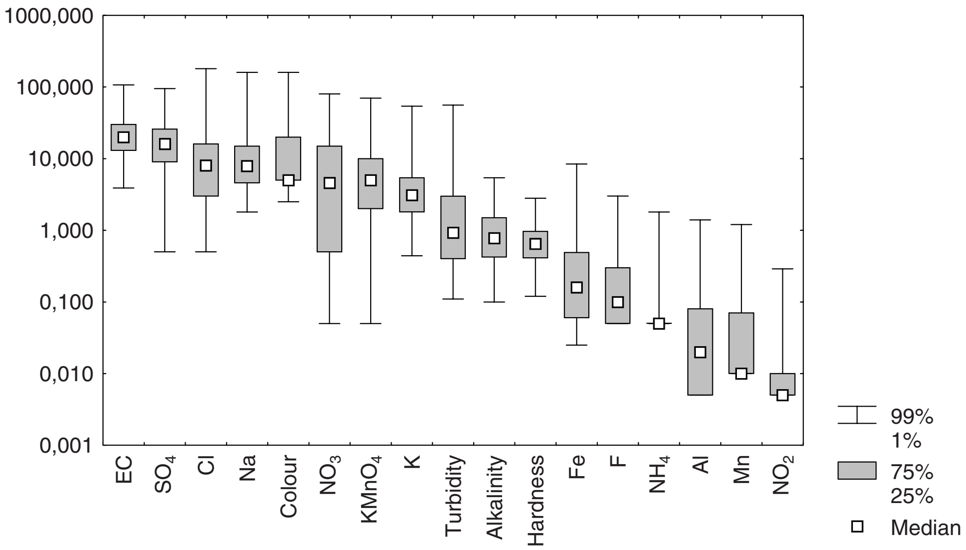


Fig. 7. Box plot illustrating the ranges of all the variables analysed in the 1421 well water samples except for bacterial counts and pH, sorted on the basis of decreasing median concentrations. The concentrations are in mg l^{-1}, except for alkalinity and total hardness (mmol l^{-1}) and electrical conductivity (mS m^{-1}).

Table 5. General characteristics of well water quality in Finland in 1990. The number of wells is 1421.

Variable	Unit	Median	Aritm. Mean	Geom. Mean	Min.	Max.	Std.dev.*	Kurtosis	Skewness
Total coliform bacteria (TC)	number in 100 ml	4	68.43	4.83	< 1	7400	326	241.76	13.55
Faecal coliform bacteria (FC)	number in 100 ml	< 1	2.82	0.63	< 1	1190	39	740.75	26.53
Faecal streptococci (FS)	number in 100 ml	< 1	12.22	0.95	< 1	6250	174	1156.6	32.75
Electrical conductivity (EC)	mS m^{-1}	20.0	25.9	19.9	2.1	510.0	27.3	105.71	7.85
pH		6.60	6.73	6.70	4.80	9.00	0.64	0.35	0.68
Alkalinity	mmol l^{-1}	0.78	1.16	0.79	< 0.01	11.00	1.11	9.77	2.47
Total hardness	mmol l^{-1}	0.65	0.78	0.62	0.07	15.00	0.68	138.22	7.95
Colour number	Pt mg l^{-1}	5	17.7	8.5	< 5	400	32.5	41.94	5.39
Turbidity	FTU	0.93	4.52	1.22	0.04	152.00	11.92	55.54	6.44
$KMnO_4$ consumption	mg l^{-1}	5	9.0	4.2	< 1	224	14.8	58.78	6.25
SO_4	mg l^{-1}	16	20.7	14.6	< 1	212	18.7	18.61	3.23
Cl	mg l^{-1}	8	20.0	7.6	< 1	1770	72.3	303.52	15.38
F	mg l^{-1}	0.1	0.37	0.15	< 0.1	5.8	0.66	13.62	3.29
NO_3	mg l^{-1}	4.6	11.28	2.63	< 0.1	233.0	18.6	35.17	4.50
NO_2	mg l^{-1}	0.05	0.03	< 0.01	< 0.01	8.0	0.24	842.24	27.32
NH_4	mg l^{-1}	< 0.1	0.16	< 0.1	< 0.1	33.0	1.2	448.18	19.60
Na	mg l^{-1}	7.90	17.14	9.09	1.30	580	36.56	83.80	7.82
K	mg l^{-1}	3.10	6.15	3.31	0.06	230.00	13.57	104.29	8.84
Fe	mg l^{-1}	0.16	0.65	0.19	< 0.05	42.00	1.99	166.78	10.73
Mn	mg l^{-1}	0.10	0.10	0.03	< 0.02	6.30	0.32	171.89	11.28
Al	mg l^{-1}	0.02	0.12	0.03	< 0.01	6.00	0.35	111.44	8.89

* standard deviations of $\log_{10}$ transformed TC, FC and FS are 1.00, 0.34 and 0.59, respectively.

Table 6. General characteristics of well water quality in bedrock wells in 1990. The number of wells is 325.

Variable	Unit	Median	Aritm. Mean	Geom. Mean	Min.	Max.	Std.dev.*	Kurtosis	Skewness
Total coliform bacteria	number in 100 ml	< 1	41.85	1.63	< 1	5000	319	191.17	13.28
Faecal coliform bacteria	number in 100 ml	< 1	0.89	0.58	< 1	19	2	48.34	6.65
Faecal streptococci	number in 100 ml	< 1	1.78	0.63	< 1	133	9	15.53	11.41
Electrical conductivity	mS m^{-1}	25.0	33.3	26.1	5.1	406.0	34.0	52.28	5.96
pH		7.20	7.15	7.11	5.10	8.90	0.77	-0.76	-0.20
Alkalinity	mmol l^{-1}	1.40	1.70	1.25	0.03	7.20	1.23	2.14	1.26
Total hardness	mmol l^{-1}	0.69	0.85	0.70	0.07	6.20	0.64	18.67	3.27
Colour number	Pt mg l^{-1}	5	17.9	8.5	< 5	250	30.8	18.80	3.84
Turbidity	FTU	0.75	3.31	1.01	0.04	44.00	6.74	14.10	3.54
$KMnO_4$ consumption	mg l^{-1}	4	8.2	3.2	< 1	160	15.1	48.33	6.15
SO_4	mg l^{-1}	17	21.8	15.2	< 1	110	18.2	5.80	2.01
Cl	mg l^{-1}	10	29.1	9.7	< 1	1090	93.56	81.38	8.54
F	mg l^{-1}	0.3	0.68	0.30	< 0.1	5.8	0.90	7.55	2.42
NO_3	mg l^{-1}	1.2	6.95	1.05	< 0.1	74.0	11.7	8.17	2.61
NO_2	mg l^{-1}	< 0.01	0.02	< 0.01	< 0.01	3.40	0.19	310.00	17.43
NH_4	mg l^{-1}	< 0.1	< 0.1	< 0.1	< 0.1	2.0	0.13	140.16	10.75
Na	mg l^{-1}	14.00	32.74	16.24	1.40	580.00	54.39	39.07	5.19
K	mg l^{-1}	2.90	4.26	3.10	0.06	36.00	4.60	20.47	3.99
Fe	mg l^{-1}	0.13	0.55	0.16	< 0.05	23.00	1.59	125.46	9.76
Mn	mg l^{-1}	0.02	0.11	0.03	< 0.02	2.70	0.23	53.52	6.12
Al	mg l^{-1}	0.01	0.07	0.02	< 0.01	2.00	0.18	45.09	5.73

*standard deviations of $\log_{10}$ transformed TC, FC and FS are 0.84, 0.25 and 0.35, respectively.

Table 7. General characteristics of well water quality in dug wells and springs in 1990. The number of wells is 1096.

Variable	Unit	Median	Aritm. Mean	Geom. Mean	Min.	Max.	Std.dev.*	Kurtosis	Skewness
Total coliform bacteria	Number in 100 ml	6	76.41	6.69	< 1	7400	326.88	257.51	13.69
Faecal coliform bacteria	Number in 100 ml	< 1	3.40	0.64	< 1	1190	44.48	571.69	23.32
Faecal streptococci	Number in 100 ml	< 1	15.36	1.08	< 1	6250	198.48	893.55	28.79
Electrical conductivity	mS m^{-1}	19.0	23.7	18.4	2.1	510.0	24.5	150.40	9.07
pH		6.50	6.60	6.58	4.80	9.00	0.54	1.63	0.81
Alkalinity	mmol l^{-1}	0.66	1.00	0.69	< 0.01	11.0	1.01	16.96	3.24
Total hardness	mmol l^{-1}	0.62	0.76	0.60	0.07	15.00	0.69	165.46	9.09
Colour number	Pt mg l^{-1}	5	17.6	8.5	< 5	400	33.0	47.09	5.76
Turbidity	FTU	1.00	4.88	1.29	0.05	152.00	12.90	49.12	6.18
$KMnO_4$ consumption	mg l^{-1}	5	9.3	4.6	< 1	224	14.741	62.50	6.31
SO_4	mg l^{-1}	16	20.4	14.5	< 1	212	18.9	22.06	3.57
Cl	mg l^{-1}	7	17.3	7.1	< 1	1770	64.4	526.43	20.32
F	mg l^{-1}	< 0.1	0.28	0.12	< 0.1	4.7	0.54	15.36	3.64
NO_3	mg l^{-1}	5.20	12.56	3.46	< 0.1	233.0	20.0	32.48	3.64
NO_2	mg l^{-1}	< 0.01	0.026	< 0.01	< 0.01	8.00	0.26	850.11	27.97
NH_4	mg l^{-1}	< 0.1	0.18	< 0.1	< 0.1	33.0	1.4	346.95	8.43
Na	mg l^{-1}	7.10	12.52	7.65	1.30	446	27.64	135.66	10.34
K	mg l^{-1}	3.10	6.70	3.38	0.19	230.00	15.20	84.12	8.02
Fe	mg l^{-1}	0.17	0.68	0.20	< 0.05	42.0	2.09	164.18	10.67
Mn	mg l^{-1}	< 0.02	0.09	0.03	< 0.02	6.30	0.34	167.00	11.52
Al	mg l^{-1}	0.03	0.14	0.03	< 0.01	6.00	0.38	97.09	8.43

* standard deviations of $\log_{10}$ transformed TC, FC and FS are 0.99, 0.36 and 0.64, respectively.

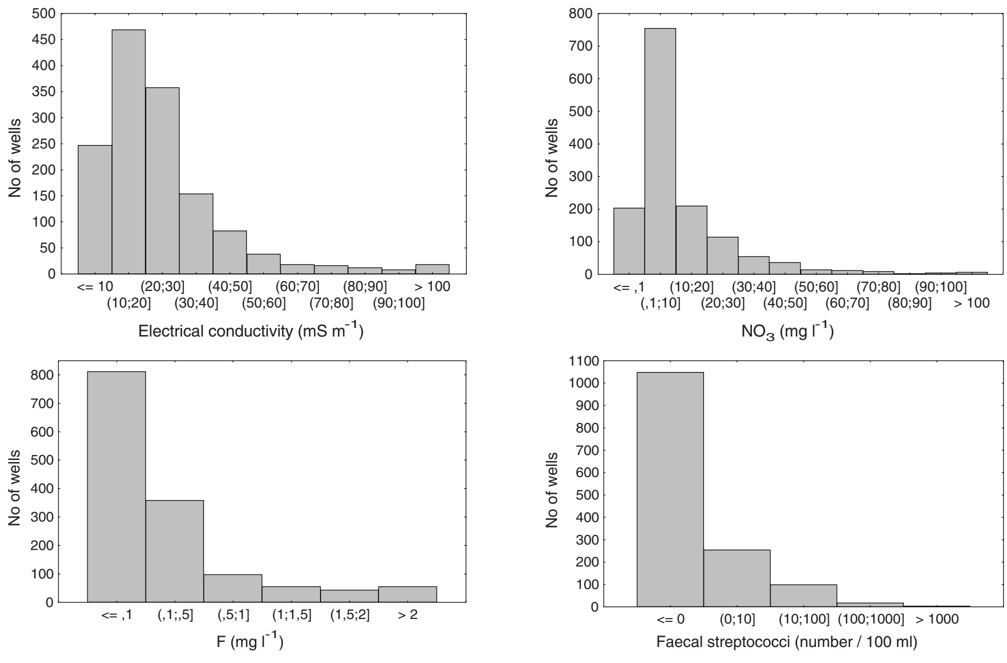


Fig. 8. Example histograms for variables of types I (electrical conductivity), II (nitrate), III (fluoride) and IV (number of faecal streptococci), demonstrating their distribution patterns.

Type I: normally or log-normally distributed variables

Hydrogeochemical variables are rarely normally distributed, and log-normally distributed data are more common in geology (Krumbein 1937). The log-normal distribution is distinguished from the normal distribution by its positive skewness and because its geometric mean equals the median (Rock 1988).

Some of the present variables were log-normally distributed, but not pH, which is intrinsically logarithmic and therefore followed a normal distribution curve. The variables having their geometric mean and median values close to each other belonged to this group (Table 5 and Fig. 9). The probability-probability plots show the log-normal distribution of these variables as well (Fig. 10), indicating that EC, alkalinity, total hardness, turbidity, SO_4, Cl, K and Fe are all log-normally distributed. The histogram of electrical conductivity is shown in Fig. 8 as an example of log-normal distribution. The histogram is positively skewed.

Type II: Variables with high kurtosis and a skewed distribution

The existence of data outliers is characteristic for this material, which means that many variables are highly skewed and not normally or log-normally distributed. Most of the present variables showed unimodality, with less than 30 % of cases below the detection limit, but with a large number of extreme values. Colour, $KMnO_4$ consumption, NO_3, Na and Al fall into this group (Tables 5 and 8). As the histogram for nitrate indicates, 10 % of the analysis were below the detection limit, small concentrations predominated and there were a lot of extreme values (Fig. 8).

The majority of advanced statistical methods and tests, which are based on the assumption that the data are normally distributed, are not suitable for use with the present data, and since normal distributions of the variables could not be obtained, non-parametric and robust statistical methods were preferred (Rock 1988, Ranta *et al.* 1991, Reiman and Filzmoser 1999). Log-transformed data are used throughout when performing robust parametric statistical tests such as principal component analysis, as was the case in other similar hydrogeochemical studies (e.g. Frapportti 1994) and has more recently been recommended by Reiman and Filzmoser (1999).

Type III: variables with many samples below the detection limits

The distributions of Mn, F, NH_4, NO_2, total coliform bacteria, faecal coliform bacteria and faecal streptococci are difficult to describe because a large proportion of the samples had concentrations

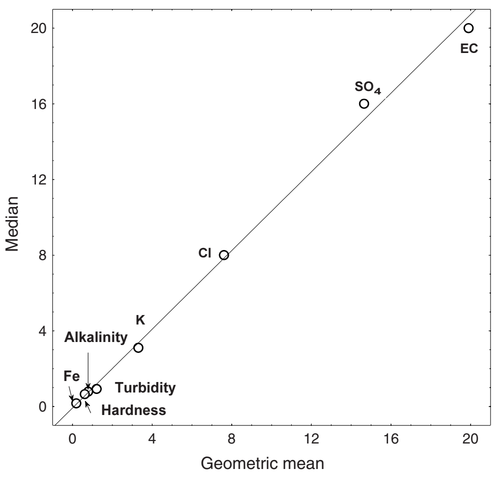


Fig. 9. Geometric means for EC, alkalinity, total hardness, turbidity, SO_4, Cl, K and Fe plotted against median values, displaying the log-normality of these variables.

Table 8. Analysis below the detection limits. The units are as in Table 1.

Variable	Detection limit	Frequency	%
Total coliformbacteria	1 / 100 ml	562	39.5
Faecal coliform bacteria	1 / 100 ml	1257	88.5
Faecal streptococci	1 / 100 ml	1047	73.7
Alkalinity	0.01	1	0.1
Colour number	5	317	22.9
$KMnO_4$ consumption	1	92	6.5
SO_4	1	19	1.3
Cl	1	75	5.3
F	0.1	654	46.0
NO_3	0.1	143	10.1
NO_2	0.01	1030	72.5
NH_4	0.1	1301	91.6
Fe	0.05	247	17.4
Mn	0.02	717	50.5
Al	0.01	404	28.4

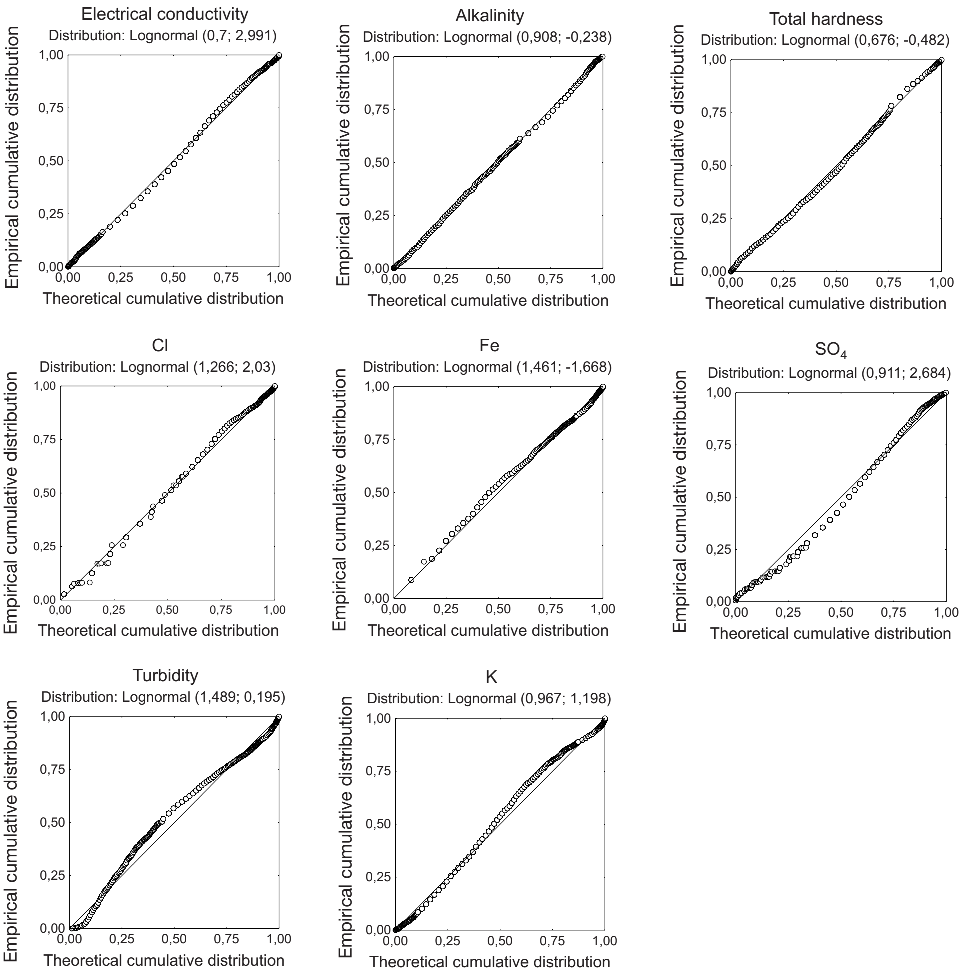


Fig. 10. Probability-probability plots of selected variables, indicating their log-normal distributions (N = 1421).

below the detection limit of the analytical method (Table 8) and thus a substantial number of samples could not be characterised by a true measured value. When a high proportion of the values are below the detection limit, the data will not reach a normal or log-normal distribution, as demonstrated by Reiman and Filzmoser (1999). Mn and F were variables affected in this way here. As seen in Fig. 8, the histogram for fluoride indicated that small concentrations predominated and that a significant number of values fell below the detection limit.

Type IV: Indicator variables

Nitrite, ammonium and bacteria are typical indicators of pollution (Table 5, Table 8 and Fig. 8), natural background concentrations of which should generally be below the detection limit, so that higher concentrations occur only on account of pollution. This is especially true of bacteria, in that it is more significant to know whether there are faecal bacteria in a water sample or not than to know how many bacteria can be counted in 100 ml.

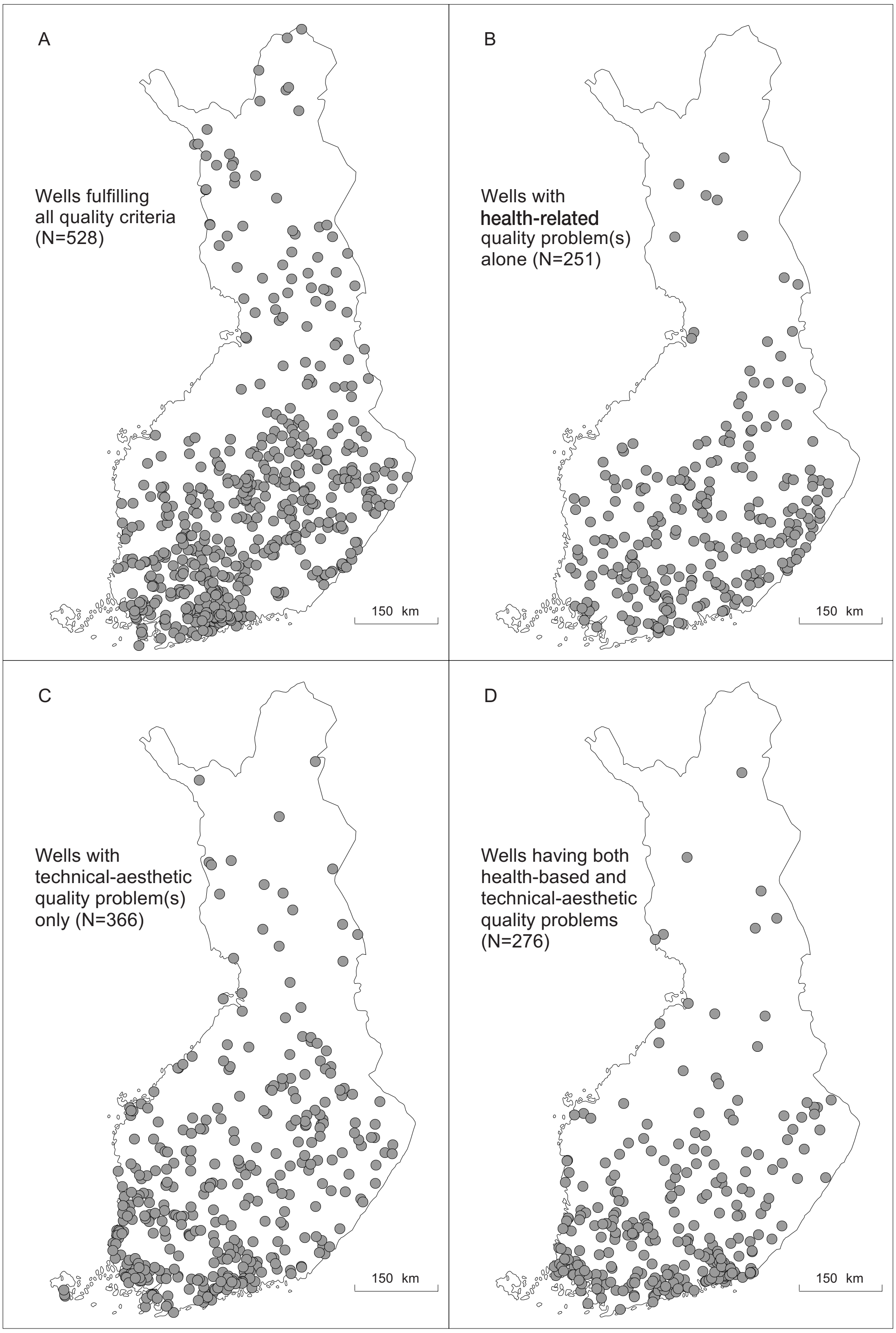


Fig. 11. Spatial distribution of water quality problems in wells in Finland.

3.3 Usability of well water for domestic purposes

3.3.1 General

When the usability of the well water was assessed by comparing the quality variables with the standards for potable water set by the Finnish health authorities (Ministry of Social Affairs and Health 1994, see Tables 3–4), only 528 wells were found to fulfil all the quality requirements and targets for drinking water , i.e. 37.2 % of those analysed (Table 9 and Fig. 11). This means that over 60 % had one or more quality constraints, which is a lot as compared with municipal groundwater intakes, where 33 % had quality constrains (Kujala-Räty *et al.* 1998). The number of wells which had health-related quality problems alone was 251, while 366 wells had one or more technical-aesthetic quality defects alone and 276 had both health-related and technical-aesthetic quality problems (Fig. 11).

Table 9. Well water quality in autumn 1990.

Variable	Concentrations exeeding recommended highest or lowest values	Number of wells	%
No defects		528	37.2
Health-related quality defect(s)		**527**	**37.1**
Total coliform bacteria	≥ 100 /100 ml	167	11.8
Faecal coliform bacteria	≥ 1 / 100 ml	164	11.54
F	> 1.5 mg l^{-1}	98	6.9
NO_3	> 25 mg l^{-1}	187	13.2
NO_2	> 0.1 mg l^{-1}	32	2.3
Technical-aesthetic quality defect(s)		**642**	**45.2**
Al	> 0.2 mg l^{-1}	194	13.7
NH_4	> 0.5 mg l^{-1}	29	2.0
Cl	> 100 mg l^{-1}	38	2.7
$KMnO_4$ consumption	> 20 mg l^{-1}	137	9.6
Mn	> 0.2 mg l^{-1}	151	10.6
Fe	> 0.5 mg l^{-1}	345	24.3
SO_4	> 250 mg l^{-1}	0	0
Na	> 150 mg l^{-1}	15	1.1
pH	< 6.0	104	7.3
Turbidity (FTU)	≥ 5	263	18.5
Colour number	≥ 20 mg l^{-1}	350	25.3

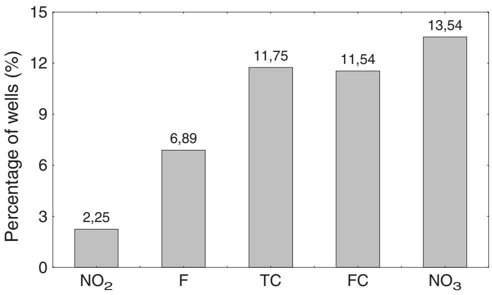


Fig.12. Percentages of wells which did not fulfil the health-related quality requirements in 1990.

Water quality was poorest in the wells situated along the south coast of Finland (Fig. 11), whereas most of those in northern and central Finland met the quality requirements and recommendations.

3.3.2 Health-related quality defects

Household water fulfilling all the health-related requirements was available for use in 63 % of the households in autumn 1990 (Table 9), i.e. there where 527 wells in which the water was inadequate from a health point of view. The most common quality defects are shown in Fig. 12. The nitrate concentration was too high in 13.5 % of the wells, total coliform bacteria in almost 12 %, faecal coliform bacteria in 11.5 % and the fluoride concentration in 7 %.

High concentrations of nitrogen compounds in groundwater, including nitrate and nitrite, which are classified as hazardous to health, are a big problem in many countries. Nitrate can be reduced in the human body to nitrite, which binds to haemoglobin in the blood and prevents it from liberating oxygen. The Finnish Ministry of Social Affairs and Health has consequently set the maximum concentration for nitrate in domestic water at 25 mg l^{-1} and that for nitrite at 0.1 mg l^{-1} (Ministry of Social Affairs and Health 1994). Lahermo and Backman (1999) have expressed concern over the possible coexistence of nitrates with pathogenic bacteria in groundwater. The present results pointed to a high nitrate content in inland wells more often than in the coastal areas (Fig. 13C), while it was only in a few wells inland (2.3 %) that

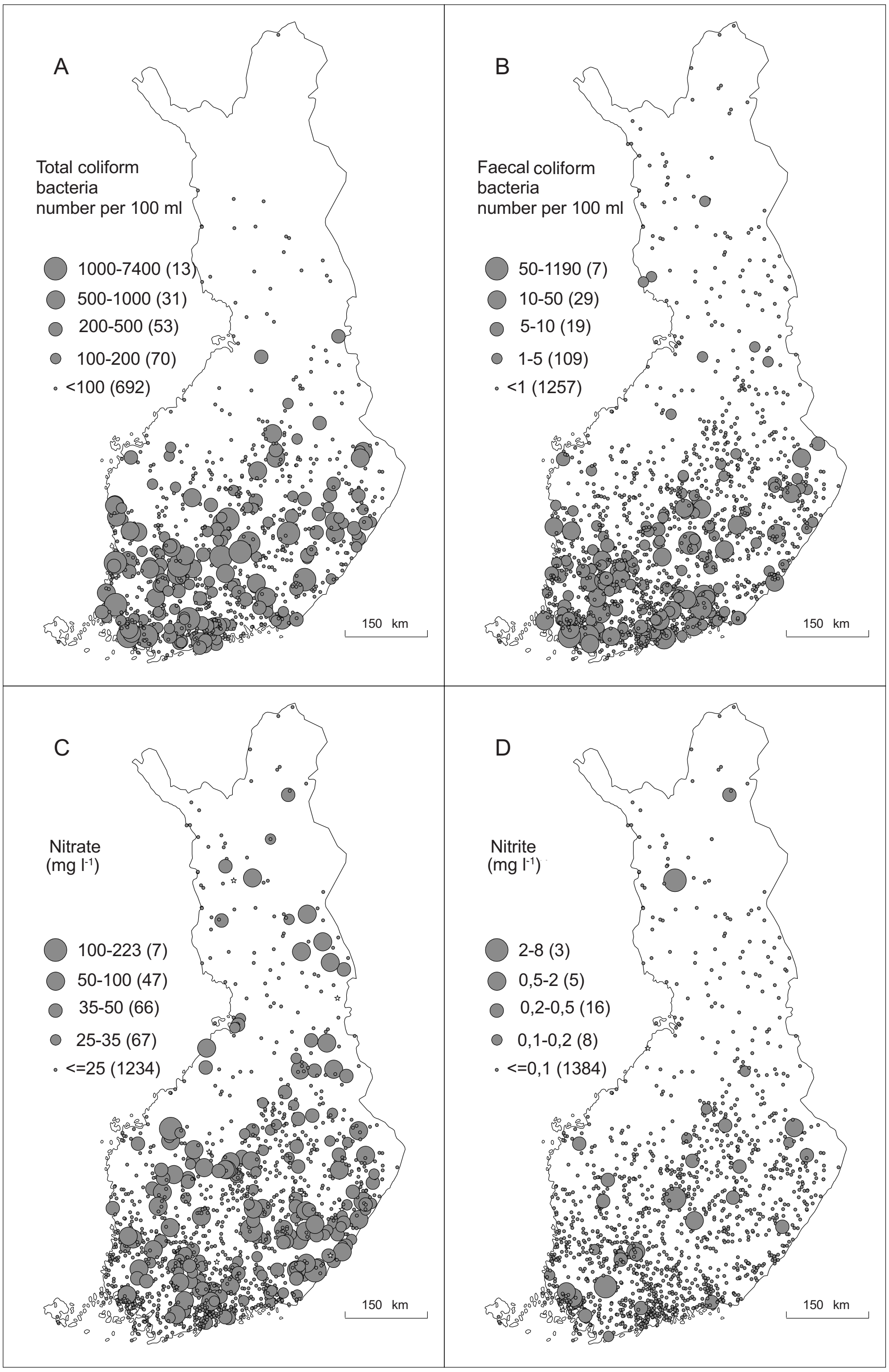

A
Total coliform bacteria number per 100 ml
1000-7400 (13)
500-1000 (31)
200-500 (53)
100-200 (70)
<100 (692)
150 km
B
Faecal coliform bacteria number per 100 ml
50-1190 (7)
10-50 (29)
5-10 (19)
1-5 (109)
<1 (1257)
150 km
C
Nitrate (mg l-1)
100-223 (7)
50-100 (47)
35-50 (66)
25-35 (67)
<=25 (1234)
150 km
D
Nitrite (mg l-1)
2-8 (3)
0,5-2 (5)
0,2-0,5 (16)
0,1-0,2 (8)
<=0,1 (1384)
150 km

Fig. 13 cont.

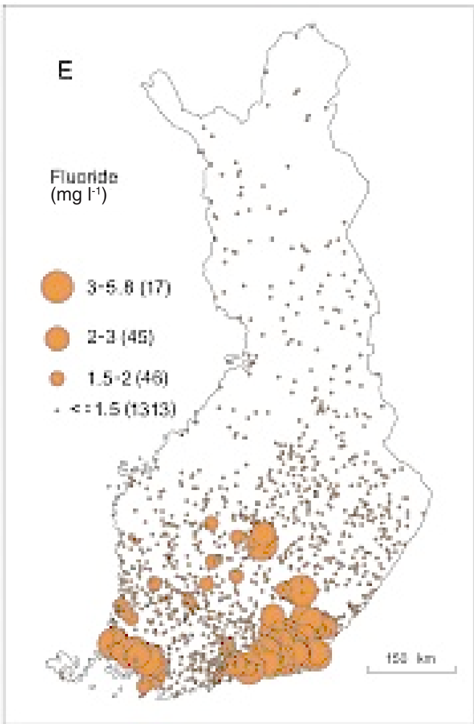


Fig. 13. Spatial distribution of wells in which the recommended maximum concentrations for A) total coliform bacteria, B) faecal coliform bacteria, C) nitrate, D) nitrite and E) fluoride were exceeded.

the nitrite concentration exceeded the stated maximum (Fig. 13D).

The health effects of groundwater having a high fluoride content have been documented in Finland in the form of dental fluorosis and skeletal fluorosis (Kurttio *et al.* 1999, Anon. 2000, Lahermo and Backman 2000). Dental fluorosis has been observed at an early stage in form of a slight mottling of the teeth, while regarding skeletal effects, Kurttio *et al.* (1999) observed that fluoride in drinking water increased the risk of hip fractures among women.

Fluoride is almost as usual problem in bedrock wells in Finland as in Norway, where water in 16.1 % of bedrock wells exceed the highest recommended concentration for drinking water (Anon. 2000, Midtgård *et al.* 1997). The present findings were that a high fluoride content seems to be a problem in limited areas in south-eastern and south-western Finland and in some wells in the more central parts of southern Finland (Fig. 13E). A connection between fluoride and rapakivi granite has been shown previously (Wäre 1967, Lahermo 1970, Rönkä 1983, Lahermo and Backman 2000), and high fluoride concentrations are certainly most common in bedrock wells, but they are to be found in other wells, too. The fluoride concentrations recorded here in the rapakivi areas of southern Finland exceeded the recommended maximum in almost every case.

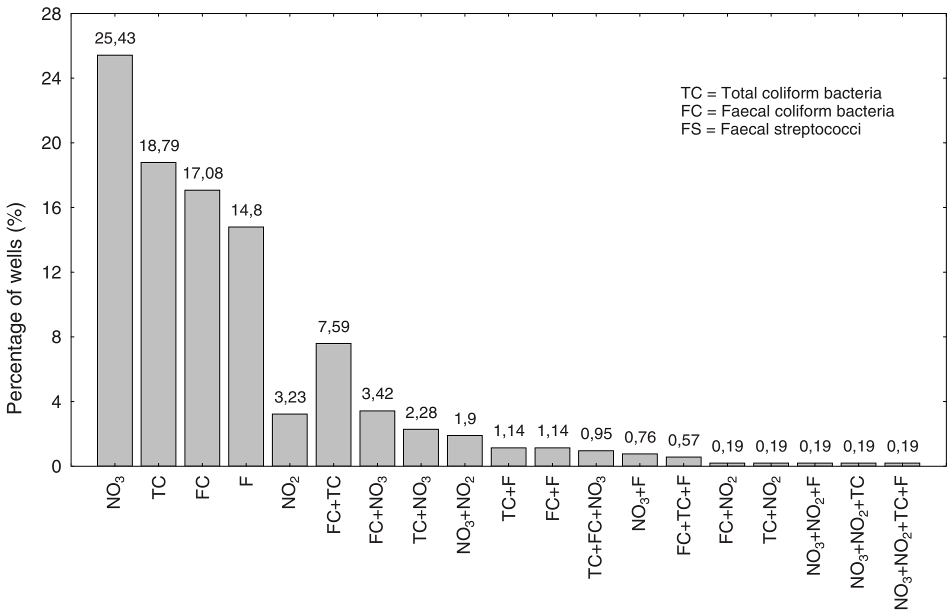


Fig. 14. Wells having health-related quality defects, by type of pollutant. The number of wells included in this graph is 527.

Safe drinking water needs to be free from microbial pathogens which cause human diseases and are known to contaminate groundwater. The predominant health complaint that result from microbially contaminated water is generalised acute gastrointestinal infection resulting in fever, diarrhoea and/or vomiting (Macler and Merkle 2000). Contamination of groundwater with pathogens is generally believed to result from the migration of faecal material into the subsurface. Faecal contamination can reach groundwater from various sources, such as leaking septic systems and sewers, cattle grazing or wild animals. Groundwater may be recharged with bacteria from the soil or vegetation and the implications of the occurrence of total coliform bacteria for human health are not direct (Howell *et al.* 1995, Niemi *et al.* 1997, Macler and Merkle 2000). The existence of bacteria is a problem, as shown in Fig. 13A and 13B, and one that is most severe in the wells of the more densely populated region of Southern Finland.

It can be seen from Fig. 14 that there is in most cases only one health-related quality defect in the wells that have quality problems, the most common individual pollutant, nitrate, explaining one quarter of the cases. Total coliform bacteria and faecal coliform bacteria were detected in the same water sample only in 8 % of the wells, but total coliform bacteria alone were observed in 19 % and faecal coliform bacteria in 17 %. Fluoride was the only health-related quality defect in 15 % of these problem wells.

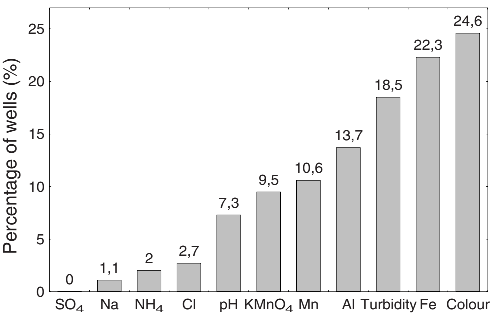


Fig. 15. Percentages of wells failing to fulfil the technical-aesthetic quality requirements in 1990. The number of wells is 1421.

3.3.3 Technical-aesthetic defects

There were 642 wells (45 %) in which the water was not technically or aesthetically satisfactory mainly because target values for colour, iron, turbidity, aluminium, manganese, $KMnO_4$ consumption or pH could not be attained (Fig. 15). None of the samples had an excessively high sulphate content, and only in a few cases were the concentrations of sodium, ammonium or chloride found to be too high. High iron concentrations in connection with high colour and turbidity values seemed to be the main technical-aesthetic water quality problems in Finnish wells.

Wäre (1960) and Natukka (1960, 1963) had already noted in the1950's and 1960's that the main water quality problems affecting private and municipal wells in Finland were generally acidity and high concentrations of CO_2, iron and manganese.

There were only a few wells, usually located in the coastal area, where the limit value for sodium was exceeded (Fig. 16G), indicating the marine origin of the water, and the chloride content rarely exceeded the limit (Fig. 16E). Chloride in groundwater can also be assumed to be predominantly of marine origin, either via salts in precipitation, from relict seawater trapped in the pores of the aquifer or from recent seawater intrusions. Other possible sources may be anthropogenic, e.g. from road salts, fertilisers, waste water or refuse dumps (Banks *et al.* 1998a, Lahermo *et al.* 1990).

Ammonium is not of direct importance for health in the concentrations to be found in drinking water, but it can indicate faecal contamination, cause taste and odour problems and result in nitrite formation or distribution (WHO 1996), so that the Ministry of Social Affairs and Health has set the maximum acceptable level of ammonium at 0.5 mg l^{-1}. Samples having a high ammonium concentration were rare in the present material (Fig. 16F).

The pH of the well water was occasionally above the recommended minimum value of 6, but never lower than 4.8 (Fig. 16A), while iron content, colour and $KMnO_4$ consumption often exceeded the standards in coastal wells (Fig. 16H, 16B and 16D). A high iron content will stain laundry and sanitary ware and spoil the taste of the water. According to WHO (1996), iron concentrations of 1-3 mg l^{-1} can be regarded as acceptable for people drinking anaerobic well water. High manganese concentrations are more equally distributed over southern Finland (Fig. 16I). Manga-

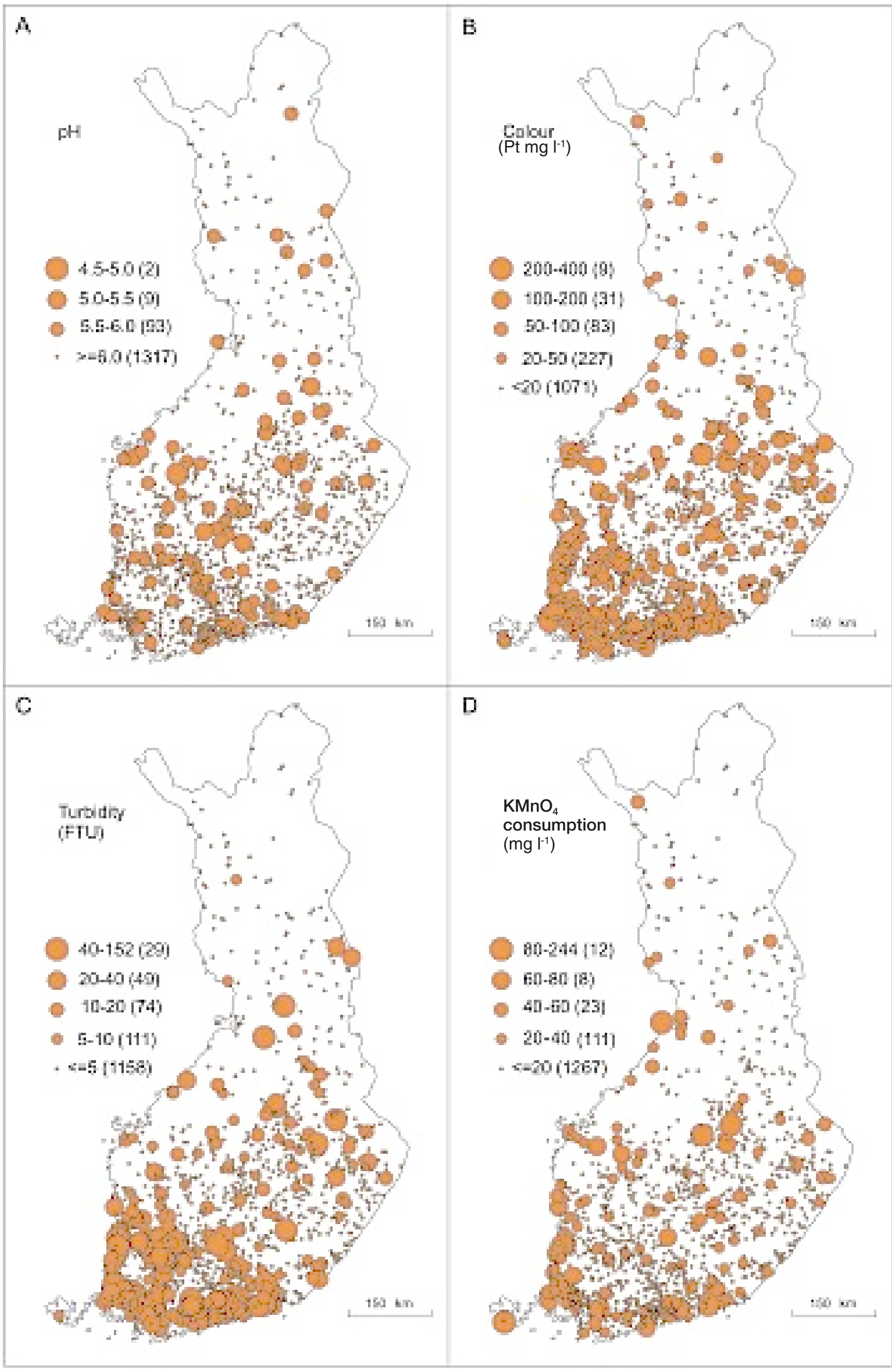

A
pH
4.5-5.0 (2)
5.0-5.5 (9)
5.5-6.0 (93)
>=6.0 (1317)
150 km
B
Colour
(Pt mg l-1)
200-400 (9)
100-200 (31)
50-100 (83)
20-50 (227)
<20 (1071)
150 km
C
Turbidity
(FTU)
40-152 (29)
20-40 (49)
10-20 (74)
5-10 (111)
<=5 (1158)
150 km
D
KMnO4
consumption
(mg l-1)
80-244 (12)
60-80 (8)
40-60 (23)
20-40 (111)
<=20 (1267)
150 km

Fig. 16 cont.

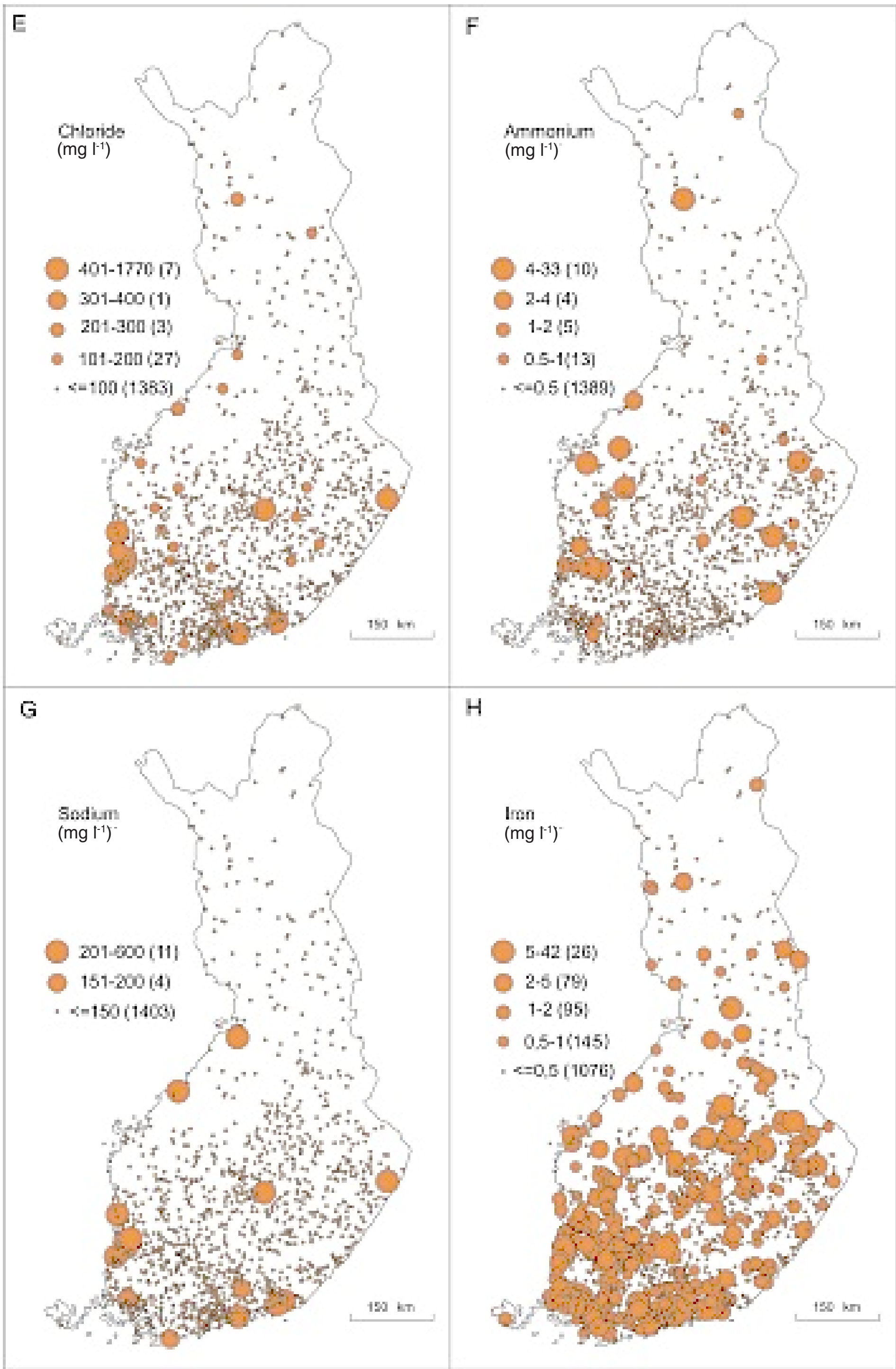

E
Chloride
(mg l-1)
401-1770 (7)
301-400 (1)
201-300 (3)
101-200 (27)
<=100 (1383)
150 km
F
Ammonium
(mg l-1)
4-33 (10)
2-4 (4)
1-2 (5)
0.5-1(13)
<=0.5 (1389)
150 km
G
Sodium
(mg l-1)
201-600 (11)
151-200 (4)
<=150 (1403)
150 km
H
Iron
(mg l-1)
5-42 (26)
2-5 (79)
1-2 (95)
0.5-1(145)
<=0.5 (1076)
150 km

Fig. 16 cont.

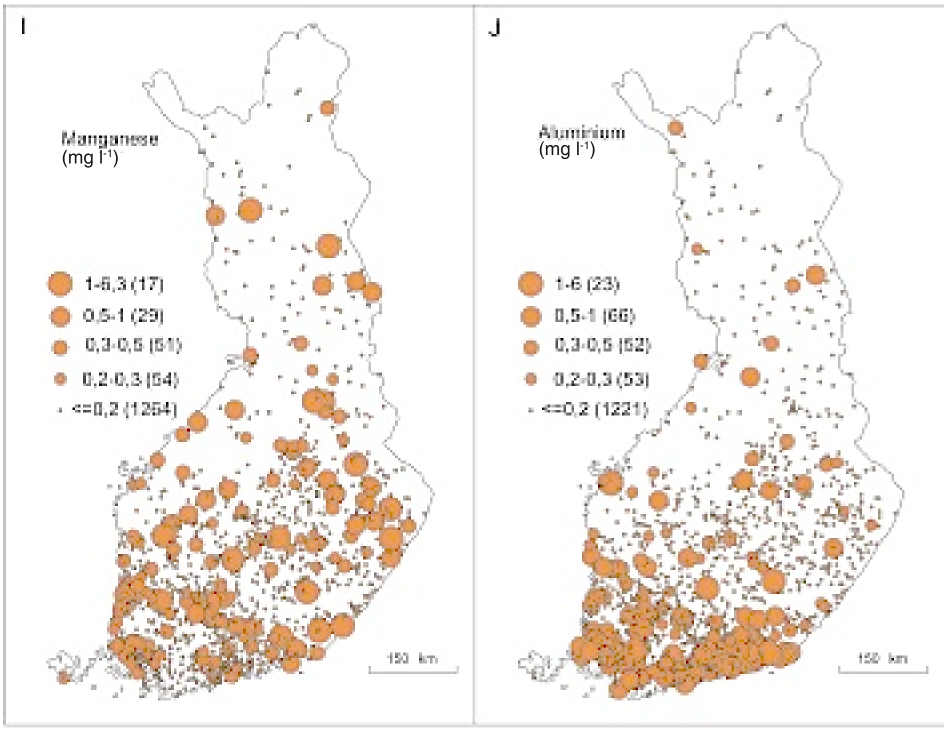


Fig. 16. Spatial distribution of wells in which the limit values for A) pH, B) colour, C) turbidity, D) $KMnO_4$ consumption, E) chloride, F) ammonium, G) sodium, H) iron, I) manganese and J) aluminium were exceeded (dark dots). Small dots represent wells in which the limit values were not exceeded.

nese may cause harm to consumers through taste problems, discoloration of laundry and sanitary ware and water turbidity (WHO 1996).

Turbidity and aluminium concentrations are likewise especially high in wells situated along the south coast (Fig. 16C and 16J). Aluminium content of groundwater is generally 0.01-0.1 mg l^{-1} (Matthess 1982) due to its low solubility in water at near-neutral pH. The solubility of aluminium is highly dependent on pH (Drever 1997), leading Hem (1992) to explain that reported concentrations of 1.0-mg l^{-1} or more in water at near-neutral pH probably represent particulate material.

The present results suggest that well water quality is more often lower on the coasts than in Central Finland or in Lapland (Fig. 11). This coastal effect was observed by Natukka (1963), for example, who found the best groundwater quality in the eskers of Central Finland and the poorest quality in the esker aquifers of the western coastal area.

3.3.4 Comparison of dug wells and bedrock wells

A comparison of the water quality in dug wells (including springs) associated with surficial sedimentary aquifers with that in bedrock wells showed that the health-related quality criteria were achieved better in all respects except for fluoride in the bedrock wells than in the dug wells (Fig. 17a). In the case of nitrate and the bacterial count, the proportion of the dug wells having a quality problem was about twice as high as for bedrock wells, and the same held true for technical-aesthetic quality defects (Fig. 17b). There are more problems involved in meeting the target values in the case of dug wells for all variables other than manganese, chloride and sodium. The observation that manganese is a greater problem in bedrock wells is likely to be ascribable to the reducing conditions prevailing in deep bedrock aquifers, which mobilise man-

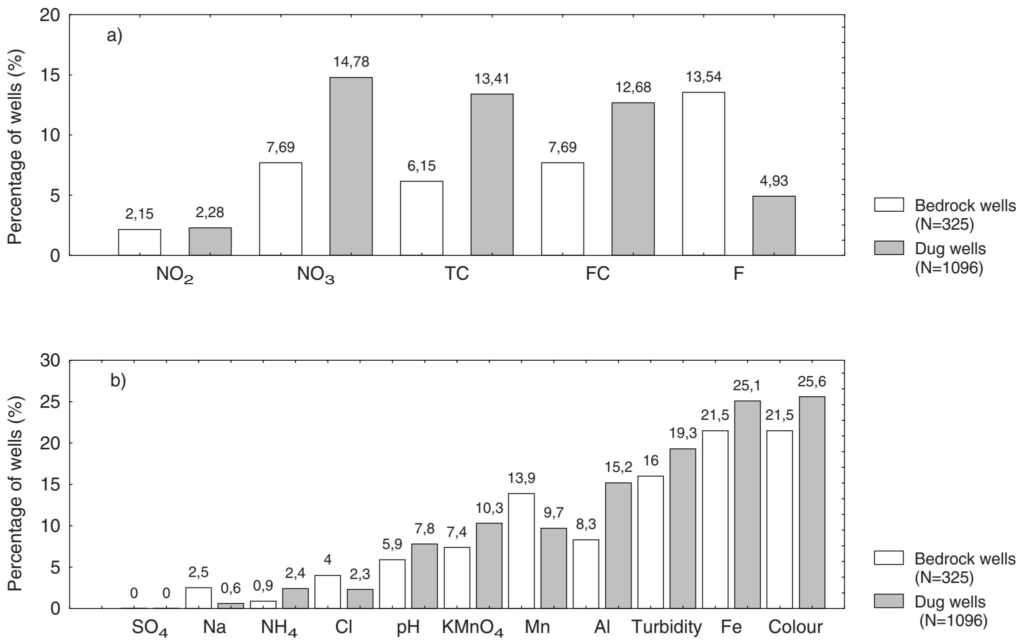


Fig. 17. Comparison of water quality problems in1990 between dug and bedrock wells. Columns describe the proportions of the wells in which the health-related and technical-aesthetic quality requirements were not fulfilled.

ganese in the Mn(II) stage. Similar results were reported by Banks *et al.* (1998b) regarding aquifers in Norway.

3.4 Extra-regional effects

3.4.1 Geochemistry of water in dug wells versus deeper bedrock wells

Extra-regional effects stem from geological factors, as discussed in Chapter 1, but include marine influence as well. In this chapter it will be considered whether the location of wells in bedrock or in surficial sediments has any effect on water quality.

The data were subdivided into two categories, wells dug into unconsolidated sediments and drilled bedrock wells, in order to characterise the two types of aquifer. To clarify the differences between these groups, cumulative frequency distribution curves for 21 variables are provided for dug wells and bedrock wells in Fig. 18. For all the variables (except for pH, which is already a log-transformed variable) a log scale is used on the x-axis. In these graphs data falling below the analytical detection limits are plotted at values corresponding to half of the detection limit in each case.

The number of log cycles on the x-axis gives an impression of the variance of the variables. There are obviously differences in chemical composition between the groundwater in bedrock and in sedimentary aquifers.

The pH, conductivity and alkalinity values were higher in the bedrock water than in the water from dug wells, suggesting greater mineralisation as a result of longer residence times, and F and Na concentrations were also enriched in the bedrock groundwater. The same can be seen when comparing results from different surveys of groundwater quality (e.g. Laakso 1966, Rönkä 1983, Hyyppä 1984, Lahermo *et al.* 1990, Mäkelä 1990). The average EC value obtained here for water from bedrock wells (arithmetic mean = 33.3 mS m^{-1}) was higher than reported by Laakso (1966) or Rönkä (1983), but similar to that observed by Lahermo *et al.* (1990). On the other hand, the average F and Na concentrations in the bedrock wells (F: 0.68 mg l^{-1}, Na: 32.7 mg l^{-1}) was higher than that observed by Lahermo *et al.* (1990) (F:0.42 mg l^{-1}, Na: 25.1 mg l^{-1}) possibly due to the different sampling network.

The present material is in fact dominated by wells located in southern Finland.

In the case of hardness and SO_4 concentrations, the distributions for the two well types were about the same, and the Mn and Cl concentrations were only slightly higher in the bedrock wells. This may be explained either by the lithological sources and relict marine salts in the bedrock waters or by old marine salts and anthropogenic contamination in the dug wells. This result is not perfectly in line with previous studies, where higher Mn and Cl concentrations were reported in bedrock wells than in shallow wells (Pönkkä 1981, Rönkä 1983, Lahermo *et al.* 1990). The Cl obtained here (29.1 mg l^{-1}) were lower on average than those reported by Lahermo *et al.* (1990) (36.6 mg l^{-1}), but higher than in the data of Rönkä (1983).

The Fe, K, turbidity, colour, $KMnO_4$ consumption and Al variables exhibit higher concentrations in dug wells than in bedrock wells. Most of these variables indicate the influence of surface water rich in organic matter and/or clay particles. Mäkelä (1990) reports generally higher Fe concentrations in bedrock aquifers than in Quaternary aquifers, and the divergent results obtained here are probably due to the different spatial distribution of sampling points. The number of shallow wells receiving their water from coastal, clay-covered aquifers representing an anaerobic groundwater environment is high in the present material, by contrast to the findings of Mäkelä (1990), which are all based on sampling points in Central Finland.

The concentrations of nitrogen compounds (NO_2, NO_3, NH_4) are generally fairly low, although some higher figures are recorded, and the concentrations are higher in well water representing sedimentary aquifers than in bedrock wells, indicating the vulnerability of shallow groundwater. Mäkelä (1990) arrived at similar conclusions. The presence of bacteria (TC, FC and FS) is more common in wells draining shallow sedimentary aquifers than in bedrock wells, although there is evidence of pollution in a number of bedrock wells, too.

3.4.2 Bedrock type

The main factor affecting water quality in wells drilled into bedrock seems to be the rock type (Lahermo 1970, Hyyppä 1973, Pönkkä 1981, Rönkä 1983) and the mode of weathering of the particulate minerals (Rönkä *et al.* 1981). The bedrock of Finland belongs to the Precambrian Fennoscandian Shield and is composed of Archaean and Proterozoic igneous and metamorphic rocks (Fig. 19). The Archaean basement complex in eastern and northern Finland comprises intermediate and felsic granitoids, gneisses and migmatites, and a greenstone belt. This complex consists mainly of slowly weathering rocks, and only the greenstone belt, being composed of metavolcanic and metasedimentary rocks, contains more basic components and alkali earth and metal cations (Koljonen 1992, Lahermo *et al.* 1996, Lehtinen *et al.* 1998).

There are large, relatively uniform Svekokarelian granitoid complexes in Central Finland and Lapland (Fig. 19), and other, smaller granitoid plutons occur within the schist belt. These granite areas are geochemically the least reactive in Finland (Koljonen 1992).

The granitoids of Central Finland are surrounded by the Svekofennian and Karelian schists and gneisses (Fig. 19). These consist mainly of metasedimentary rocks (such as schists and migmatites) and metavolcanics, being connected locally with sulphide ores and sulphide mineral occurrences. There are both felsic and mafic plutonic rocks in the schist area, releasing larger amounts of Ca, Mg, Na and K than in other bedrock areas (Koljonen 1992, Lahermo *et al.* 1996).

Rapakivi granites are found over large areas of Southeast and Southwest Finland and in the Åland archipelago (Fig. 19). The rapakivi granite batholiths are characterised by high K concentrations and low concentrations of Ca, Mg, Na and Fe. The F concentration (up to 0.23 %) is much higher than in other bedrock types, and this is reflected in the chemistry of the surface water and groundwater in these areas (Koljonen 1992, Laakso 1966, Lokka 1950, Lahermo *et al.* 1990, Lahermo and Backman 2000).

In northern Finland there are two large bedrock units: the Lapland granulite belt and the Central Lapland greenstone belt. According to Lahermo *et al.* (1996), the Peräpohjola and Kuusamo schist belts are the best-buffered areas in Finland as far as soil and water are concerned. This is due to their mafic and ultramafic rocks and local limestone deposits (Lahermo *et al.* 1996). The youngest rock units found in Finland are small deposits of Jotnian sedimentary rocks (sandstone and claystone) in western Finland and a small slice of Caledonian rocks in Lapland.

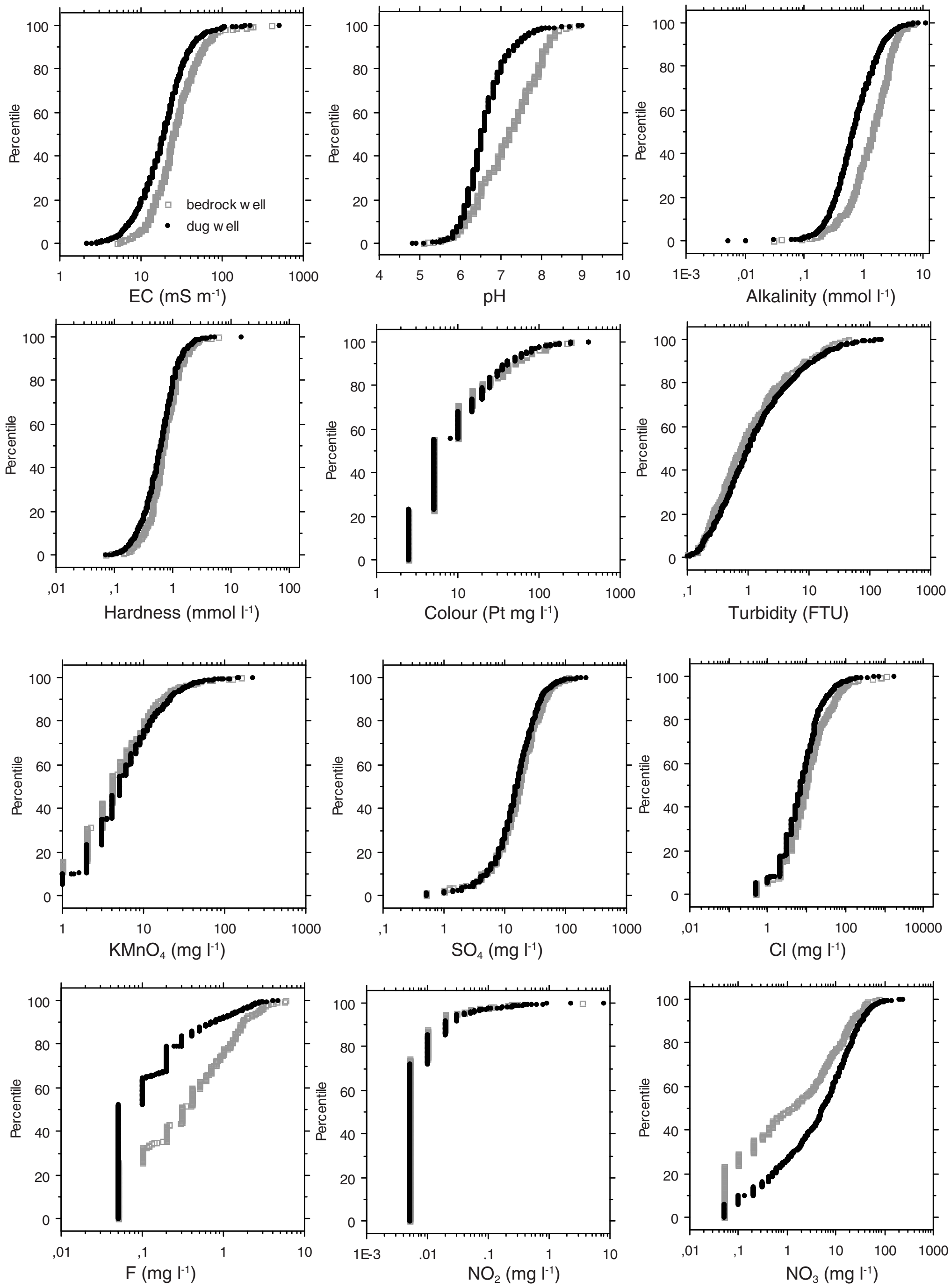

Percentile
bedrock well
dug well
EC (mS m-1)
pH
Alkalinity (mmol l-1)
Hardness (mmol l-1)
Colour (Pt mg l-1)
Turbidity (FTU)
KMnO4 (mg l-1)
SO4 (mg l-1)
Cl (mg l-1)
F (mg l-1)
NO2 (mg l-1)
NO3 (mg l-1)

Fig. 18 cont.

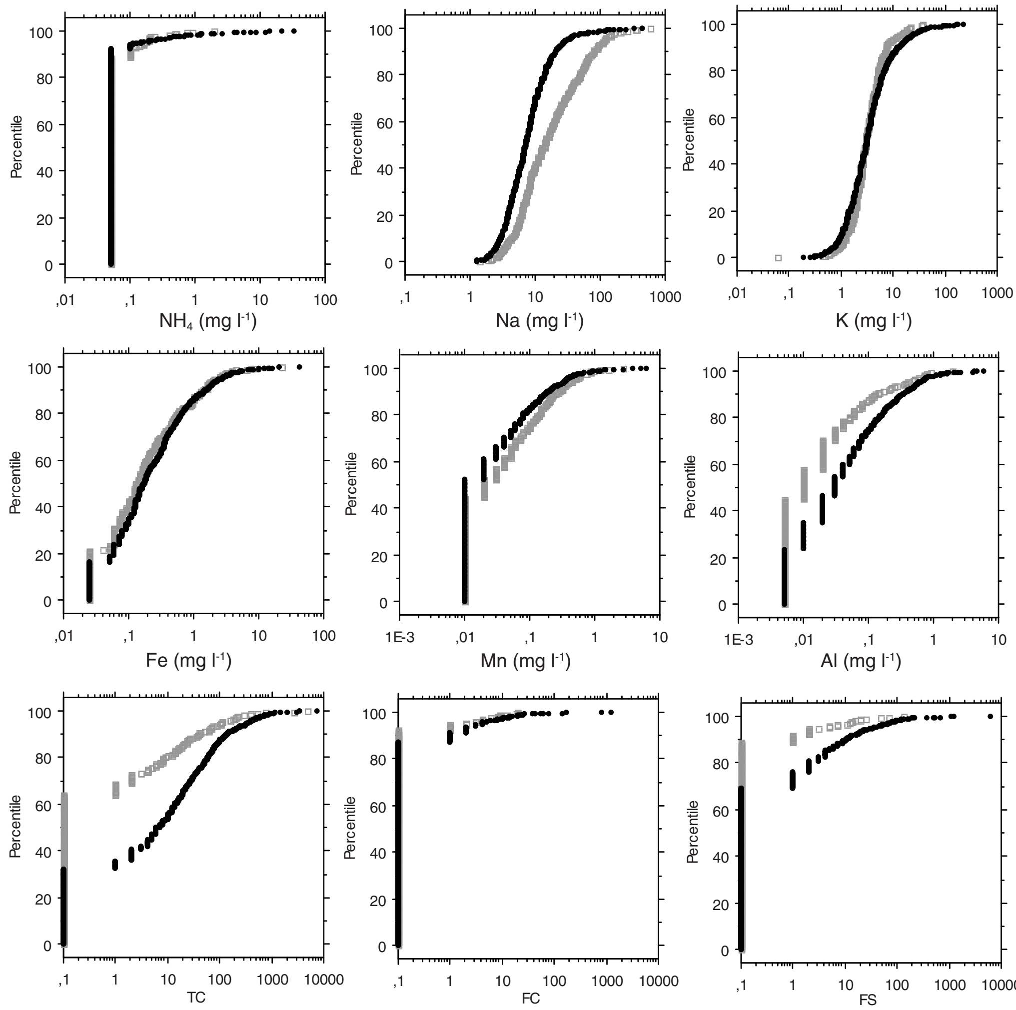


Fig. 18. Cumulative plots of water quality variables on a logarithmic scale for comparison of the distributions between dug wells and bedrock wells.

More than half of the wells (54 %) were situated in areas dominated either by rocks of the gneiss/schist group (mica gneiss, vein gneiss, gneiss, mica schist, greywacke, phyllite, migmatite) or by members of the TTG series (tonalite, trondhjemite, granodiorite, quartz diorite, granite gneiss, gneiss granite) and one third in granitic areas, of which 136 were in rapakivi granite areas and 91 in S-type granite (microcline granite) areas. There were 77 wells in areas with intermediate or felsic volcanic rocks (amphibolite, tuffite, leptite, hornblende gneiss, potassium feldspar gneiss, potassium feldspar schist), but only 15 in Jotnian claystone or sandstone areas. The distribution of the bedrock wells into these bedrock groups followed closely the distribution of dug wells in the material (Fig. 20).

The composition of the bedrock influences not only the chemical composition of the groundwater in the fissures and fractures, but also that of the groundwater in the Quaternary deposits (Lahermo 1970, Pönkkä 1981, Banks *et al.* 1998 b).

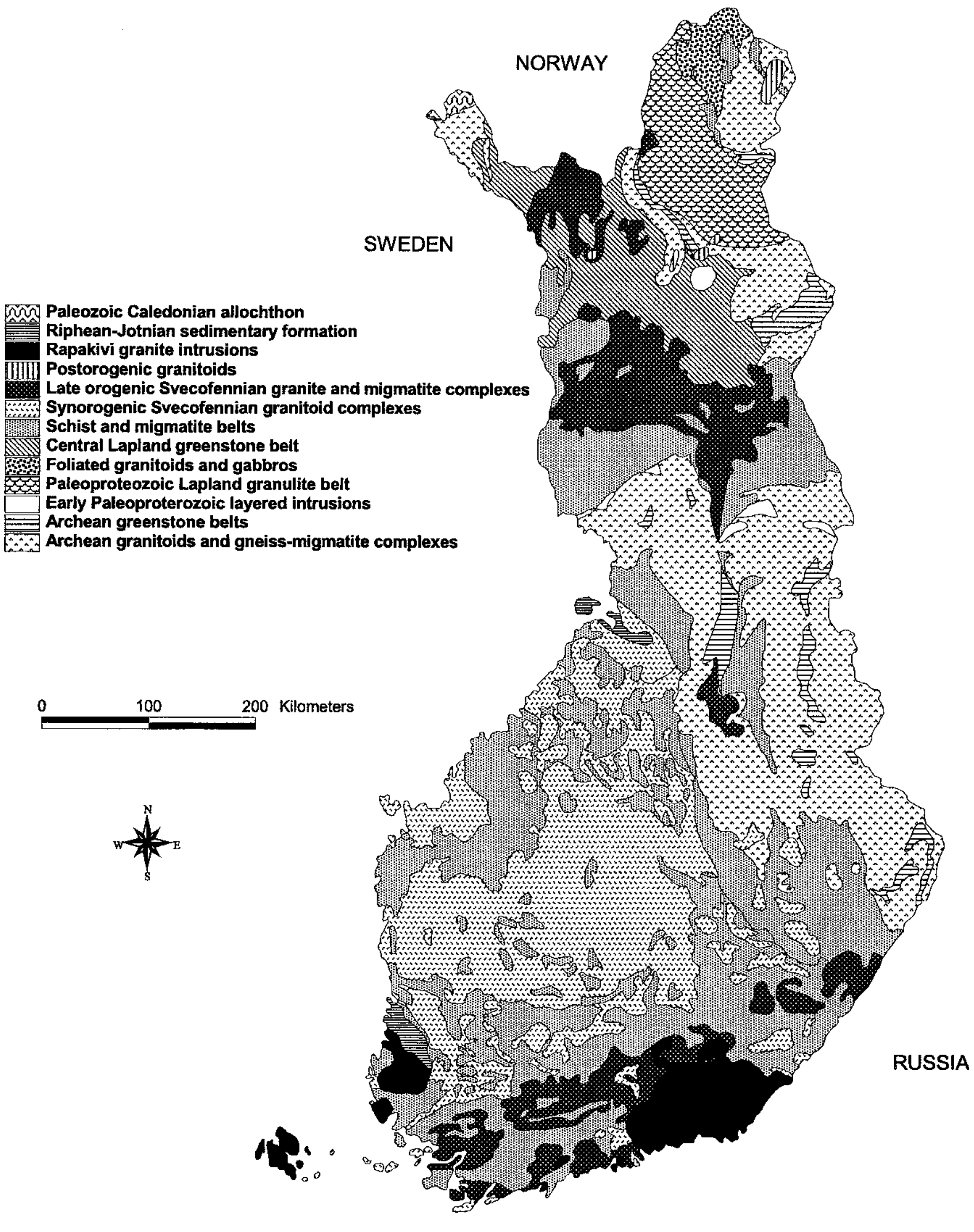


Fig. 19. Generalised map of the Precambrian bedrock of Finland. Modified after Korsman *et al.* (1997) by Karro and Lahermo (1999).

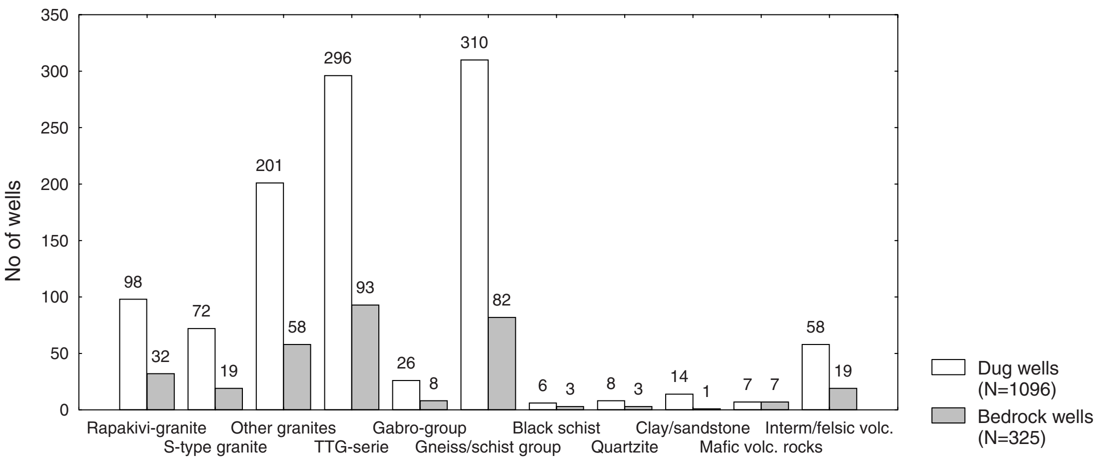


Fig. 20. Shallow dug wells and drilled bedrock wells, by bedrock type (lithological group).

Glaciofluvial material, being derived from various distances, represents a mixture of local rocks (Virkkala 1958), and for this reason, and because of the low amounts of fine-grained fractions present in the material, the electrolyte content of groundwater discharging from sorted material does not reflect the composition of the local bedrock as closely as does the groundwater in till deposits, which better represent the local rock complex. A number of the variables studied here reflect the dependence of the groundwater chemistry on the lithology, especially in the case of the bedrock wells, but also to a minor extent in the dug wells.

Electrical conductivity was lowest in the wells belonging to the groups of other granites, the TTG series and quartzite (Fig. 21). Hyyppä (1984) observed earlier that the groundwater in Finland containing the highest concentrations of dissolved solids was to be found in the areas of rapakivi granite and basic rocks, while that with the lowest electrolyte concentrations was related to quartzitic and granitic bedrock areas.

The lowest pH values seemed to be found in the bedrock wells drilled into rapakivi granite or in areas belonging to the other granite group (Fig. 21), as also observed by Rönkä (1983), who found that the average pH of the groundwater contained in mafic bedrock was 7.0, that in areas composed of felsic plutonic rocks and silicic schists was 6.8 and that in rapakivi granite areas was 6.3.

The alkalinity of the water was highest in the S-type granite bedrock aquifers. The differences in water acidity between the bedrock groups were nevertheless much smaller among the wells dug into Quaternary sediments.

The chloride content was generally higher in the rapakivi areas than in the other bedrock wells (Fig. 21), but this could be explained by their location in the coastal regions and under marine influence. The results reported by Rönkä (1983) are similar, in that he also found that chloride concentrations in groundwater derived from mafic rocks were twice as high as those concentrations in groundwater from felsic rocks. It has been observed previously (Lahermo 1970, Mäkelä 1990, Lahermo *et al.* 1990) that in addition to chloride, the concentrations of ammonium and other salts are higher in the bedrock water of the rapakivi area than in central and northern Finland.

Aluminium and fluoride concentrations seem to be closely connected with the rapakivi areas in the case of both bedrock wells and dug wells (Fig. 21 and Fig. 22). Fluoride concentrations are also somewhat higher in some wells in S-type granite areas. The connection between high fluoride concentration in the groundwater and the presence of rapakivi granite has been demonstrated earlier by Wäre (1960), Natukka (1963), Laakso (1966), Lahermo (1971), Rönkä (1983), Soveri (1985) and Lahermo *et al.* (1990) and recently by Lahermo and Backman (2000). In the present material, the median and mean fluoride concentrations in all the wells located in rapakivi granite (N = 130) were both 1.6 mg l^{-1}, while those in bedrock wells in rapakivi areas (N = 32) were 1.85 mg l^{-1} and 2.1 mg l^{-1}, respec-

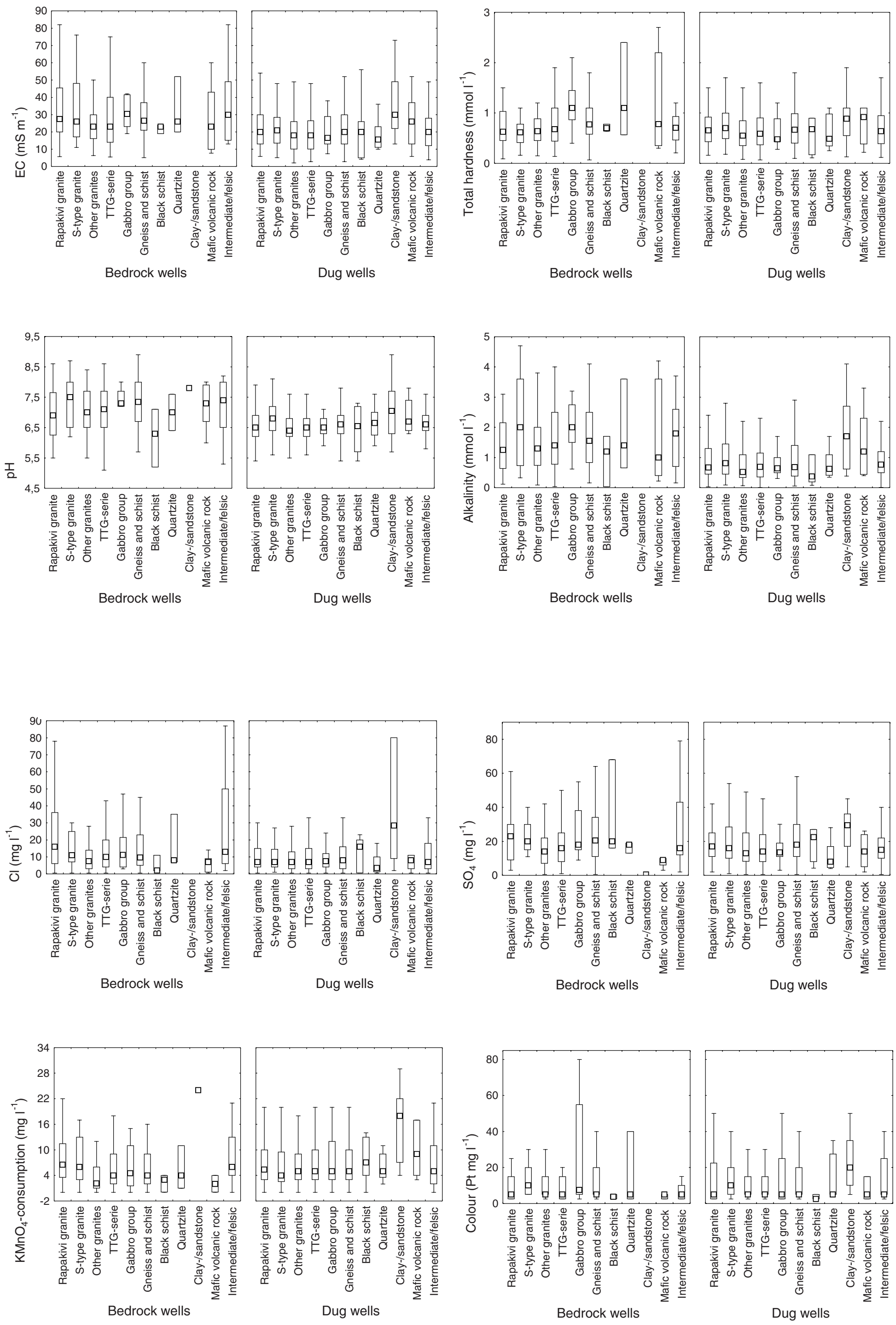

EC (mS m⁻¹)
Total hardness (mmol l⁻¹)
pH
Alkalinity (mmol l⁻¹)
Cl (mg l⁻¹)
SO4 (mg l⁻¹)
KMnO4-consumption (mg l⁻¹)
Colour (Pt mg l⁻¹)
Rapakivi granite
S-type granite
Other granites
TTG-serie
Gabbro group
Gneiss and schist
Black schist
Quartzite
Clay-/sandstone
Mafic volcanic rock
Intermediate/felsic
Bedrock wells
Dug wells

Fig. 21 cont.

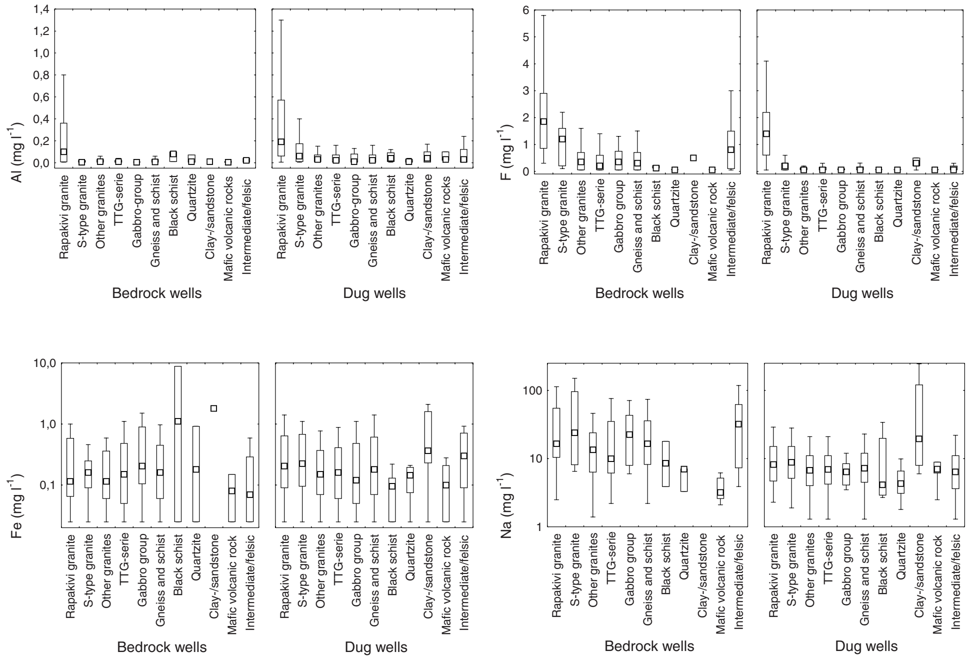


Fig. 21. Box plots indicating the distributions of water quality variables in groundwater from bedrock wells and dug wells, classified by underlying rock type. (Median; Box: 25 %, 75 %; Whiskers: 10 %, 90 %)

tively. Rönkä (1983) arrived at a comparable average fluoride concentration in bedrock wells in the Vehmaa rapakivi area (2.0 mg l^{-1}).

The wells in rapakivi granite areas can be further subdivided according to the type of rapakivi concerned (Fig. 22): Bodom granite, pyterlite, homogeneous, even grained rapakivi granite and viborgite. A significant difference in well water quality occurs only in the case of Bodom granite areas, however, where fluoride concentrations are lower than in wells in other rapakivi areas, in line with the findings of Lahermo and Backman (2000).

The fluorine concentration in rapakivi granites is 0.20–0.42 % and that in other granites 0.05–0.14 % (Sahama 1945, Simonen and Vorma 1969, Vorma 1976). In other igneous and metamorphic rocks the range is as low as 0.01–0.05 %. One of the most fluorine-rich components is fluorite, which commonly occurs as an accessory mineral in rapakivi granite. Fluorine is also found in apatite, biotite and the amphiboles, however, which are commonly present in rapakivi and other granites, as also in topaz and tourmaline (Rankama and Sahama 1950).

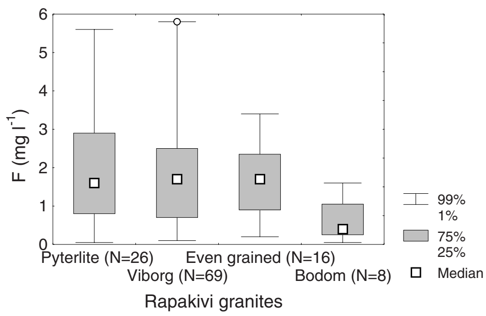


Fig. 22. Fluoride concentrations in bedrock water related to the groups of rapakivi granites.

Fluoride showed significant ($p < 0.01$) but moderate correlations (Spearman rank coefficient) with alkalinity ($\rho = 0.45$) and Na ($\rho = 0.46$), weak positive correlations with EC ($\rho = 0.35$) and pH

($\rho = 0.39$) and a negative one with NO_3 ($\rho = -0.33$). The significant correlation with sodium may indicate that this has a role in enhancing the solubility of minerals such as fluorite. According to Boyle (1992) base cation exchange processes (Ca and Mg with Na) often contribute to a rise in fluoride concentrations. Another explanation for the correlation is the existence of rapakivi areas in the vicinity of the sea. The fluoride content of the present seawater is as high as 1.2–1.4 mg l^{-1}, but is markedly lower in the Baltic Sea (Lahermo and Backman 2000). Rönkä (1983) and Soveri (1985) observed a strong positive correlation (Pearson) between fluoride concentration and pH (Soveri 1985; $\rho = 0.78$).

The high aluminium concentrations recorded in southern Finland cannot be explained by the till geochemistry. The highest values for the aluminium content of till have been observed in the granulite zone of Lapland, whereas there is not much aluminium enriched in till in the rapakivi granite areas, which have the lowest aluminium concentrations of all rock types (Koljonen 1992). Al-F complexation is thought to contribute to the high Al and F concentrations in rapakivi areas (Lahermo *et al.* 1996). According to Wenzel and Blum (1992), the high solubility of fluoride in an acid environment may be explained by occurrence of soluble $[AlF]^+$ and $[AlF_2]^+$, while Haidouti (1995) suggests that the main cause of Al mobilisation is the presence of fluorides, which remove previously stable Al hydroxides. Stable soluble fluoride complexes are formed mainly at pH < 6 (Hem 1992), but the pH of the groundwater in the wells where high Al and F concentrations occur is not always under 6. Karro (1999) suggests that the low pH values (4.4–4.6) of precipitation and snow meltwater may contribute to the effectiveness of F leaching from the unsaturated zone into the groundwater in the form of Al and Fe complexes.

There were only three wells drilled into black schist in the present series, and the number of dug wells in such areas was also low (9). In spite of the low number of cases, however, this category differs from all the others, having lower than average alkalinity and pH and higher iron and sulphate concentrations (Fig. 21). This may be connected with the high concentration of sulphide minerals in these rocks and the sediments derived from them (Loukola-Ruskeeniemi 1999). Lahermo (1970) has observed earlier that the groundwater of subsilicic greenstone and black schist areas in Central Lapland contains many times more electrolytes than that of granite areas.

3.4.3 Quaternary history of the area

Late-glacial and post-glacial clay and silt deposits are common in the coastal areas of Finland, which have emerged from the regressive phases of the Baltic Sea. It is assumed that relict sea water trapped as pore water in the marine clay and silt sediments during the last deglaciation contributes to the electrolyte content of the groundwater in these sediments (Lahermo *et al.* 1990). As will be seen in the next chapter, the well water obtained from the clay-covered aquifers in the present material differs from the water in the other types of aquifer, but there are also significant differences in water quality between the groups of 'Litorina clay' and 'other clays'. In that colour, turbidity, $KMnO_4$ consumption, SO_4 ,Cl, F, Na, K, Fe and Al concentrations were higher in the wells dug through Litorina clay than in wells in other clay areas (Fig. 23). It has been estimated by Eronen *et al.* (1979) that the Litorina Sea phase (about 7500–5000 years ago) was more saline (5 ‰ – 10 ‰) than the present Baltic Sea. The presence of acid sulphate soils along the coast of western Finland may be another explanation for the different water quality in this group. Palko (1994) has estimated the area of acid sulphate soils to be as large as 3360 km^2, and Åström and Björklund (1995) found that a high incidence of acid fine-grained sediments in a catchment will result in an increase in the concentrations of Al, Ca, K, Mg, Mn, Na and SO_4 etc. in the stream water. Lahermo *et al.* (1994) have shown in the same area that the sulphur and sulphate concentrations in stream sediments and the sulphate concentrations in groundwater have similar spatial distributions.

3.4.4 Atmospheric deposition and distance from the seashore

A notable proportion of the chlorides, nitrates, potassium, sodium and sulphate in groundwater and surface water may be connected with precipitation (Lahermo 1970, Lahermo *et al.* 1990). The concentrations of certain elements in rain water vary depending on the density of settlement, industry,

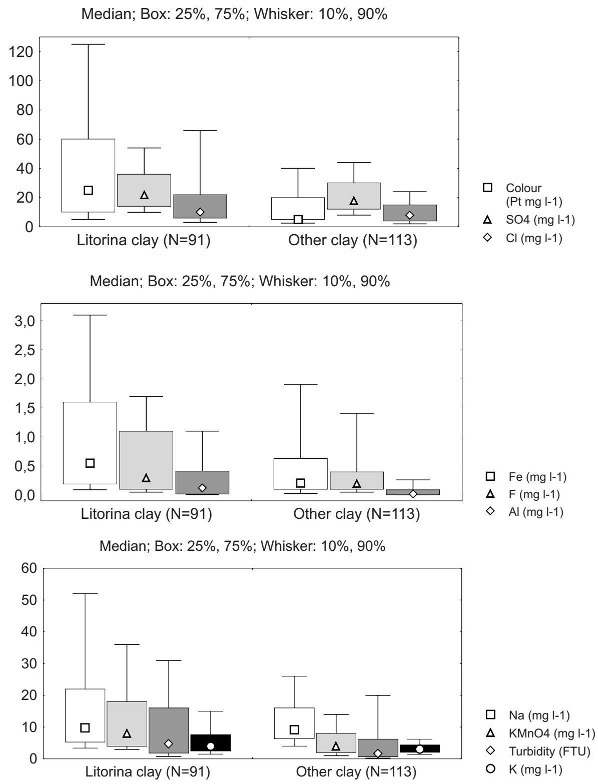


Fig. 23. Selected water quality variables in groundwater from 'Litorina clay' and 'Other clays'.

the amount of rain, the length of rainy periods and the distance from the sea (Rankama and Sahama 1950). The chemical composition of rain water is shown in Table 1. The marine effect will be discussed here as well, as many of the atmospheric elements are of marine origin.

Atmospheric deposition in Finland in the early 1990's, as measured at 15 stations, is described by Järvinen and Vänni (1992, 1993). The electrical conductivity of rain water varies in the range 3.3–1.4 mS m^{-1}, decreasing from the south coast towards northern Finland (Järvinen and Vänni 1992). These results are used here to describe the average composition of annual deposition. A regional-scale grid model for sulphur and nitrogen compounds has also been developed (Hongisto 1998), and these maps for the year 1990 will be compared with the well water quality results.

It has been observed earlier by Backman *et al.* (1999) that airborne sulphur and nitrogen deposition causes acidification of groundwater in pristine areas of Finland. The sulphate content of atmospheric deposition is in the range 390–2700 mg m^{-2}, the highest loads being encountered regionally in southern Finland, caused by the industries of the St. Petersburg district. There are also emissions from the metallurgical industries on the Kola Peninsula that affect atmospheric deposition in northeastern Lapland (Järvinen and Vänni 1992 and 1993, Backman *et al.* 1999, Mälkönen 1998, Lahermo *et al.* 1994). The regional distribution pattern of SO_4 concentrations of well water (Fig. 24) resembles this distribution of sulphuric fallout. According to Backman *et al.* (1999), the SO_4 content of groundwater has increased especially markedly in southern Finland, confirming the impact of atmospheric deposition on certain areas of Finland. But there are other sulphate sources as well as airborne sulphur, and it is these that generally seem to predominate. Lahermo *et al.* (1994) report that stream water sulphate concentrations in Finland are controlled by sulphur deposition, primary

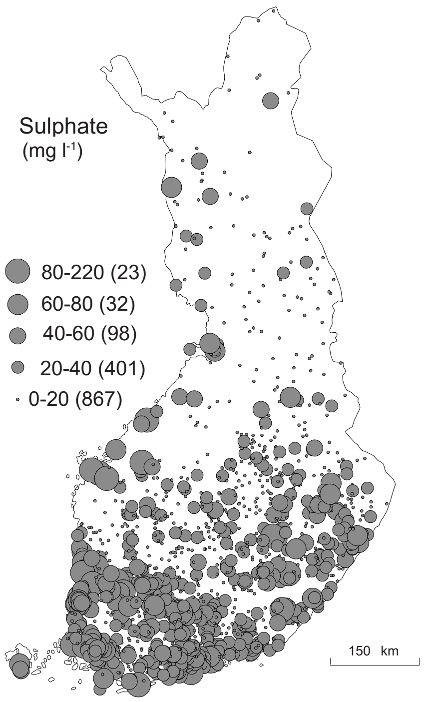
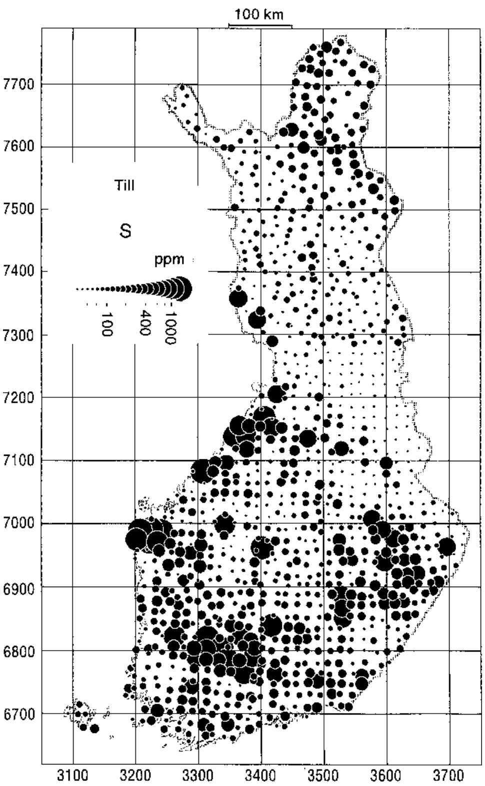
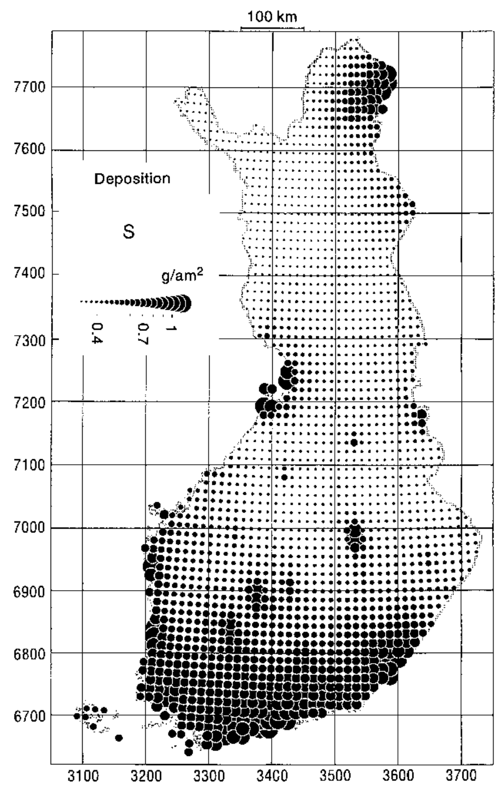


Fig. 24. Sulphate concentrations in well water compared with those in the fine fraction (< 64 um) of glacial till and modelled anthropogenic sulphur deposition (grams per square metre per year) in Finland in summer 1990 (Lahermo *et al.* 1994).





sulphide minerals in the bedrock and till and sulphides/sulphates in postglacial clay and silt deposits dating back to past marine phases of Baltic Sea. The same seems to hold true for well water (Fig. 24), sulphate concentrations in which are controlled also by sulphur in the fine fraction of glacial till (Fig. 24). The effects of other sulphate sources will be discussed in later chapters.

Atmospheric nitrate deposition is approximately 2/3 of that of sulphate, while its areal distribution is very similar. The distribution patterns of nitrogen compounds in well water do not follow the calculated fallout particularly closely, however, indicating the dominant effect of other factors. This will be discussed later.

Atmospheric deposition of chloride and sodium, mainly of marine origin, is highest in southern Finland and along the coast, decreasing towards the north and increasing again in the northernmost parts of Finland because of the marine effect caused by the Arctic Ocean (Järvinen and Vänni 1992). The majority of the wells studied here (62 %) were located more than 50 km from the sea (Fig. 25), while only 1.5 % were situated 10 km or less from the sea, 71 of them immediate adjacent to it (≤ 1 km).

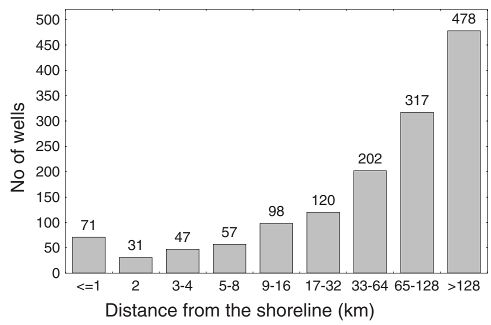


Fig. 25. Distances of the studied wells from the seashore.

Location closer than 10 km to the sea is reflected in elevated SO_4, Cl and Na concentrations in the well water and high electrical conductivity (Fig. 26). This was seen in all types of well, the concentrations decreasing as the distance from the seashore increased.

Concentrations of chloride are generally higher along the coast of Finland, and the spatial distribution of chloride in well water (Fig. 27) indicates that it can be assumed to be predominantly of ma-

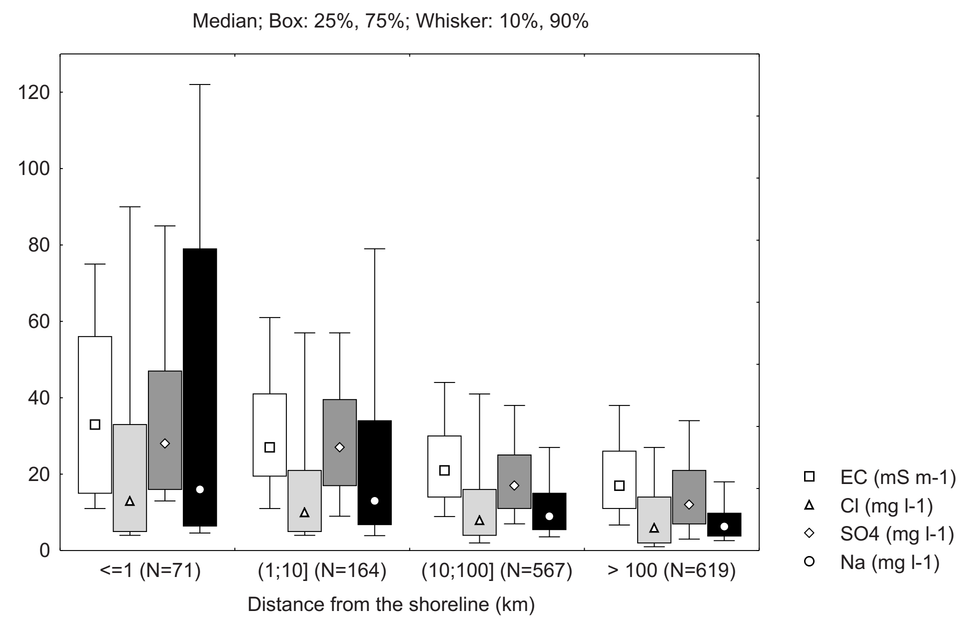


Fig. 26. Relations of electrical conductivity, chloride, sulphate and sodium concentrations of well water to the distance of the well from the seashore.

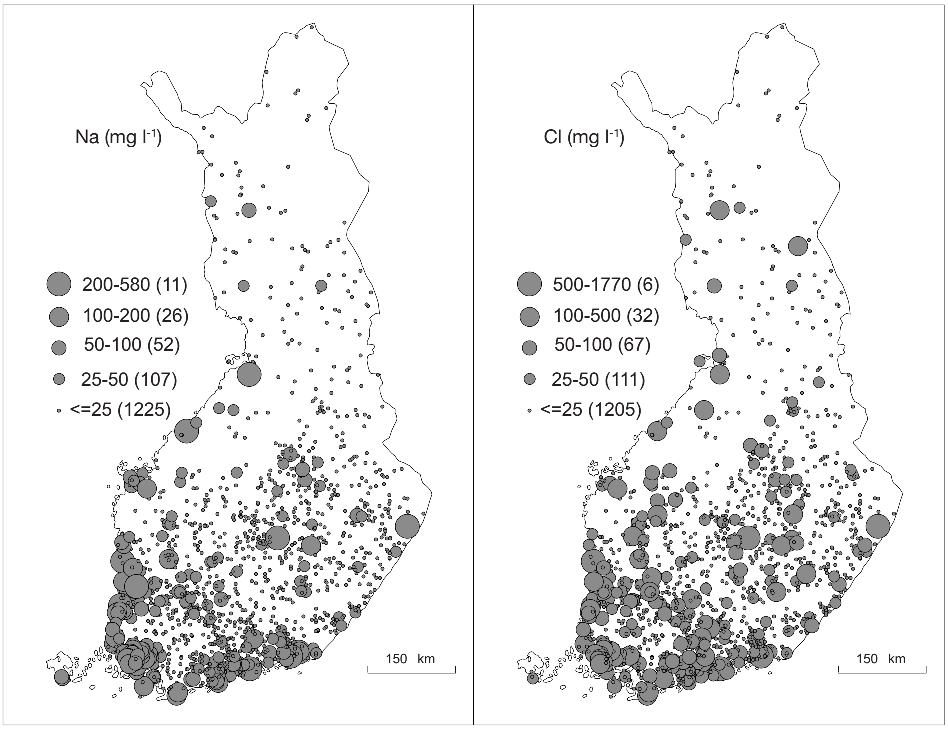


Fig. 27. Spatial distributions of sodium and chloride. Concentrations are generally higher along the coast of Finland.

rine origin, either via salts in precipitation, leaching of relict sea water salts from clays or intrusion of present sea water. Other possible sources may be anthropogenic, e.g. fertilisers and road salts, or lithological (Banks *et al.* 1998b, Lahermo *et al.* 1990). Sodium shows a rather similar distribution pattern to that of chloride.

3.4.5 Long-term changes in water quality

The possibility of long-term changes in the quality of well water was investigated by comparing the present results with the data of Wäre (1960). The analytical methods used in 1958 and 1990 were based on the same principles, but levels of accuracy have improved since Wäre's time. The detection limits were harmonised to the less precise of the two data sets (Table 10), which renders the lowest values comparable. Total coliform bacteria were analysed in 1958 as well, but the samples had been stored for such a long time before analysis that the results were not valid and are not used in this comparison.

The distributions for the years 1958 and 1990 are compared by means of the non-parametric Mann-Whitney's U-test for two unrelated populations (Rock 1988, Ranta *et al.* 1991, Table 11).

Table 10. Harmonised detection limits for certain variables.

Variable	Year 1958	Change	Year 1990
Colour	2	→	5
F	0.05	→	0.1
NH_4	0.05	→	0.1
Fe	0.02	→	0.05
Mn	0.05	←	0.02

Table 11. Scores (*Z*) and significance (*p*) of differences between the well water quality variables in 1958 (N is from 2579 to 2625) and 1990 (N = 1096), based on the Mann-Whitney U-test. Only significant differences ($p < 0.05$) are shown, and highly significant differences ($p < 0.0001$) are indicated in bold type.

Variables	Mann-Whitneyn U-test	
	Z	*p*
EC	**6.08**	**< 0.0001**
pH	**4.35**	**< 0.0001**
Alkalinity	**12.42**	**< 0.0001**
Colour	**8.38**	**< 0.0001**
$KMnO_4$ consumption	**26.87**	**< 0.0001**
SO_4	**4.39**	**< 0.0001**
Cl	**10.03**	**< 0.0001**
NO_2	2.72	0.0066
NH_4	3.24	0.0012
Fe	**4.63**	**< 0.0001**
Mn	2.90	0.0038

As shown in Table 11, there are highly significant differences ($p < 0.0001$) in the values for electrical conductivity, alkalinity, colour and $KMnO_4$ consumption and in the chloride, iron and sulphate concentrations between the late 1950's and early 1990's. These changes are demonstrated in terms of arithmetic means, trimmed means (10 %) and medians for the variables in year 1958 and 1990 in Table 12. Because of outliers and extreme values, median and trimmed mean represent the whole population better than the arithmetic mean, as noted earlier by Tarvainen (1996).

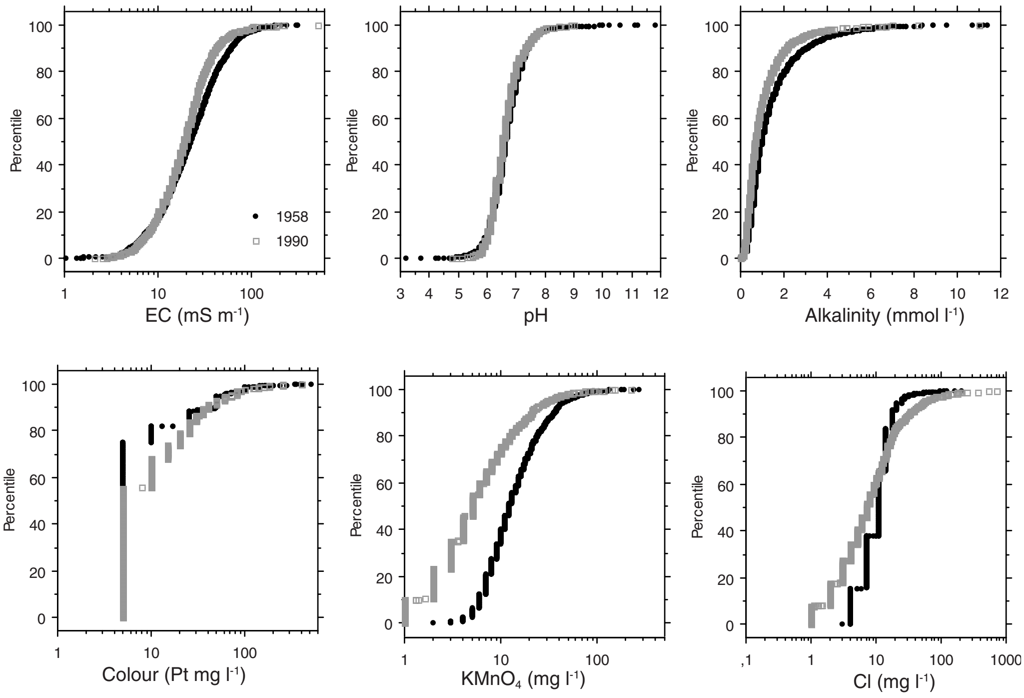
Cumulative frequency distribution curves for the most important variables in year 1958 and 1990 are presented in Fig. 28, where some of the variables are presented on a log scale. As may be seen in Table 12 and Fig. 28, the pH of the well water had decreased slightly from 1958 to 1990, as had conductivity, while there had been significant changes in alkalinity and iron concentrations. $KMnO_4$ consumption had decreased to a moderate extent, as also had conductivity when the median and trimmed mean values were compared. Soveri *et al*. (2001) observed a decrease in chloride content at 55 % of his groundwater stations over the period 1975–1999. The arithmetic mean for chloride had increased, however, mainly as a result of some very salty wells affected by present or relict sea water intrusion or road salts. The iron concentrations were generally lower in the 1990's than in the late 1950's and the colour indices and sulphate concentrations had increased.

A clear decrease in alkalinity and increase in sulphate concentration (Table 12) and electrical conductivity can be an indicator of extra-regional acidification, as also observed at groundwater monitoring stations by Backman *et al.* (1999) and Soveri *et al.* (2001), who maintained that groundwater acidification had ceased by the early 1990's and that alkalinity values had been increasing since then in many places.

The notable decrease in iron and chloride concentrations and in $KMnO_4$ consumption (Table 12) may indicate changes in site-specific factors affecting the wells, such as the depth of the well or the way in which it has been constructed.

Table 12. Arithmetic means, trimmed means (10 %) and medians for the variables in 1958 and 1990. Units are the same as in Table 3. The number of wells is 2615–2625 for year 1958 and 1096 for year 1990.

Variable	Mean 1958	Mean 1990	Trim.mean 1958	Trim.mean 1990	Median 1958	Median 1990	Median 1958–median 1990
EC	29.7	23.7	25.0	20.1	22	19	3
pH	6.68	6.60	6.65	6.56	6.60	6.50	0.10
Alkalinity	1.41	1.00	1.16	0.81	1.00	0.66	0.34
Colour	15.9	18.2	7.7	10.9	5	5	0
$KMnO_4$ consumption	17.9	9.4	14.4	6.5	12	5	7
SO_4	19.6	20.4	16.2	17.4	13.0	15.5	-2.5
Cl	12.0	17.3	10.6	9.2	11	7	4
F	0.26	0.31	0.15	0.16	0.1	0.1	0
NO_3	14.22	12.57	8.82	8.47	5.0	5.2	-0.2
NO_2	0.04	0.03	0.01	0.01	0.01	0.01	0
NH_4	0.29	0.22	0.1	0.1	0.1	0.1	0
Fe	0.98	0.69	0.50	0.30	0.30	0.17	0.13
Mn	0.08	0.12	0.05	0.06	0.05	0.05	0



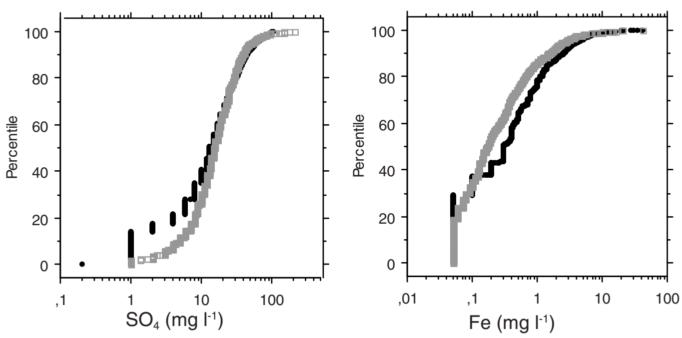


Fig. 28. Cumulative frequency distribution curves for electrical conductivity, pH, alkalinity, colour, $KMnO_4$ consumption, chloride, sulphate and iron. A log-scale is used on the *y*-axis for most of the variables. All data below the detection limit are plotted here at the value represented by the detection limit, because results falling below the limits were not separated from those at the limits in the 1958 data of Wäre (1961a, 1961b).

The changes are not as substantial on a national scale as had been assumed in the preliminary investigation into well water acidification (Korkka-Niemi 1990), although they differed in magnitude between areas (Korkka-Niemi 1990) and were clearer in wells dug into small sand deposits and till sediments than elsewhere. The spatial variation in the changes can also be seen in these data (Korkka-Niemi *et al.* 1993).

3.5 Regional effects

Water quality effects arise from factors related to the residence time of the water in the aquifer and the time available for geochemical reactions with solid organic and mineral matter, and on the proximity of the aquifer system to the surface. They are therefore largely dependent on the type and nature of the aquifer. Aquifers in Finland are much less broad in extent than the geological factors discussed above, however, and can be considered to be regional or local in scale.

As suggested by Soveri (1985), the permeability of the soil may have the greatest effect on concentration differences between groundwater from different soil-type regions. There is a very limited amount of information available on the detailed hydraulic properties of Finnish aquifers. Calculations of hydraulic conductivity using particle size analysis, infiltration tests and slug-tests (Salonen *et al.* 2001) have yielded figures of $9 \cdot 10^{-5}$... $3 \cdot 10^{-2}$ m s^{-1} (median $8 \cdot 10^{-4}$) for a small esker formation, $4 \cdot 10^{-5}$ ...$1 \cdot 10^{-3}$ m s^{-1} (median $6 \cdot 10^{-4}$) for beach deposits, $1 \cdot 10^{-7}$ ... $6 \cdot 10^{-5}$ m s^{-1} (median $4 \cdot 10^{-5}$) for silty sediments and $3 \cdot 10^{-6}$ ... $1 \cdot 10^{-4}$ m s^{-1} (median $4.5 \cdot 10^{-5}$) for till deposits. Fairly similar values ($8 \cdot 10^{-5}$....$1 \cdot 10^{-3}$ m s^{-1}) are reported by Mäkelä and Peltokangas (1991) for esker aquifers.

The test pumpings performed in southern Finland and analysed by Pönkkä (1981) show that the average yield of ice-marginal end moraines is 847 m^3 d^{-1}, that of longitudinal eskers 664 m^3 d^{-1}, that of lee-side accumulations 285 m^3 d^{-1} and that of clay-covered aquifers 280 m^3 d^{-1}, although the normal yield for a typical small Quaternary aquifer hosting a private well is probably even smaller. The yield of a Quaternary aquifer is usually much higher than that of a bedrock aquifer, the average yield of wells drilled into bedrock being 30–60 m^3 d^{-1} (Laakso 1966, Rönkä 1983, Mäkelä 1990).

The role of the grain size of the aquifer material and the residence time in controlling the water chemistry has been discussed earlier in Finland by Pönkkä (1981) and Lahermo (1970), for instance. Pönkkä (1981) concluded that the main factor affecting water chemistry in glaciofluvial aquifers in the southern part of Finland is residence time. The effects of hydraulic factors associated with the aquifer, described in terms of the depositional environment and surficial sediment, will be discussed below, and since seasonal variation in groundwater quality may be regarded as a regional-scale factor, seasonal variations in water quality will also be discussed in this connection.

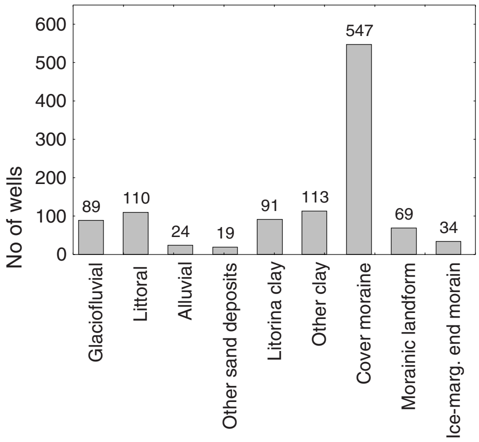


Fig. 29. Dug wells grouped by the depositional environment of the aquifer.

3.5.1 Aquifer type

As shown in Fig. 29, half of the wells in Quaternary deposits studied here were dug into cover moraine. Altogether 69 wells were located in drumlins or other morainic deposits, and 34 in ice-marginal end moraines such as that of Salpausselkä, 91 in Litorina clay layers, 110 in other clay areas, i.e. clays deposited during the Baltic Ice Lake, Yoldia Sea or Ancylus Lake phases of Baltic Sea, 89 in glaciofluvial gravel or sand deposits and 110 in littoral deposits. Only a few wells (24) had been dug into fluvial or other sand deposits.

The chemical composition of groundwater can vary greatly depending on the genetic type of the Quaternary deposits (Fig. 30). Average electrolyte concentrations were lowest in glaciofluvial sand/gravel deposits and highest in the wells dug through Litorina clay, generally increasing from coarse-grained deposits to fine sediments in the order: other sand and gravel deposits < glaciofluvial deposits < littoral deposits < ice-marginal end moraine complex < cover moraine < morainic landform < fluvial deposits < other clay < Litorina clay. Electrical conductivity, hardness, manganese and sodium concentrations were lowest in the glaciofluvial aquifers and other sand aquifers. Wells in other sand and gravel deposits had a low electrolyte content, because these are usually small lee-side deposits where the residence time of the water is rather short. The specific surface of the mineral grains increases with decreasing grain size, whereupon water-rock interaction becomes more intensive. Thus a higher proportion of the fine-grained fraction in till, silt or clay material will be reflected in a higher electrolyte content of the groundwater than in glaciofluvial aquifers.

Natukka (1960, 1963) and Lahermo (1970) have both earlier detected that the groundwater occurring in eskers and other glaciofluvial deposits is poorer in dissolved ionic matter than that residing in till deposits. Lahermo (1970) found in south-eastern Finland that the average electrolyte concentration of groundwater was lowest in small littoral and till deposits, slightly higher in esker deposits and by far the highest in deposits covered by clay, while Pönkkä (1981) subsequently postulated that the high electrolyte content, pH and hardness of the water in clay-covered aquifers must be caused by the long retention time of the water and the large amounts of free carbon dioxide in it. The fact that nitrate concentrations do not follow the other variables (Fig. 30) will be discussed further in next section.

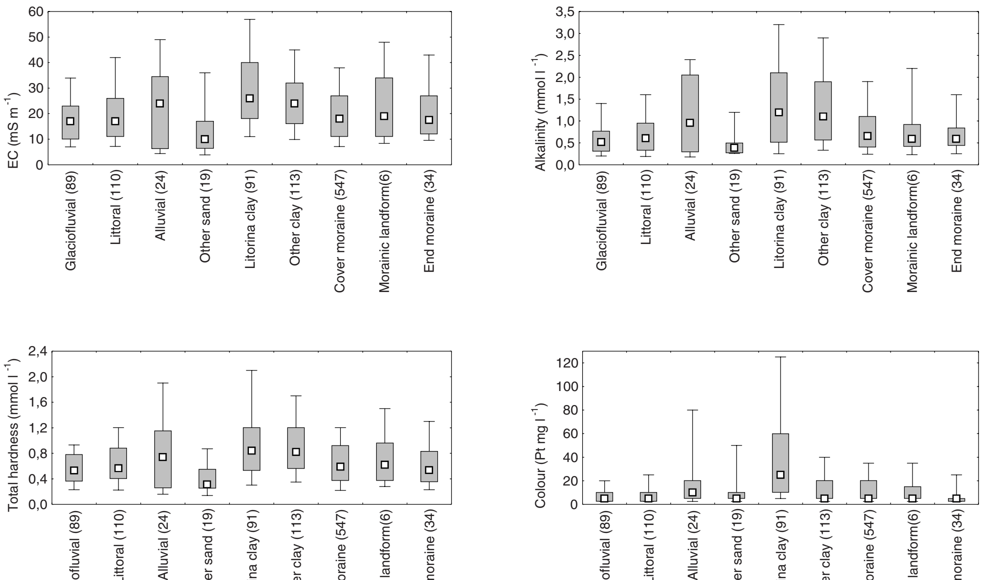


Fig. 30 cont.

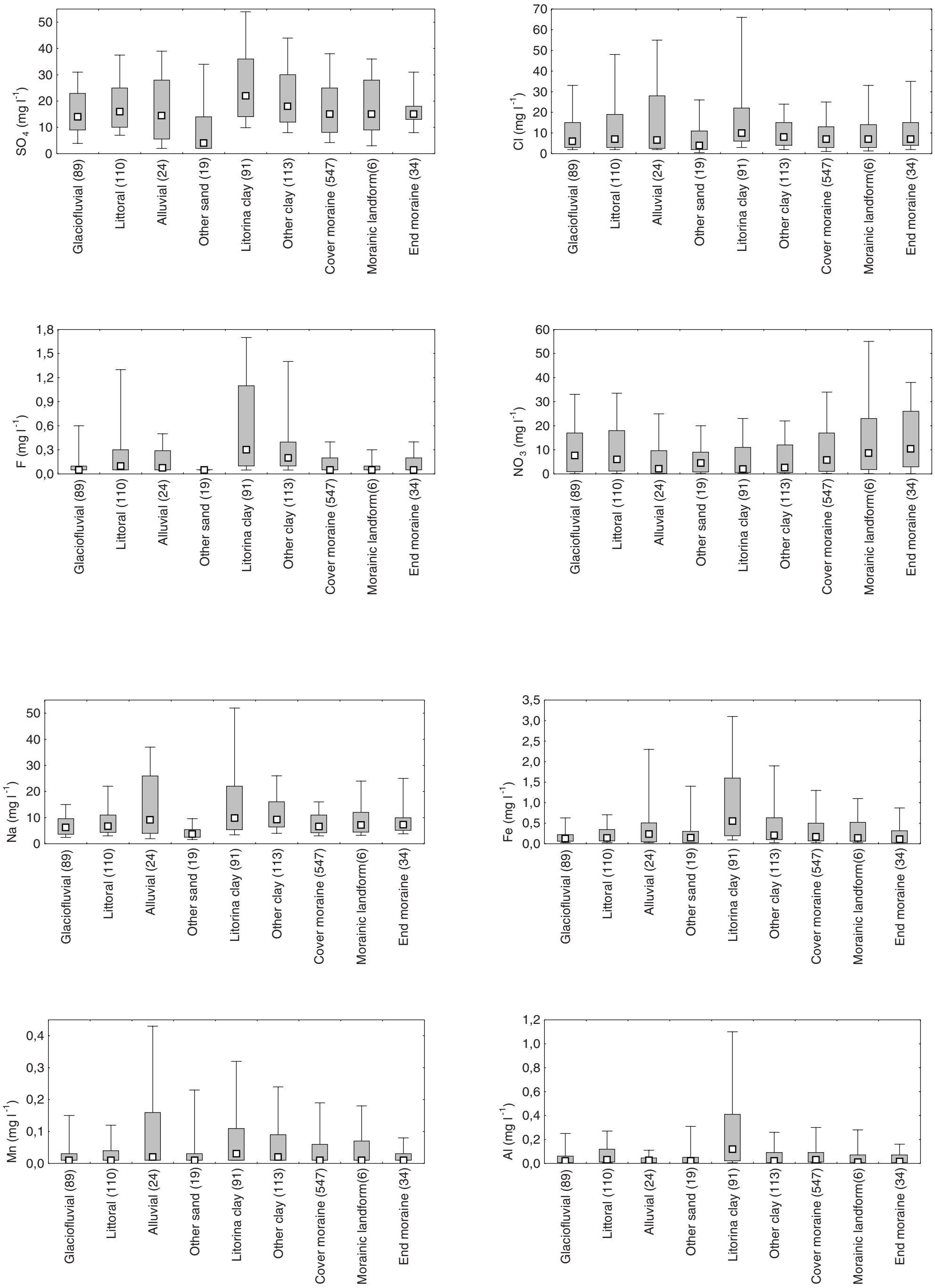


Fig. 30. Well water quality by depositional environment of the aquifers. (Median; Box: 25 %, 75 %; Whiskers: 10 %, 90 %)

3.5.2 Surficial sediment

In his examination of the effects of retention time on the quality of groundwater, Soveri (1985) concludes that the finer the soil is in terms of grain size, the longer will be the residence time and the more time there will be for soil processes to alter the quality of the infiltrating water and ultimately that of the groundwater. This was clearly seen above when the water quality results were grouped by depositional environment (Fig. 30). It will be concerned in this section with the importance of the type of surficial sediment as an indicator of residence time.

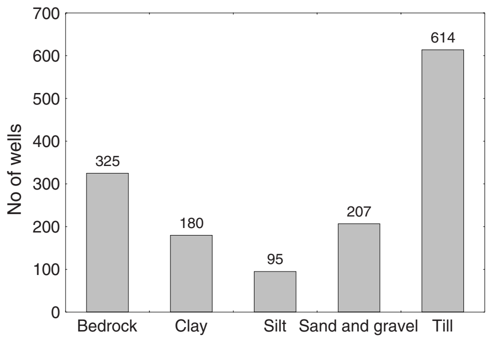


Fig. 31. Wells classified by type of surficial sediment, as deduced from geological maps. Illustration includes also bedrock wells (bedrock).

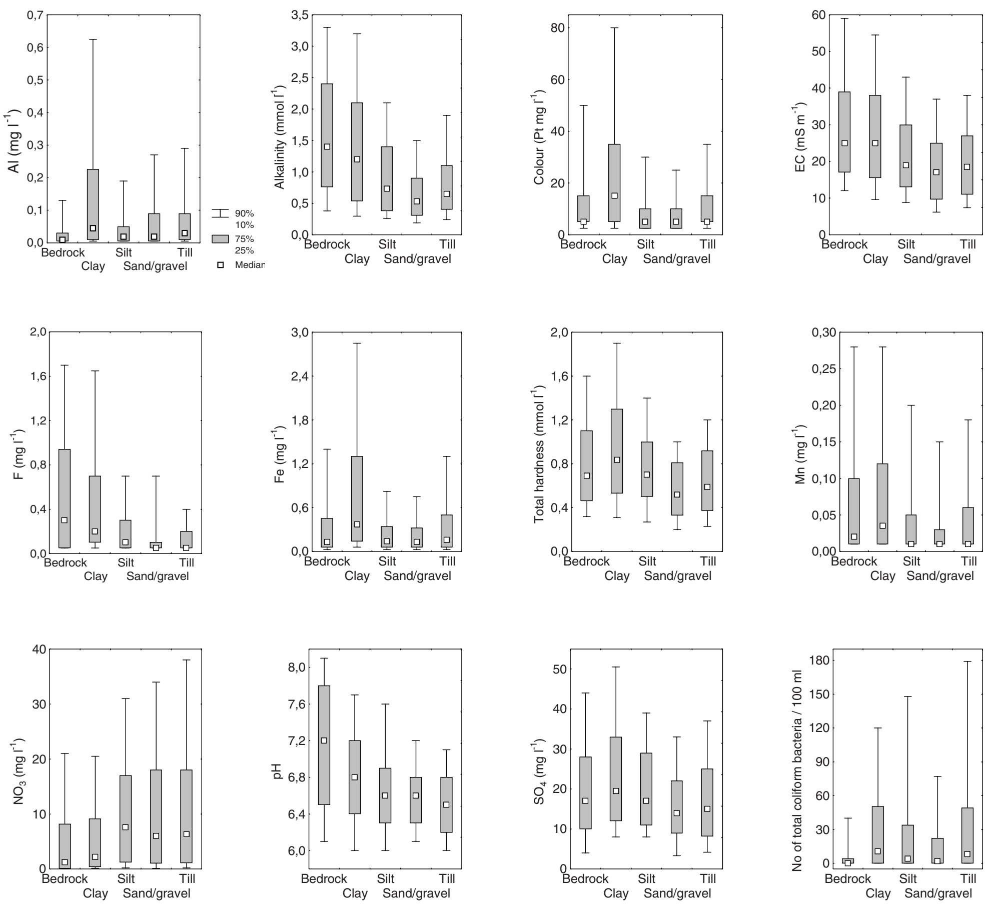


Fig. 32. Box plots of selected water quality variables grouped by the surficial sediment near the well. Illustrations include also bedrock wells (bedrock).

The most common surficial sediment around the wells, as deduced from geological maps, was till (Fig. 31), although there were also many wells (207) situated on sandy material, 95 that had been dug into silty material and 180 where the surface was covered with clay. Wells constructed in clay-covered aquifers were especially common in the south-western parts of the country and to a lesser extent in the south-east. Wells in till-covered surroundings were evenly distributed throughout the country.

The pH of the well water was highest in the clay-covered areas and lowest in the till areas (Fig. 32), the buffering capacity of the groundwater being higher the smaller the grain size of the sediment. This can be seen from Fig. 32, where alkalinity increases from sand/gravel to till, silt and eventually clay. Electrical conductivity, total hardness, colour and sulphate and fluoride concentrations behave similarly. Iron and aluminium concentrations are also highest in clay-covered aquifers, where nitrate concentrations are lowest. There are more total coliform bacteria in water from wells dug into till and clay-covered areas than in areas of silt, sand or gravel deposits.

The categorisation of water quality according to type of surficial sediment can also be related to the quality requirements and targets for domestic water supplies, as in Tables 13 and 14, which show the percentages of the wells affected by certain quality problems grouped by the type of surficial sediment.

Fluoride is a major health risk in bedrock wells and wells dug into clay-covered aquifers, whereas nitrate is not a risk in these wells, but wells dug into coarse sediments often tend to have excessively high nitrate concentrations. The presence of bacteria (TC and FC) is more often a problem in wells dug into clay, silt or till than elsewhere. Wells dug into clay-covered deposits are clearly the most problematic as far as the technical-aesthetic quality objectives are concerned (Table 14).

Table 13. Percentages of wells having health-related quality defects classified by the surficial sediment in their surroundings.

	Surficial sediment (number of wells)				
Defect	Bedrock(325)	Clay(180)	Silt(95)	Sand/gravel(207)	Till(614)
TC ≥100/100 ml	6.2	13.3	12.6	7.7	15.5
FC ≥ 1/100 ml	7.7	18.9	14.7	6.8	12.5
F > 1.5 mg l^{-1}	13.5	11.7	4.2	4.4	3.3
NO_3 > 25 mg l^{-1}	7.7	7.2	12.6	16.9	16.6
NO_2 > 0.1 mg l^{-1}	2.2	2.8	1.1	1.0	2.8

Table 14. Percentages of wells having technical-aesthetic quality defects classified by the surficial sediment in their surroundings.

	Surficial sediment (number of wells)				
Defect	Bedrock(325)	Clay(180)	Silt(95)	Sand/gravel(207)	Till(614)
pH	5.9	6.7	9.5	6.3	8.3
Colour	21.5	46.1	16.8	16.9	23.8
Turbidity	16.0	40.0	12.6	10.6	17.0
$KMnO_4$	7.4	13.3	4.2	7.7	11.3
Cl mg l^{-1}	4.0	5.0	3.2	1.5	1.6
NH_4 mg l^{-1}	0.9	2.2	1.1	1.9	2.8
Na mg l^{-1}	2.5	0	2.1	0.0	0.7
Fe mg l^{-1}	21.5	42.2	16.8	4.5	24.9
Mn mg l^{-1}	13.9	14.4	9.5	9.2	8.5
Al mg l^{-1}	8.3	26.1	8.4	13.5	13.7

Water quality in wells dug into sand/gravel deposits is normally good, although aluminium concentrations are too high in 13.5 % of cases.

A clay cover plays an important role with regard to the groundwater quality of an aquifer, as nitrate concentrations were lowest in the clay-covered aquifers and highest in those located in morainic deposits, ice-marginal end moraines or glaciofluvial deposits. The state of oxidation of nitrogen compounds and the occurrence of iron and manganese in groundwater are principally governed by the oxygen balance, which depends on the geological structure and flow patterns of the aquifer (Champ *et al.* 1979, Domenico and Schwartz 1998). The groundwater in clay-covered aquifers and in bedrock fissures and fractures has often been found to be deficient in oxygen, mainly due to its long retention time and various reactions in which the decomposition of organic matter and the oxidation of iron, manganese and nitrogen compounds consume the oxygen and the groundwater becomes reducing with respect to these compounds (Lahermo 1970, Hatva 1989, Hem 1992, Backman *et al.* 1999, Karro 1999).

In addition to the state of oxidation, the geochemical behaviour and speciation of iron, manganese, nitrogen and sulphur compounds will also be affected by microbial processes (Edmunds 1986). The total number of microbes in soil and groundwater depends on factors such as the amounts of nutrients, pH, temperature and the redox state, the figures recorded in groundwater being in the range 100–1000 microbes/ml (Hem 1992, Heinonen-Tanski 1987, Seppänen 1987). Faecal microbes, which live in an anaerobic environment, also thrive in aquifers having a low oxygen content (Heinonen-Tanski *et al.* 1998).

The concentrations of the various nitrogen compounds depend on the degree of nitrification and denitrification. The NO_3 /NH_4 ratios depicted in Fig. 33 indicate the lower redox potentials observed in clay-covered aquifers and bedrock aquifers relative to other types of shallow aquifers. The lack of nitrate in these two groups suggests that some microbiological reduction of nitrate to nitrite and ammonium has occurred, as suggested earlier by Seppänen (1987) and Lahermo and Kutvonen (1990). It can also be so, that the environment may be unsuitable for nitrification due to the lack of oxygen and bacteria, so that nitrogen will remain in the form of ammonium or nitrite.

There was more soluble iron in the water from the clay-covered aquifers than elsewhere (Fig. 30), and the situation was the same with manganese, although not so clearly so, because, as Hem (1992) has concluded, Mn^{2+} ions are much more stable with regard to oxidation than those of ferrous iron, so that manganese may remain in solution even in water that has a moderate oxygen content. Hatva (1989) detected over ten times higher median values for iron and three times higher manganese concentrations in clay-covered synclinal aquifers than in anticline oxygenated unconfined aquifers.

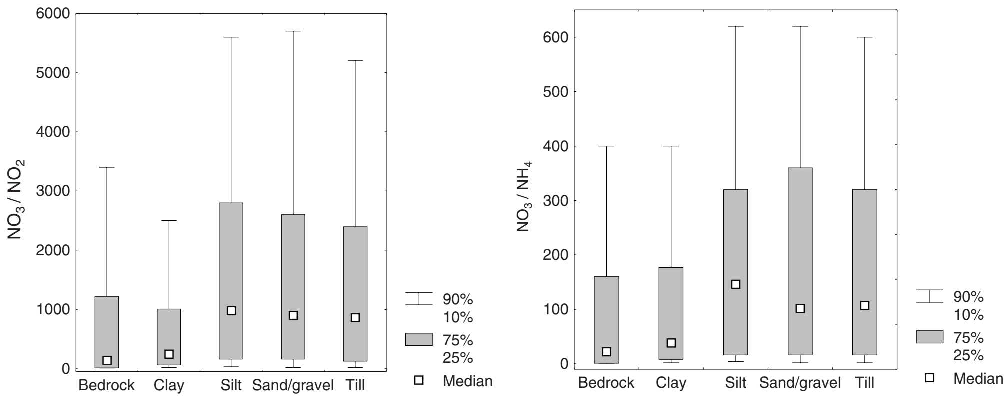


Fig. 33. NO_3/NO_2 and NO_2/NH_4 ratios in well water from different types of aquifer.

3.5.3 Seasonal variation in well water quality

Although the quality of the groundwater in large Quaternary aquifers and bedrock aquifers tends to remain more or less constant, a clear seasonality has been observed in small, shallow aquifers (Lahermo 1970, Mälkki 1977, Soveri 1985, Backman 1999 *et al.,* Soveri *et al.* 2001). Oxygen-rich gravitational water from melting snow infiltrating into shallow groundwater zone has a powerful effect on water quality (Mälkki 1977), leading Soveri *et al.* (2001) to conclude that meltwater in spring (minimum pH and alkalinity) and the nutrient intake of herbivores in summer (minimum nitrate content) are the most important factors controlling seasonal variations in groundwater quality as observed in Finland.

In the present survey, possible seasonal changes in well water quality were ascertained by examining 405 wells in three seasons: autumn 1990, spring 1991 and summer 1991 (Table 15). The weather conditions at and before the time of sampling affect the quality of well water. Autumn 1990 was a rainy season in some places and dry in others, while winter 1990-1991 was almost snowless in southern Finland, so that there was little snow to melt in the spring. It appeared to be difficult to describe the general weather conditions. Samples were taken in the northern parts of the country at the beginning of the sampling period and somewhat later in the south in order to compensate differences in the weather.

The non-parametric Wilcoxon Signed Ranks Test for related populations (Ranta *et al.* 1991, Rock 1988) was used to confirm possible seasonal variation in water quality, and the whole set of water data for the 405 wells was assessed using the SPSS and STATISTICA software (Table 16). There were statistically significant differences ($p < 0.001$) between the seasons in electrical conductivity, alkalinity, hardness, $KMnO_4$-consumption, F, K, Fe, Mn, total coliform bacteria and faecal coliform bacteria (Table 16). The electrical conductivity of the water was highest in autumn and lowest in spring, having 2.1 mS m^{-1} differences in its mean values (Table 15), while alkalinity was generally 0.11 mmol l^{-1} lower in spring than in summer, when it was at its highest. Total hardness was also lowest in spring, on account of

Table 15. Arithmetic means, trimmed means (10 %) (T) and medians of water quality variables for 405 wells in autumn 1990, spring 1991 and summer 1991.

Variable	Mean Autumn 1990	Mean Spring 1991	T.mean. Summer 1991	T.mean Autumn 1990	T.mean Spring 1991	Median Summer 1991	Median Autumn 1990	Median Spring 1991	Summer 1991
TC	4.3 *	2.5 *	3.6 *	14.69	7.06	11.02	2	0	3
FC	0.64 *	0.53 *	1.06 *	0.04	0.00	0.70	0	0	0
EC	27.7	25.7	26.9	22.7	21.1	22.4	20.0	19.0	20.0
pH	6.73	6.69	6.67	6.68	6.63	6.61	6.60	6.60	6.50
Alkalinity	1.28	1.22	1.34	1.05	1.00	1.13	0.82	0.74	0.92
Hardness	0.76	0.73	0.78	0.67	0.64	0.67	0.60	0.59	0.63
Colour	21.4	17.6	18.8	12.1	10.4	10.4	5.0	5.0	5.0
Turbidity	4.48	3.35	3.50	1.85	1.58	1.61	0.85	0.90	0.85
$KMnO_4$	9.8	10.4	10.0	7.0	7.7	6.9	5.7	6.5	5.0
SO_4	20.4	20.7	21.8	16.6	17.1	17.3	14.0	14.0	15.0
Cl	20.4	19.3	21.2	10.2	9.9	10.4	7.0	7.0	7.0
F	0.65	0.57	0.68	0.44	0.38	0.45	0.11	0.10	0.10
NO_3	10.52	10.03	10.45	6.34	6.11	5.82	3.40	3.00	2.60
NO_2	0.04	0.02	0.04	0.01	0.01	0.01	0.01	0.01	0.01
NH_4	0.20	0.18	0.16	0.10	0.10	0.10	0.10	0.10	0.10
Na	20.35	19.40	21.81	10.82	10.59	11.63	7.30	7.10	7.40
K	6.18	5.80	5.86	3.69	3.62	3.67	3.10	3.00	2.90
Fe	0.64	0.57	0.58	0.32	0.23	0.26	0.16	0.12	0.14
Mn	0.10	0.10	0.11	0.05	0.04	0.05	0.02	0.02	0.02
Al	0.17	0.16	0.14	0.07	0.08	0.06	0.03	0.04	0.03

* Geometric means are calculated for microbiological variables instead of arithmetic means.

Table 16. Wilcoxon scores (*Z*) and significance levels (*p*) for the 405 wells analysed three times within a year: autumn 1990, spring 1991 and summer 1991. Only significant differences ($p < 0.05$) are shown, and highly significant differences ($p < 0.001$) are indicated in bold type.

	Wilcoxon signed Rank Test					
	summer-spring		spring-autumn		autumn-summer	
Variable	*Z*	*p*	*Z*	*p*	*Z*	*p*
TC	2.87	0.012			**3.96**	**< 0.0003**
FC	**8.12**	**< 0.0003**	**6.49**	**< 0.0003**	**4.97**	**< 0.0003**
EC	**4.59**	**< 0.0003**	**6.65**	**< 0.0003**	3.23	0.003
pH	2.54	0.033	3.28	0.003		
Alkalinity	**7.78**	**< 0.0003**			**4.89**	**< 0.0003**
Total hardness	**4.97**	**< 0.0003**	3.17	0.006		
Colour						
$KMnO_4$	4.39	< 0.0003				
SO_4					3.23	0.003
F	**6.17**	**< 0.0003**	**6.80**	**< 0.0003**		
NO_3					**2.34**	**0.057**
NO_2	2.75	0.018	2.51	0.036		
Na	2.52	0.036	2.81	0.015		
K	2.93	0.009	**3.79**	**< 0.0003**		
Fe			**4.82**	**< 0.0003**	2.25	0.075
Mn	3.59	< 0.0003	2.71	0.021		
Al	3.39	0.003				

the diluting effect of the large volume of water infiltrating from melting snow, ground frost and rainfall.

The greatest difference in $KMnO_4$ consumption was between the summer and spring (Table 15), the highest values generally being recorded in spring. According to Backman *et al.* (1999), seasonal variation in groundwater quality is reflected in $KMnO_4$ consumption and colour, high values being obtained in spring and autumn by virtue of the infiltration of meltwater and autumn rain into the groundwater. Lahermo (1970) reported similar results for surface water, but not for groundwater.

The fluoride content was generally highest in summer, which is in line with the observations of Karro (1999), who suggested that this is due to the low pH of the precipitation and meltwater contributing to the effectiveness of fluoride leaching into the groundwater from the unsaturated zone.

The iron content of well water was somewhat higher in autumn than at other seasons (Table 15). According to Lahermo (1970), these seasonal variations may partly be due to the changes in the groundwater table. As the level of the groundwater rises, iron compounds that have been precipitated into the soil are likely to redissolve under reducing conditions and in response to possible leaching of dissolved organic matter into the water.

Total coliform bacteria and faecal coliform bacteria were low in the spring samples maybe because of the lack of fresh waste water contamination. Total coliforms then increased in summer and reached their maximum in autumn, when the surface run-off generated by the rains enhanced the infiltration rate. The numbers of faecal coliforms were clearly highest in the summer samples, possibly because of cattle grazing outside in the fields (see also Korkka-Niemi *et al.* 1993).

Since most of the variables seem to be related to access of surface runoff to the well, those having a visible surface water effect were excluded from the data set and the Wilcoxon scores recalculated, as there was no intention here to examine water quality problems related to faulty well construction. Bedrock wells were also excluded from the next run (Table 17), because bedrock aquifers vary so much in size and depth that no common seasonal variations in water quality could be found (see Lahermo 1970). The retention time in some bedrock wells may be several years, while in others it can be very short because of direct connections to the surface via fractures and fissures.

Highly significant differences ($p < 0.001$) in electrical conductivity, alkalinity, hardness, $KMnO_4$ consumption, F, total coliform bacteria and faecal coliform bacteria were obtained for the

Table 17. Wilcoxon scores (*Z*) and significance levels (*p*) for the 209 dug wells and springs where no surface water effect was visible. Only significant differences ($p < 0.05$) are shown, and highly significant differences ($p < 0.001$) are indicated in bold type.

	Wilcoxon signed Rank Test					
	summer-spring		spring-autumn		autumn-summer	
Variable	*Z*	*p*	*Z*	*p*	*Z*	*p*
TC	**3.65**	**< 0.0003**				
FC	**6.26**	**< 0.0003**	**4.96**	**< 0.0003**	**3.66**	**< 0.0003**
EC	3.16	0.006	**3.72**	**< 0.0003**		
pH	2.99	0.009				
Alkalinity	**4.43**	**< 0.0003**			**2.40**	**0.048**
Hardness	**3.78**	**< 0.0003**				
$KMnO_4$	**4.20**	**< 0.0003**				
F	**3.18**	**< 0.0003**	3.08	0.006		
Na					2.44	0.045
Fe			3.23	0.003		
Al	2.96	0.006				

209 dug wells and springs where no surface water effect was visible (Table 18). The differences were smaller (*Z* values) than in the whole set of data, however, which means that the seasonal variation in well water quality is partly explained by the surface water effect increasing in spring. The fact that Backman *et al.* (1999) were unable to find any seasonal fluctuations in the isotopic composition of shallow groundwater at 50 sites indicates that the retention time exceeded the annual water circulation. The same may also hold true in most of the present domestic wells.

It can be concluded that water quality in individual Finnish domestic wells can vary greatly in the course of a year, but there is no systematic variability in water quality depending on the season of sampling except in the numbers of bacteria and in some indicators of surface water infiltration. Seasonal variations are larger in shallow wells (< 5 m) than in deeper ones, and the quality actually depends on interacting meteorological (rain, meltwater), hydrological (retention) and site-specific variables (well construction technology, depth of the well). The natural variation in groundwater quality is masked by other factors, especially by the surface water factor. According to the questionnaire, 27.4 % of the well owners had detected seasonal variation in water quality and 17 % were of the opinion that the quality was poorest in spring. The quality variables characterising individual wells may nevertheless vary more than can be explained by seasonal variation in the aquifers (Soveri *et al.* 2001), suggesting that a sample from a well does not always represent only the water quality in the aquifer but may also reflect on the quality of the well itself.

3.6 Site-specific effects

3.6.1 Type of well construction

Most of the wells sampled in the present survey (80 %) were dug wells or springs, the rest being wells drilled into the bedrock (Fig. 34, 35 and 36). None of the households used surface water.

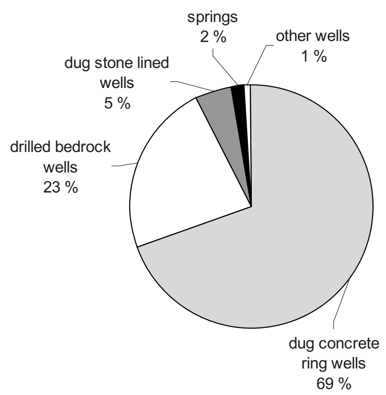


Fig. 34. Types of the wells included in the survey (N = 1421).

(A)

(B)

Fig. 35 cont.

(C)

(D)

Fig. 35. Types of wells sampled in 1990: A) dug well, B) bedrock well C) spring D) stone lined well.

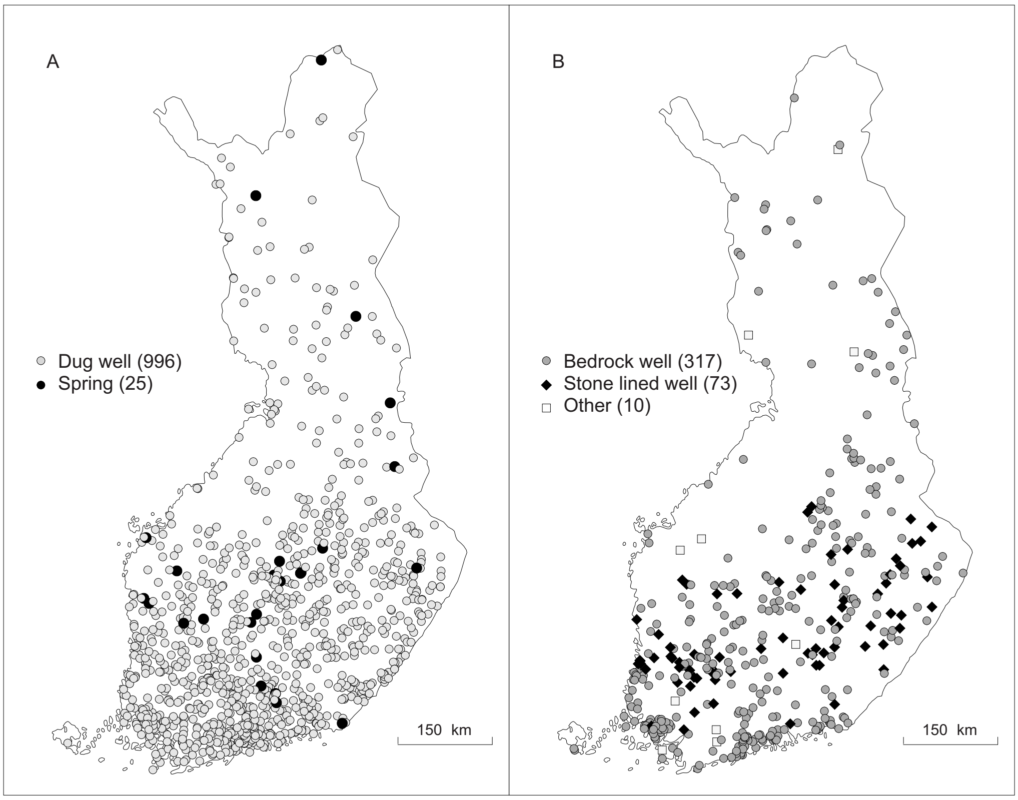


Fig. 36. Spatial distribution of the types of well sampled in 1990: A) dug wells and springs and B) bedrock wells, stone lined wells and other type of wells.

Dug wells lined with concrete rings were the most common type all over the country (Fig. 36). Bedrock wells were most common in the southern coastal areas, but they were also to be found in Central and Northern Finland. Wells of the old type, lined with stones, are still in use in the southwest and in the east. Springs could rarely be found in the inner parts of the country.

The average water consumption of the households was estimated to be 0.5 m^3 per day. Further treatment of the water was extremely rare, and there were just 14 households (1 %) in which it was alkalised or iron and/or manganese was removed from it.

The largest differences in water quality were observed between the dug wells and the bedrock wells, as discussed in section 3.4.1. Using the same data, Korkka-Niemi *et al.* (1993) showed that the water in the wells lined with stones was of the poorest quality microbiologically, had the lowest pH and often had a high nitrate content. This may indicate that the stone rings are seldom tight, and that contaminated, acid, humus-rich water can enter the supply. The water quality in the springs resembled rainwater in having low concentrations of dissolved solids, but in many cases it was of poor microbiological quality, again because of the defective well casing.

3.6.2 Location of the well

Two-thirds of the wells were situated in the yard of a house, the rest being on arable land or in forest (Fig. 37 and 38). The water in the wells located in yards generally had higher sodium concentration and also slightly higher iron, manganese and potassium concentrations than that of the wells locat-

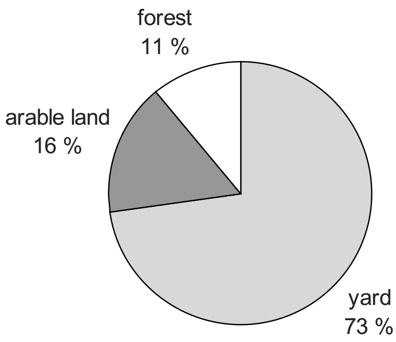


Fig. 37. Locations of the wells in relation to land use.

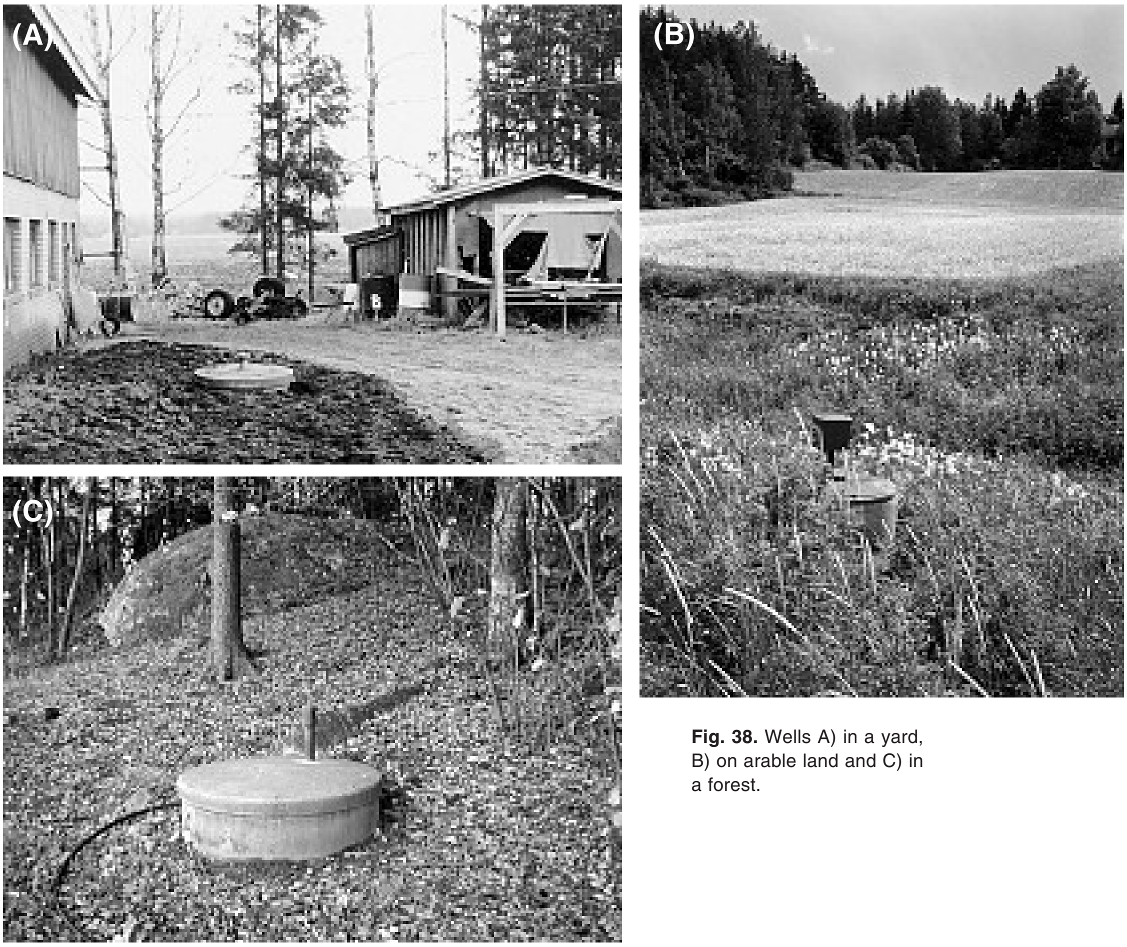


Fig. 38. Wells A) in a yard, B) on arable land and C) in a forest.

ed elsewhere (Fig. 39). A high colour value, $KMnO_4$ consumption and turbidity were characteristic of the wells on arable land, which also had the highest concentrations of nitrate and aluminium of all the types. It is significant that the water was microbiologically somewhat better in the wells located in forest than in those in other types of environment, and the concentrations of almost all ions were lower in the forest wells (Fig. 39).

There are no lithological sources of nitrogen compounds (Rönkä 1983, Hem 1992), and the natural nitrate concentration in groundwater is less than 0.2 mg l^{-1} (Lahermo and Backman 1999). The elevated concentrations recorded here were therefore derived from a variety of anthropogenic sources, principally waste from animal husbandry and agricultural practices such as the use of fertilisers and animal manure, or leaking feed packages. Domestic waste originating from broken sewage pipes and septic tanks, or in some cases from fur farming, refuse dumps, airports and industrial activities, elevate the concentrations of nitrogen compounds in groundwater (Lehtikangas *et al.* 1995, Hatva and Suomela 1999, Lahermo and Backman 1999).

Nitrogen occurs in soil mainly in organic form, from which it is ionised to ammonium by soil microbes. Concentrations of ammonium will therefore be highest near sources of contamination. In next step, the ammonium oxidises first to nitrite and then to nitrate. Since this nitrification reaction takes place relatively quickly, most of the nitrogen in soil and groundwater will be in the form of nitrate (Åkerla *et al.* 1985, Lehtikangas *et al.* 1995).

The infiltration of nitrogen through the soil zone and into the groundwater depends on the thickness of the unsaturated zone, the stratigraphy of the deposits and the permeability and adsorption capacities of the soil material. In areas where the soil is coarse-grained, the adsorption of mineral material is low and the nitrogen is efficiently flushed into the groundwater, whereas in poorly permeable clay and silt soils nitrogen loss to the vegetation is effective and the clay particles can also absorb part of the nitrogen (Hatva and Lahermo 1990, Lehtikangas *et al.* 1995).

A large proportion of the nitrogen compounds in groundwater seem to be of agricultural origin (Britschgi 1989). The Spearman correlation (ρ) between NO_3 and K, an indicator of the use of potassium nitrate fertiliser, was much higher in the coarse-grained aquifers ($\rho = 0.55$) than in the clay-covered ones ($\rho = 0.23$) or in the whole set of data ($\rho = 0.40$).

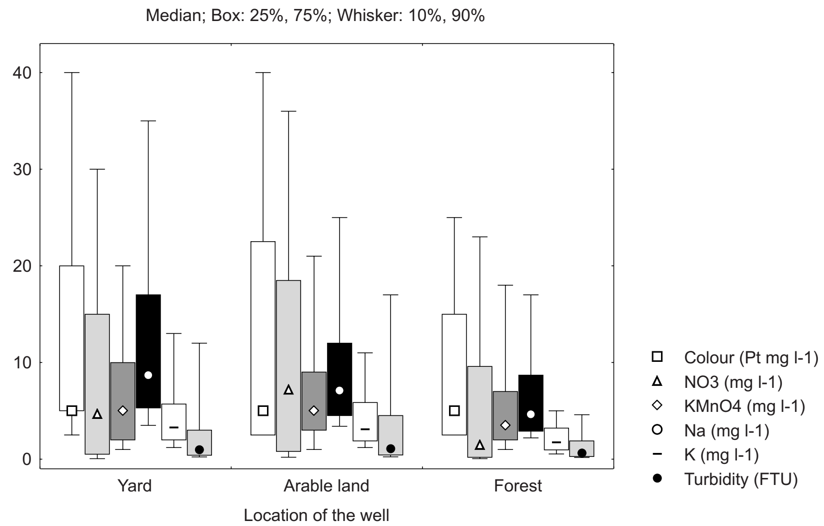


Fig. 39. Box plots of colour, nitrate, $KMnO_4$ consumption, sodium, potassium and turbidity grouped by the primary form of land use near the well.

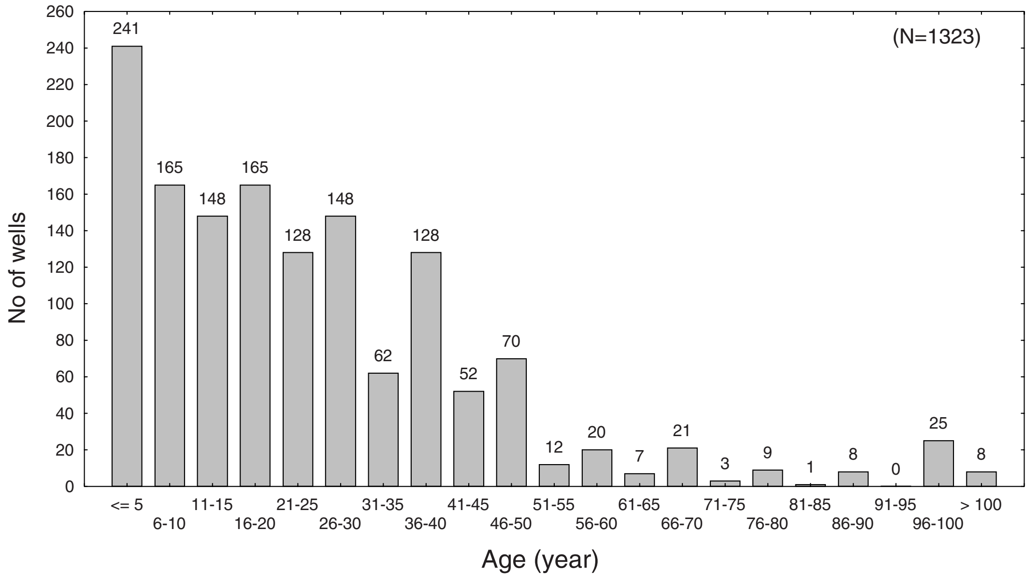


Fig. 40. Age of the wells (N = 1323).

3.6.3 Age of the well

Most of the wells (87 %) were less than 50 years old (Fig. 40), the average age being 27 years and the median 20 years. The stone ring wells were the oldest ones, with an average age of 52 years, the concrete ring wells being 26.6 years old on average and the drilled bedrock wells only 14.9 years old.

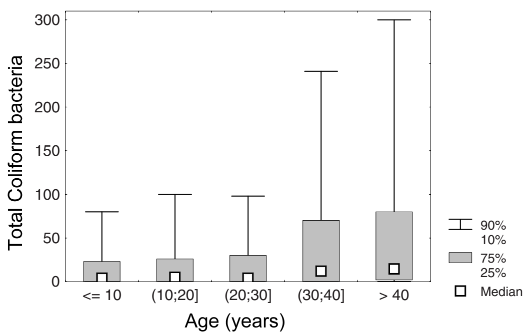


Fig. 41. Age of well versus total coliform bacteria, in dug wells and springs (N = 1093).

The older the well, the higher the count of total coliform bacteria in the water, the lower the pH and alkalinity and the higher the nitrate and sulphate contents (Korkka-Niemi *et al.* 1993). One explanation for this may be that the younger wells are more often bedrock wells. Only the distribution of total coliform bacteria classified by age of the well acts similarly in all types of wells, pointing to a direct effect of age (Fig. 40). It is more probable that there will be more total coliform bacteria in the water from old wells (Fig. 41).

3.6.4 Depth of the well

Hyyppä (1984) and Karro and Lahermo (1999) have established that as far as common household wells drilled into the bedrock are concerned, the average concentrations of dissolved elements increase as a function of well depth. As regards shallow dug wells, not much is known about the effect of well depth on water quality. The wells were generally shallow, 72 % of them being less than eight metres deep (Fig. 42). The dug wells were about 6 m deep (mean 5.8 m), the drilled bedrock wells as much as 57 m deep (mean), but the springs only 1.6 m deep (mean).

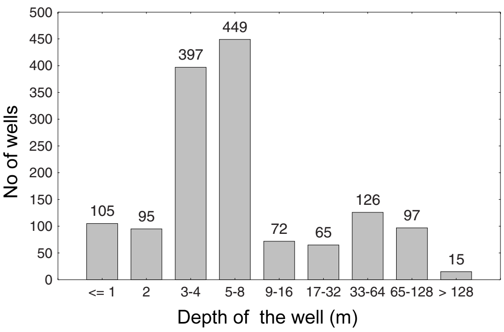


Fig. 42. Depths of the wells, shown on a log(2) scale.

The number of total coliform bacteria, nitrate concentrations, $KMnO_4$ consumption, turbidity, potassium and iron concentrations decreased with increasing depth of the well in the case of the bedrock wells (Fig. 43), while pH, alkalinity and manganese concentration increased with well depth.

The water quality was generally somewhat better in the shallow dug wells (< 4 m) than in the deeper dug wells (Korkka-Niemi *et al.* 1993), and the depth of the well had weakly significant ($p < 0.05$) positive Spearman correlations with pH ($\rho = 0.19$), turbidity ($\rho = 0.27$), alkalinity ($\rho = 0.26$), hardness ($\rho = 0.26$), nitrate ($\rho = 0.16$), chloride ($\rho = 0.15$), manganese ($\rho = 0.13$), sulphate ($\rho = 0.17$), potassium ($\rho = 0.25$) and sodium ($\rho = 0.25$) in the dug wells and springs (N=1017). The relation between depth of dug wells or springs and the chemical composition of the well water is demonstrated in Fig. 44. Similarly pH, alkalinity, EC and hardness increase markedly and chloride and nitrate slightly with increasing well depth.

3.6.5 Well condition and penetration of runoff through the casing

It was estimated during the sampling that only about half of the well structures were in good condition (N = 820) and the other half in satisfactory (N = 518) or poor (N = 83) condition. Bacteria were clearly more abundant in wells of the latter two categories, and similarly, the worse the condition of the well, the higher were the colour and $KMnO_4$ consumption of the water (Fig. 45). Chloride, sodium, potassium, iron and manganese concentrations were also higher in the poorly con-

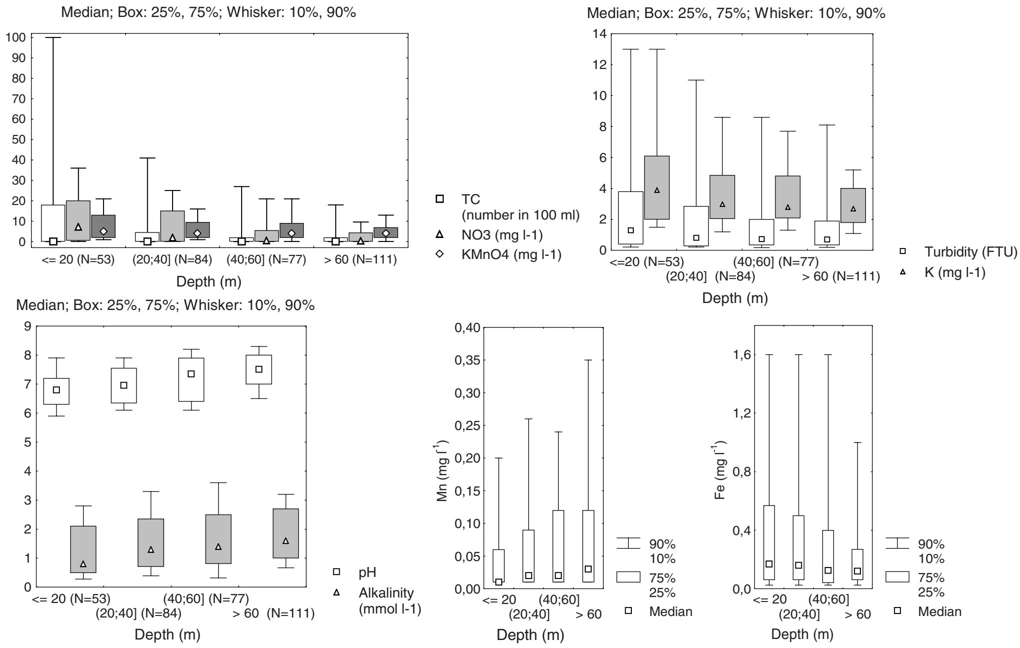


Fig. 43. Water quality variables in drilled bedrock wells by depth of well (N=325).

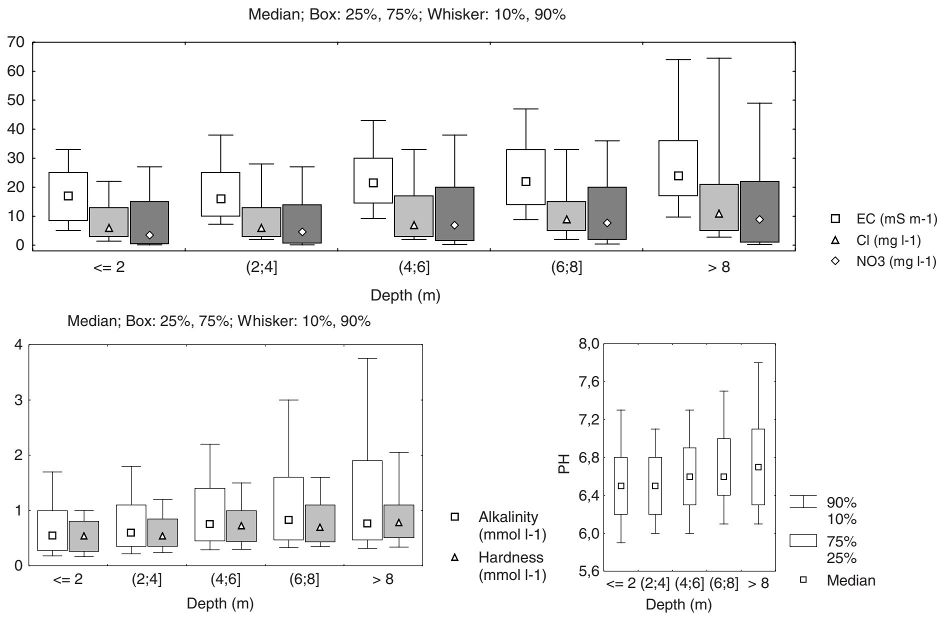


Fig. 44. Water quality variables in shallow dug wells and springs by depth of well (N = 1096).

structed wells. These deleterious changes in water quality may be attributed mainly to surface water penetrating into the well more easily when it is poorly constructed.

The questionnaire results likewise suggested that surface water had penetrated into one third of the wells. Often it had been observed flowing through the joints or from upper parts of the structure. As surface water infiltration was scarcely noticeable in the bedrock wells, they were excluded from the data for this purpose, after which the rest of the wells were divided into two groups on the basis of field observations: 1) those in which surface water penetration was highly possible, and 2) those in which no surface water effect was visible.

Wilcoxon scores (Z) and median scores (X^2) were calculated using the STATISTICA 5.0 procedure. The quality variables differed significantly

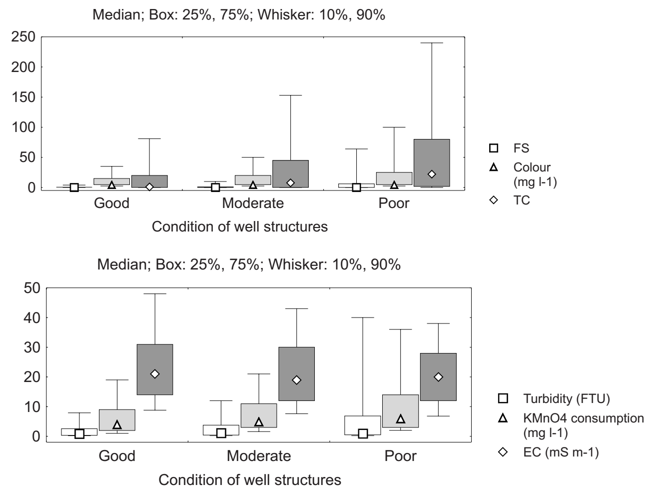


Fig. 45. Water quality variables by condition of the well structure.

($p < 0.05$) between groups 1 and 2 (Table 18).

As seen in Table 18, highly significant differences ($p < 0.001$) were found between groups 1 and 2 in turbidity and $KMnO_4$ consumpion, aluminium content, total coliforms, faecal coliforms and faecal streptococci (confirmed), which can thus be used as indicators of surface water penetration into a well. The statistical significances of the differences in the variables between these two groups are shown in Fig. 46.

3.6.6 Sources of contamination

Observations were made in connection with the sampling in autumn 1990 regarding possible sources of pollution around the wells (Table 19 and Fig. 47). The most common sources were arable land (fertilisation), cowsheds, outdoor toilets, piggeries and roads. The owners of the wells also reported on other potential sources of pollution such as peat bogs, pastureland, gardens, ditches, ponds etc.

Table 18. Wilcoxon scores and median scores for dug wells and springs (N = 1096). Group 1 (surface water penetration highly possible)contains 747 wells, except for the determination of faecal coliform bacteria (N = 147), and group 2 (surface water effect not visible) contains 349 wells (faecal coliforms, N = 87). Highly significant differences ($p < 0.001$) are indicated in bold type.

	Mann-Whitney U-test		Median Test	
Variable	*Z*	*p*	X^2	*p*
TC	**5.93**	**< 0.0001**	**28.02**	**< 0.0001**
FC	**4.35**	**< 0.0001**	**17.94**	**< 0.0001**
FS	2.29	0.0219		
FS (confirmed)	**3.95**	**0.0001**	**15.91**	**0.0001**
Alkalinity	2.59	0.0097	5.96	0.0147
Hardness	2.26	0.0232		
Colour	**4.14**	**< 0.0001**	4.80	0.0285
Turbidity	**3.96**	**< 0.0001**	**14.42**	**0.0001**
$KMnO_4$	**4.78**	**< 0.0001**	**19.28**	**< 0.0001**
F	2.32	0.0208		
Fe	**3.44**	**0.0006**		
Al	**3.95**	**< 0.0001**	**8.02**	**0.0046**

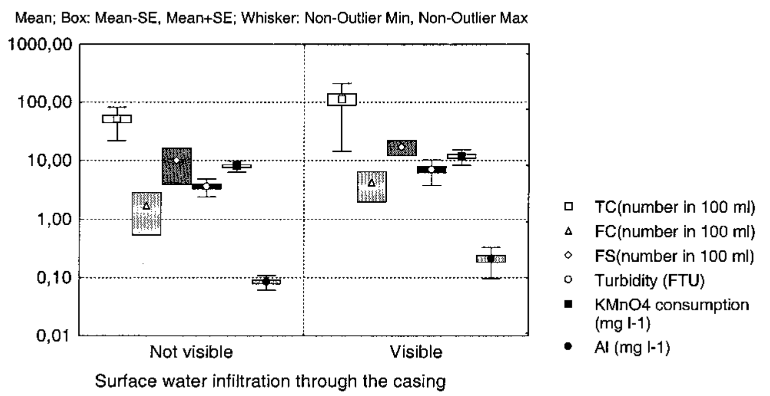


Fig. 46. Water quality variables indicating surface water infiltration into the well through poorly made joints or damaged casings.

The contaminating influence of agricultural settlement on groundwater was reflected in a marked increase in electrical conductivity, nitrate, chloride and potassium and higher incidences of total coliform bacteria, faecal coliform bacteria and faecal streptococci (Table 20).

The effects of polluting factors such as cowsheds and piggeries, outdoor toilets, arable land and roads on well water quality are shown in Fig. 48. Cowsheds and piggeries led to an increase in number of bacteria and nitrates, piggeries as an increased sulphates and cowsheds also to a lower pH and as increased alkalinity, hardness and $KMnO_4$ consumption, together with higher concentrations of chlorides, potassium and manganese Fig. 48a and Fig. 48b. The presence of arable land was particularly associated with higher concentrations of nitrate and ammonium in the well water Fig. 48d. The main sources of nitrates are nitrate fertilisers, ammonium fertilisers and organic manure used for fertilisation (Kemppainen 1982, Martikainen 1987). More faecal coliform bacteria and faecal streptococci were counted in the wells located in the fields than elsewhere Fig. 48d, while the wells located near roads showed a slight increase in chloride and sodium concentrations and in electrical conductivity Fig. 48c.

Lahermo (1970) observed the contaminating influence of agricultural settlement on groundwater in the form of a marked increase in conductivity and a decrease in the pH level, and identified chlorides and nitrates to be reliable indicators of contamination, since their concentrations were several hundreds of times higher in contaminated

Table 19. Possible sources of contamination observed near the well at the time of sampling in 1990 and their polluting effects as estimated by the well owners.

Pollution source	Total number of wells	Estimation of polluting effect		
		Obvious	Possible	No
Cowshed	532	15	99	418
Piggery	84	3	10	71
Outdoor toilet	332	5	80	247
Arable land	735	62	247	426
Road	73	4	40	29

Table 20. Chemical composition of well water 1) when there was a possible source of pollution nearby, and 2) when there were no sources of pollution nearby. The bedrock wells are not included in the table. Units are the same as in Table 2.

	Pollution source nearby (< 100m) (N = 378)		No pollution sources nearby (N = 274)	
Variable	Mean	Median	Mean	Median
TC	94.09	12.0	52.86	6.0
FC	6.89	0.0	0.598	0.0
FS	31.60	1.0	3.53	0.0
EC	26.7	20.0	21.9	18.0
Colour	20.80	5.00	15.88	5.00
Turbidity	6.34	1.05	4.41	1.00
$KMnO_4$	10.03	5.00	8.81	5.00
Cl	20.00	8.00	13.33	7.00
NO_3	14.96	6.55	11.13	4.80
NO_2	0.049	0.005	0.013	0.005
NH_4	0.287	0.05	0.121	0.05
K	9.352	3.40	4.616	2.95
Mn	0.115	0.015	0.063	0.01
Fe	0.816	0.180	0.548	0.180

Fig. 47 cont.

Fig. 47. Sources of pollution near wells: A) cowshed, B) arable land, C) roads.

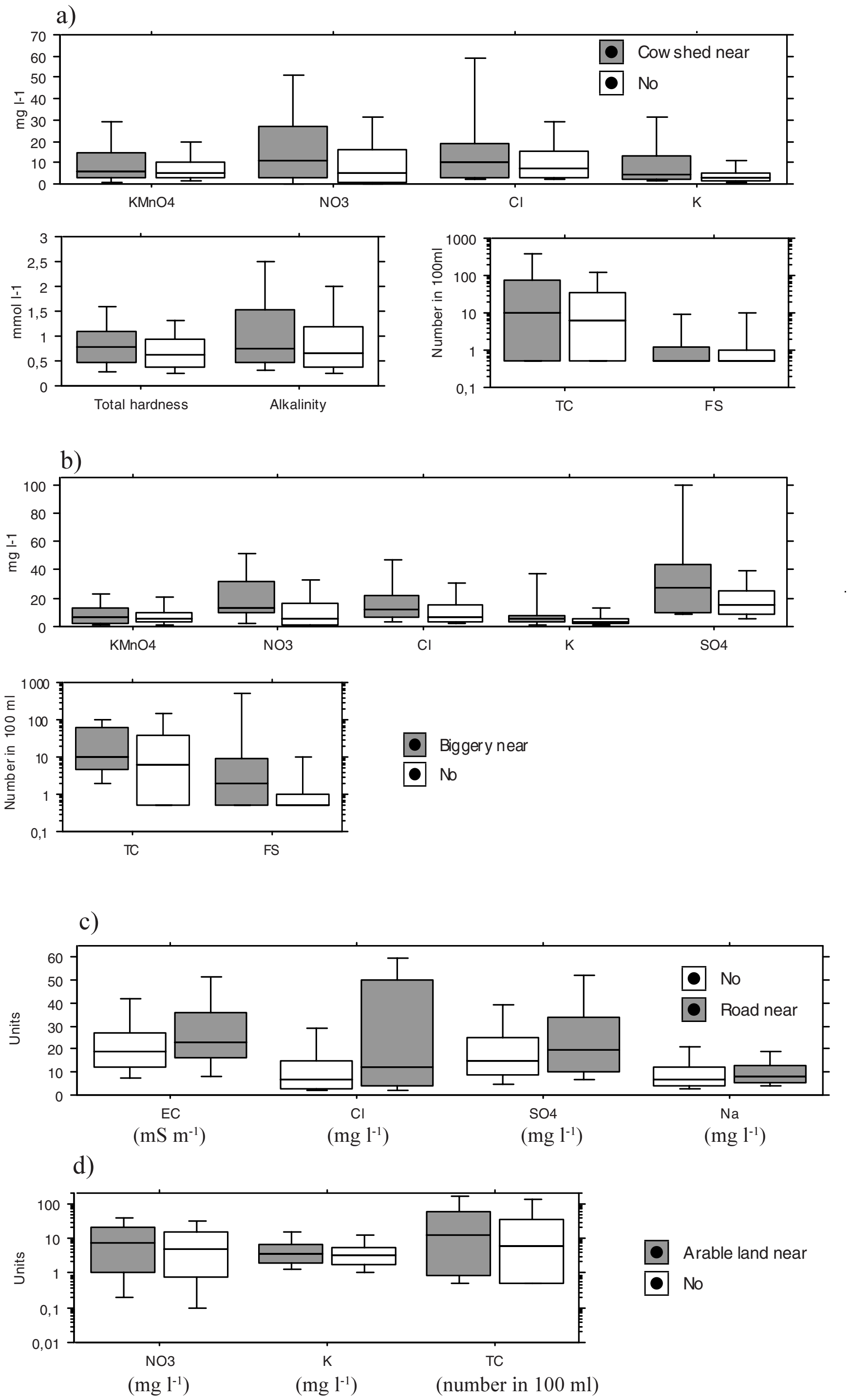


Fig. 48. Water quality variables affected by polluting factors.

water than in intact water. Backman *et al.* (1999) concluded that local pollution generally increases the amounts of dissolved solids in groundwater, and that this increase was most pronounced in the concentrations of NO_3, Cl, Na and K, which are low in natural groundwater. Analogously, Pönkkä (1981) detected the pollution in esker aquifers from increases in the concentrations of electrolytes, i.e. NO_3, Cl, Ca, Mg, HCO_3 and SO_4 and in total hardness, while Rajala (1995) observed human impact on springs in till aquifers arising from road salting and fertilisers.

The disposal of human waste constitutes a serious problem for the quality of drinking water (Heinonen-Tanski and Rajala 1998). Altogether 88 % of the houses and 211 of the cattle sheds located close to the wells studied here had a sewage system, and most of the waste water from houses (89 %) infiltrated into the soil, while the rest was discharged into rivers, brooks, ditches or lakes (6 %) or into septic tanks or municipal sewage systems (6 %). Animal waste was more often stored initially in slurry tanks or other manure reservoirs (37 %) and was allowed to infiltrate directly into the soil to a lesser extent (60 %). There were some differences in water quality between the wells influenced by sewage water infiltration (< 100 m away) and other wells (see chapter 4).

4 Discussion

4.1 Grouping of the water quality variables

4.1.1 Correlation between variables

The purpose of the grouping was to determine which variables represent the controlling factors behind well water quality and which serve best to indicate the cumulative impact on different scales. The common procedure for determining whether or not two variables are related in a population is to calculate correlation statistics from the set of samples (Frapportti 1994). Here the non-parametric Spearman rank correlation coefficient (Spearman ρ) suggested by Swan *et al.* (1995) and Ranta *et al.* (1991), for example, was computed for all the hydrogeochemical variables and for distance from the seashore (Table 21).

Although the microbiological indicators are relatively independent of the dissolved components, total coliform bacteria have a weak positive correlation with faecal coliforms, faecal streptococci, $KMnO_4$ consumption and aluminium, while faecal coliforms and faecal streptococci correlate weakly with $KMnO_4$ consumption and aluminium.

Electrical conductivity (EC) is strongly correlated with the concentrations of the main anions and cations, i.e. HCO_3 (alkalinity), Cl, SO_4, hardness (Ca and Mg), Na and K, and weakly with F and Mn. Alkalinity is strongly correlated with pH, EC and hardness and moderately or weakly with Na, F, K, Mn, SO_4 and Cl concentrations, the magnitude of the coefficient decreasing in this order. Hardness (mainly Ca and Mg) is strongly correlated with EC, alkalinity and Cl, moderately with Na, K and SO_4 and weakly with pH and Mn. The pH values have fairly weak positive correlations with EC, hardness, F and Na and weak negative correlations with NO_3 and Al. Chloride is strongly correlated with EC, hardness and Na, moderately with SO_4 and K and weakly with alkalinity, while fluoride has a significant, but moderate correlation with alkalinity and Na, weak positive correlations with EC and pH and a negative correlation with NO_3 concentrations. Potassium is strongly correlated only with EC, moderately with hardness, Na and SO_4 and weakly with Cl, $KMnO_4$ consumption, alkalinity, NO_3 and Mn. Sodium has significant strong correlations with EC and Cl, moderate ones with hardness, F, K and SO_4 and a weak one with pH and Mn. Finally, sulphates are strongly correlated with EC, moderately with hardness, Na, Cl and K and weakly with alkalinity.

Turbidity has strong positive correlations with Fe and colour and weak correlations with Mn, Al, $KMnO_4$ consumption and NO_2. Colour has marked correlations with turbidity, Fe and $KMnO_4$ consumption and weak ones with Mn, Al and NO_2. $KMnO_4$ consumption shows moderate correlations with colour and aluminium and correlates slight ones with K, Fe and Mn, while aluminium has significant correlations with all the variables, that suggest surface water contamination, as manifested by high numbers of total coliform bacteria and elevated levels of turbidity, colour and $KMnO_4$ consumption. The same variable also has positive correlations with Fe and turbidity and a weak negative one with pH.

Iron shows fairly strong correlations with turbidity and colour, as mentioned before, a moderate correlation with Mn and weak correlation with

Table 21. Spearman ρ matrix for the water quality properties and concentrations of dissolved constituents. Only statistically significant correlation coefficients ($p < 0.01$) are presented. High (> 0.6) and moderate (0.4–0.6) correlation coefficients are indicated in bold type. The number of samples included in the table was 1421, with exception of colour (N = 1385).

	TC	FC	FS	EC	Alk.	Turb.	Colour	$KMnO_4$	Hard.	pH	Al	Cl	F	Fe	K	Mn	Na	NH_4	NO_2	NO_3	SO_4
Tot. coliform																					
Faec. coliform	0.329																				
Faec. strept.	0.353	0.294																			
EC		0.071																			
Alkalinity		0.067		**0.705**																	
Turbidity	0.165	0.159	0.153	0.207	0.266																
Colour	0.203	0.147	0.116	0.221	0.296	**0.707**															
$KMnO_4$	0.316	0.229	0.260	0.271	0.206	0.309	**0.512**														
Hardness		0.095		**0.866**	**0.692**	0.251	0.204	0.206													
pH	-0.111			0.396	**0.716**	0.157	0.164		0.385												
Al	0.329	0.225	0.237	-0.063	-0.144	0.383	0.340	0.510		0.286											
Cl				**0.723**	0.322	0.090	0.105	0.153	**0.605**	0.095											
F				0.355	**0.454**	0.259	0.199	0.090	0.259	0.395	0.149	0.190									
Fe	0.150	0.113	0.151	0.166	0.194	**0.836**	**0.695**	0.346	0.190		0.348	0.079	0.173								
K	0.160	0.118	0.145	**0.613**	0.360	0.154	0.216	0.382	**0.560**	0.064	0.068	**0.477**	0.074	0.165							
Mn			0.077	0.361	0.354	**0.454**	**0.436**	0.336	0.343	0.113	0.185	0.232	0.274	**0.521**	0.325						
Na				**0.776**	**0.596**	0.191	0.176	0.175	**0.596**	0.379	-0.089	**0.646**	**0.461**	0.141	**0.432**	0.338					
NH_4				0.214	0.160	0.176	0.194	0.178	0.160	0.086		0.193	0.090	0.224	0.231	0.368	0.213				
NO_2	0.093	0.139	0.093	0.213	0.164	0.322	0.340	0.251	0.199		0.159	0.168		0.355	0.235	0.330	0.161	0.302			
NO_3	0.137	0.071	0.065	0.157	0.182	-0.225	-0.203		0.195	0.341		0.256	-0.335	-0.223	0.399	-0.190			0.144		
SO_4	0.110			**0.639**	0.314	0.192	0.184	0.258	**0.587**	0.097	0.089	**0.515**	0.233	0.125	**0.533**	0.210	**0.533**	0.065	0.127	0.230	
D	0.083			-0.267	-0.206	-0.282	-0.224	-0.212	-0.188	0.198	-0.265	-0.239	**-0.530**	-0.186		-0.115	-0.347			0.192	-0.347

TC: Total Coliform bacteria
FC: Faecal Coliform bacteria
FS: Faecal Streptococci
D: Distance from the seashore

$KMnO_4$, Al and NO_2. This is partly explained by the fact that iron occurs in complexes with humus compounds (Hem 1992, Lahermo 1970). Ferric iron may also be bound to clay minerals suspended in well water (Carrol 1958). Manganese has its strongest correlation with iron and moderate or weak correlation with EC, alkalinity, turbidity, colour, $KMnO_4$ consumption, hardness, K, Na, NH_4 and NO_2.

The nitrogen compounds seem to be fairly independent of the other variables. Ammonium has weak correlations with Mn and NO_2, and nitrite is correlated weakly with turbidity, colour, Fe, Mn and NH_4, while nitrite has a weak positive correlation with K and negative ones with pH and F.

There is a strong negative correlation (ρ = -0.53) between proximity to the sea and the F content of the well water, which can be explained by the spatial distribution of fluoride concentrations in well water, being found mainly in rapakivi areas, which are situated in the coastal areas of Finland. In addition, the concentration of fluoride in seawater is fairly high (Piispanen 1991), which could partly explain the correlation. On the other hand, the correlation between fluoride and the 'marine' variable, chloride, is not a strong one (ρ = 0.19), suggesting that the bedrock is the most important source of fluoride. The distance from the sea has a weak negative correlation with almost all the variables, implying that there is a slight marine effect on the groundwater in the wells.

4.1.2 Principal components of well water quality

As the matrix showed the existence of strong correlations between certain variables, the data were submitted to principal component analysis (PCA) in order to study the interrelated correlation patterns. PCA was carried out with all the variables, using the STATISTICA 5.0 program, the variables being log10-transformed (except pH) and the microbiological variables log10(x+1)-transformed prior to the analysis.

The values for each variable with respect to the various communalities are shown in Table 22. The term communality reflects the proportion of variance explained by the common factors. Since the first five factors had eigenvalues >1 for the whole set of chemical data for 1990, these factors were chosen for further analysis (cf. Ranta *et al.*1991).

Table 22. Communality of variables. PCA= Principal Component Analysis.

PCA factor	Eigenvalue	% total variance	Cumulative %
1	5.76	27.41	27.41
2	3.13	14.91	42.33
3	2.37	11.30	53.63
4	1.51	7.20	60.84
5	1.23	5.87	66.70

As the table also shows, these first five factors explain 67 % of the total variance. The results of the matrix for these factors are presented in Table 23, where the values in have been improved by applying 'varimax normalised' rotation to the axes, which maximises the variance of the factors.

As the factor scores are standardised, i.e. they have a mean of zero and a standard deviation of one, the scores can be related to the intensity of the hydrogeochemical process described by each factor (Dalton and Upchurch 1978). Extreme negative values reflect wells that are essentially unaffected by the hydrogeochemical processes, and positive scores reflect the wells that are most markedly affected. Near-zero scores apply to wells affected to an average degree by the hydrogeochemical process represented by the particular factor. Based on the patterns of factor loading, each factor can be interpreted as a specific single or multiple hydrochemical process.

Factor 1 (salinity) has high loadings on electrical conductivity, total hardness, Cl, SO_4, K and Na (Table 23 and Fig. 49), Factor 2 (humus-redox) is loaded heavily with respect to turbidity, colour, $KMnO_4$ consumption, Al and Fe and moderately with respect to Mn, while Factor 3 (pH) has positive loadings on pH, alkalinity and F and a negative loading on NO_3. The positive pH and alkalinity loadings of this factor reflect a good buffering capacity, while the positive F loading indicates the existence of F-rich bedrock, which is the main reason for the high F concentrations in the groundwater (Wäre 1960, Lahermo 1971). Factor 4 (pollution) includes all the indicator bacteria analysed here, reflecting faecal pollution of well water, and correspondingly, Factor 5 (contamination) includes mainly NO_2, NH_4 and Mn, the reducing compounds. Nitrite and ammonium ions have no significant lithological source, and may be associated mainly with fresh contamination by organic matter and fertilisers.

Table 23. Varimax-rotated, standardised factor matrix.

	Factor loadings				
Variable	Factor 1	Factor 2	Factor 3	Factor 4	Factor 5
TC	0.048	0.164	-0.142	**0.707**	-0.045
FC	0.021	0.103	0.086	**0.700**	0.009
FS	-0.048	0.032	-0.010	**0.735**	0.096
EC	**0.884**	0.089	0.315	0.017	0.164
pH	0.158	-0.112	**0.855**	-0.016	0.077
Alkalinity	0.481	0.040	**0.700**	0.073	0.230
Hardness	**0.805**	0.096	0.250	0.045	0.164
Colour	0.082	**0.828**	0.159	0.120	0.192
Turbidity	0.068	**0.850**	0.178	0.081	0.121
$KMnO_4$ consumption	0.283	**0.494**	-0.159	0.307	0.057
SO_4	**0.779**	0.127	-0.077	0.040	-0.103
Cl	**0.792**	0.032	0.053	-0.105	0.111
F	0.250	0.248	**0.588**	-0.022	-0.239
NO_3	0.391	-0.274	**-0.666**	0.151	0.111
NO_2	0.165	0.224	-0.066	0.045	**0.706**
NH_4	0.122	0.078	-0.018	0.036	**0.780**
Na	**0.690**	0.077	0.480	-0.051	0.083
K	**0.700**	0.065	-0.166	0.161	0.366
Fe	0.010	**0.854**	0.103	0.014	0.229
Mn	0.234	**0.486**	0.213	-0.074	**0.483**
Al	0.021	**0.653**	-0.308	0.296	-0.246

Factor 1 (salinity) explains 27 % of the variance, factor 2 (humus-redox) 15 % and factor 3 (pH) more than 10 %. Factor 4 (pollution) explains 7 % and factor 5 (contamination) 6 % (Table 22). One third of the variance in the model remains unexplained by these five factors.

The PCA factor scores can now be grouped by known categories such as type of well (Fig. 50), surficial sediment around the well (Fig. 51) and condition of the well (Fig. 52) in order to examine the information contained inside the factors.

As shown in Fig. 50, factor 1 is strongest in the bedrock wells and least important in spring wells. Also, as in Fig. 51, the better the permeability of the surface sediment, the less important factor 1 becomes. These observations confirm the implica-

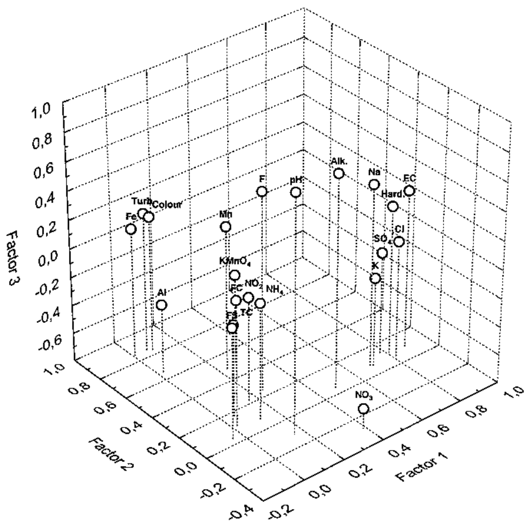


Fig. 49. PCA factor loadings of the well water variables, expressed in a 3-dimensional space using factors 1, 2 and 3 as the axes. The number of wells included was 1385.

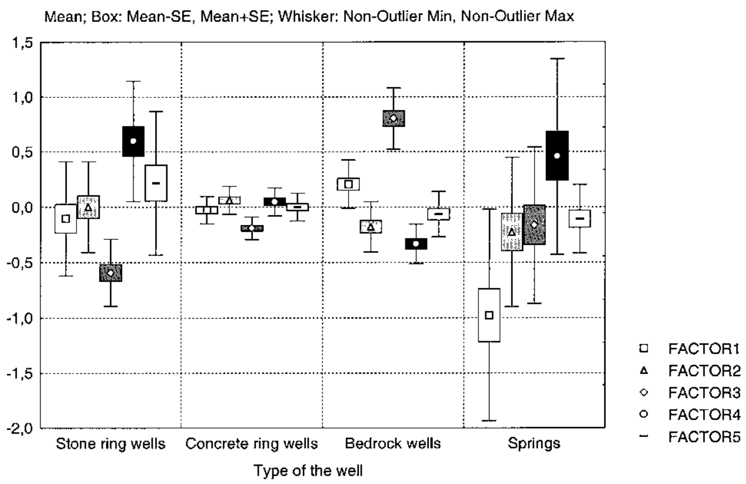


Fig. 50. PCA factor scores for the well types.

tions of factor 1 as a geological factor strongly connected with the residence time of the water in geological material. This can be considered an extra-regional factor.

Factor 2 indicates the effect of surface runoff infiltration through the well casing, which can be seen still better in Fig. 53, where the loading of factor 2 with respect to a visible surface runoff effect is much higher. It is also clear from Fig. 51 that the factor scores are generally higher in aquifers covered with clay deposits than in other groups where factor 2 predominates. This can be explained by the specific reducing environment in these wells. Factor 2 is thus mainly regional (redox conditions in the aquifer) and partly site-specific, due to humus complexes and clay particles from infiltrating surface runoff.

Factor 3 is dominant in bedrock wells but is of little meaning in shallow, poorly constructed stone ring wells (Fig. 50). This confirms its meaning as

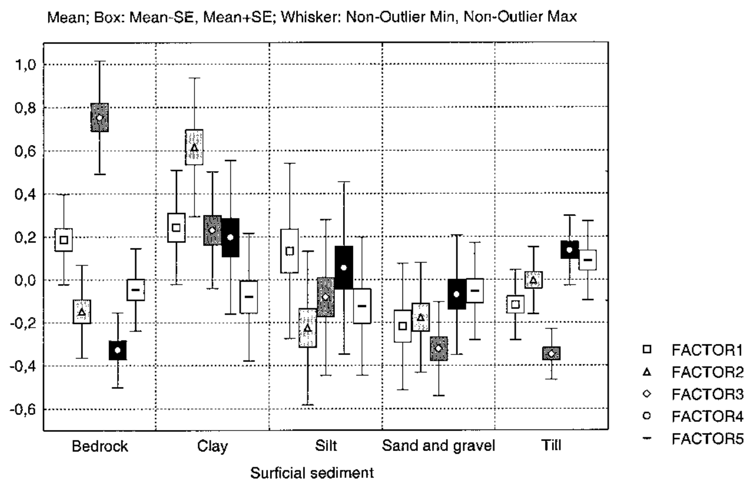


Fig. 51. PCA factor scores grouped according to the nature of the surficial sediment.

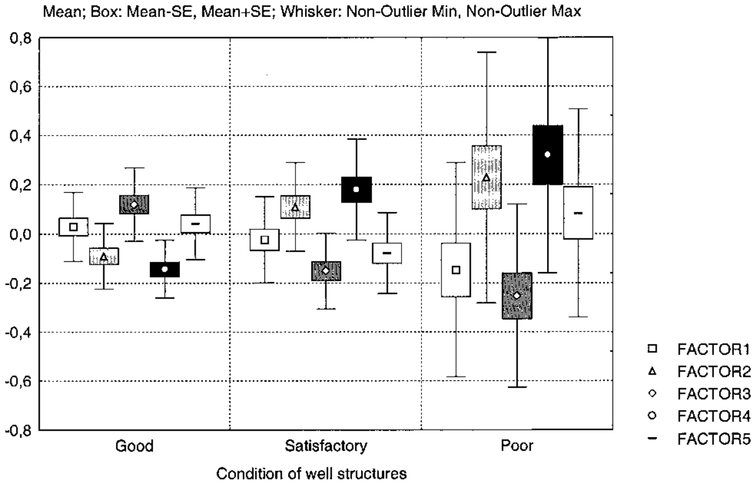


Fig. 52. PCA factor scores grouped by the condition of the well structures.

being that of describing pH and buffering capacity. This factor is predominantly extra-regional.

Factor 4 seems to be relatively dependent on the condition of the well and on the penetration of surface water into it (Fig. 52 and 53). The poorer the condition of the well structures, the more important this factor becomes. Factor 4 is also dominant in old stone ring wells and springs and is of minor importance in bedrock wells. All this confirms its meaning as an indicator of faecal contamination which cannot be detected in any other way, or on the other hand, bacteria originating from decaying plants. Factor 4 is definitely site-specific.

Factor 5 is relatively independent of the type of well, the nearby surface sediments, the condition of the well structure and the penetration of surface runoff into the well. It is weighted somewhat more heavily, however, on wells in till areas, poorly constructed wells and old stone ring wells. This is in line with the idea that it indicates site-specific contamination. It may also be partly a regional factor, as the prevailing redox environment affects the existence and activity of nitrogen and manganese bacteria.

According to Lahermo *et al.* (1990), the most powerful modifier of shallow groundwater chemistry is pollution (the 'contamination factor': NO_3, K, EC, Na, Cl, Ca, Mg, CO_2 and SO_4) followed by marine influence (the 'salinity factor': EC, Na, Cl, Ca and Mg) and geographical elevation (the 'elevation factor': SO_4, SiO_2 and F). The secondary factors were the effect of dissolved organic matter (the 'humus factor': colour, $KMnO_4$ consumption and O_2) and redox conditions (the 'redox factor': colour, Fe, Mn and O_2). Water-rock interaction and the buffering reactions controlling the pH of the water (the 'pH factor': pH, HCO_3 and CO_2) seemed to have been of minor significance.

The strongest modifier of well water quality identified here was the extra-regional or regional salinity factor (EC, total hardness, Cl, SO_4, K and Na). The variables subsumed in this factor are partly same as those listed by Lahermo *et al.* (1990), but here the factor represents a more extra-regional, 'geological' effect connected with the residence time of the water in the aquifer rather than suggesting contamination as Lahermo *et al.* (1990) concluded. However, some of the elements in this factor can be of atmospheric, marine or anthropogenic origin. The first secondary factor identified here, the partly regional and partly site-specific effect of dissolved organic compounds and the redox environment (turbidity, colourr, $KMnO_4$ consumption, Al, Fe and Mn) is again much like the secondary factors of Lahermo *et al.* (1990), while another secondary factor was the extra-regional 'pH factor' (pH, alkalinity and F) reflecting the buffering capacity of the well water. The present site-specific 'pollution factor' (total

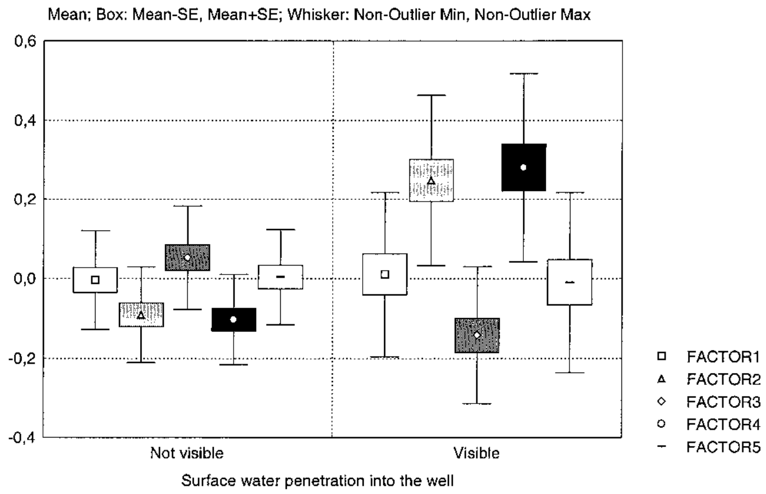


Fig. 53. PCA factor scores grouped by the existence of visible runoff penetration into the well.

coliforms, faecal coliforms and faecal streptococci) and 'contamination factor' (NO_2, NH_4 and Mn) are of minor significance.

4.2 Microbiological indicators of well water quality

4.2.1 Indicator bacteria

Infectious diseases caused by pathogenic bacteria, viruses and protozoa are the most common and widespread health risk associated with drinking water (WHO 1996), the risks occasioned by inorganic impurities being much smaller (Martikainen 1987, WHO 1996). Although the hygienic quality of the groundwater in Finland is generally high (Martikainen 1987), seven waterborne epidemics were identified in Finland in 1998, affecting about 8000 people (Miettinen *et al.* 1999). In addition to epidemics of this kind, there can be problems especially with private household wells because of the low hygienic quality of the water (Martikainen 1987).

The recognition that faecally polluted water is responsible for enteric diseases has led to the development of highly sensitive methods for verifying that drinking water is free of faecal contamination. The use of normal intestinal bacteria as indicators of faecal pollution rather than the pathogens themselves is universally accepted for the purposes of monitoring and assessing the microbial safety of drinking water (WHO 1996).

Coliform bacteria were detected in over 60 % of the 1421 wells studied in 1990, and faecal coliforms in 10 %, including a high proportion of presumptive *E. coli,* 82 % (Table 24). This latter figure was nevertheless lower than that reported by Niemi and Niemi (1991) for surface water samples, where it was as high as 96 % in pristine areas and 91 % in agricultural areas.

Faecal streptococci were thus detected in one third of the 1421 wells and confirmed in 262 of these. In 138 cases faecal streptococci were also confirmed to be *entrococcus*. The proportion of confirmed faecal streptococci (*Enterococcus*) was nevertheless only 20 % of those counted (Table 24 and Fig. 54), which is much lower than the figures obtained by Niemi and Niemi (1991) in surface water (44–82 %).

When the ratio of confirmed to presumptive faecal streptococci is high a faecal origin is most probable. In more than half of the well water samples where faecal streptococci had been counted none of them were confirmed, however, and there were only 80 wells where all the presumed faecal streptococci were confirmed as such (Table 25).

Table 24. Proportions of presumptive *E. coli* among faecal coliforms and of confirmed faecal streptococci among presumptive faecal streptococci in positive well water samples taken in autumn 1990. N = number of wells where faecal coliforms or presumptive faecal streptococci were identified. Σ denotes the sum of bacteria in all the wells tested.

	N	Σ	Confirmation %
Coliform bacteria			81.9
Faecal coliform bacteria	145	3257	
Presumptive *E.coli*	145	2668	
Faecal streptococci			19.8
Presumptive	263	15348	
Confirmed	263	3043	

Table 25. Confirmation of faecal streptococci in the well water samples.

Ratio: Confirmed FS versus presumptive FS	Number of wells
0	124
> 0 and ≤ 0.5	28
> 0.5 and ≤ 1	30
1	80

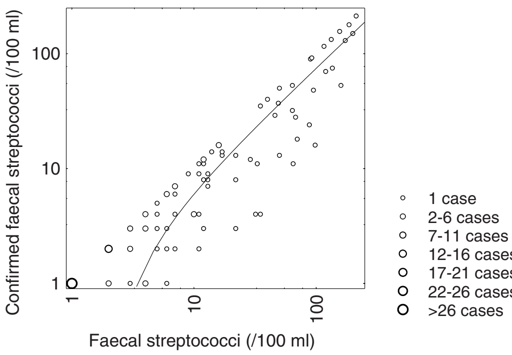


Fig. 54. Confirmed versus presumptive faecal streptococci in the same sample. The number of cases included is 259, as three extreme values had to be omitted from the data set.

4.2.2 Factors affecting the microbial quality of well water

The high incidence of total coliform bacteria (> 10 / 100 ml) in the absence of faecal coliforms in the same sample may indicate that the bacteria were of plant origin. There was visible infiltration of runoff water into 37 % of the wells having total coliforms but no faecal coliforms (N = 419), whereas the proportion for the whole set of wells was only 27 %. Also, the dug wells or springs having faecal coliforms seemed more often to be situated less than 100 m from an outdoor toilet, area of arable land or cowshed (Table 26).

Since the wells where some faecal streptococci were detected but none were confirmed were often situated in arable land, it may be assumed that the water was not faecally polluted in these cases, as the bacteria in the sample could just as well have been of plant origin. There were also wells with high numbers of FS but no FC in the same samples (275 out of a total of 374 positive wells). Counts of faecal streptococci are usually much higher than those of faecal coliforms in the manure of domestic animals, whereas the opposite is true of human faeces (Lahti 1991), and thus the ratio of faecal coliforms to faecal streptococci can be used to deduce the origin of faecal contamination in well water samples.

It is known, however, that faecal streptococci are more resistant than coliform bacteria, so that they may represent a remnant of past faecal pollution. It has also been observed that coliform bacteria can be absorbed into soil more efficiently than faecal streptococci (Gerba 1984, Martikainen 1987, Lahti 1991).

Faecal bacteria move rapidly through the soil if there is rainfall or irritation as an active driving force (Howell *et al.* 1995, Smith *et al.* 1985). They persist in manure deposits, but springs and wells are generally protected until there is some rain to move the bacteria through the soil (Thelin and Gifford 1983). This observation is also discussed in section 3.5.3, which deals with seasonal changes in water quality. As seen in Table 15, there are less total coliform bacteria in spring because of the lack of a contamination source such as cattle manure. By far the highest average numbers of faecal coliforms are recorded in summer, possibly as a consequence of cattle grazing in the fields. This has been discussed earlier by Korkka-Niemi *et al.* (1993).

Although the microbiological indicators are relatively independent of dissolved components (Table 21), total coliforms show a weak positive correlation with faecal coliforms, faecal strepto-

Table 26. Dug wells or springs having faecal coliform bacteria relative to wells where no faecal coliforms were found.

	Wells, where FC were found (N=139)		Wells, where FC were not found (N=954)	
	Number of wells	%	Number of wells	%
Outdoor toilet not near	96	69	761	80
Outdoor toilet <100m	43	31	193	20
Arable land not near	74	53	559	59
Arable land <100m	65	47	395	41
Cowshed not near	85	61	601	63
Cowshed >100m	11	8	100	10
Cowshed <100m	43	31	253	26

cocci, $KMnO_4$ consumption and aluminium, while faecal coliforms and faecal streptococci are weakly correlated with $KMnO_4$ consumption and aluminium. These weak correlations suggest that the occurrence of bacteria is connected with the leaching of organic matter into the wells. Korhonen *et al.* (1996), studying water quality in 150 rural wells in Central Finland, detected correlations, weaker than in the present material, between the number of faecal streptococci, colour and $KMnO_4$ consumption, but no correlations between faecal coliforms and any of the chemical variables analysed.

The existence of bacteria was seen in the present study to be more probable in dug wells than in bedrock wells, and there were generally more bacteria in the wells located close to waste water infiltration points (Fig. 55). Increased occurrence of bacteria in well water also showed a positive correlation with the age of the well. In their water quality programme for the region of Waterloo, Canada, Yessis *et al.* (1996) concluded that there was an increasing risk of having bacterially unsafe water in 1) dug wells compared with drilled wells, 2) households where manure is stored near the well, and 3) wells between 76 and 180 years old as compared with wells younger than 25 years.

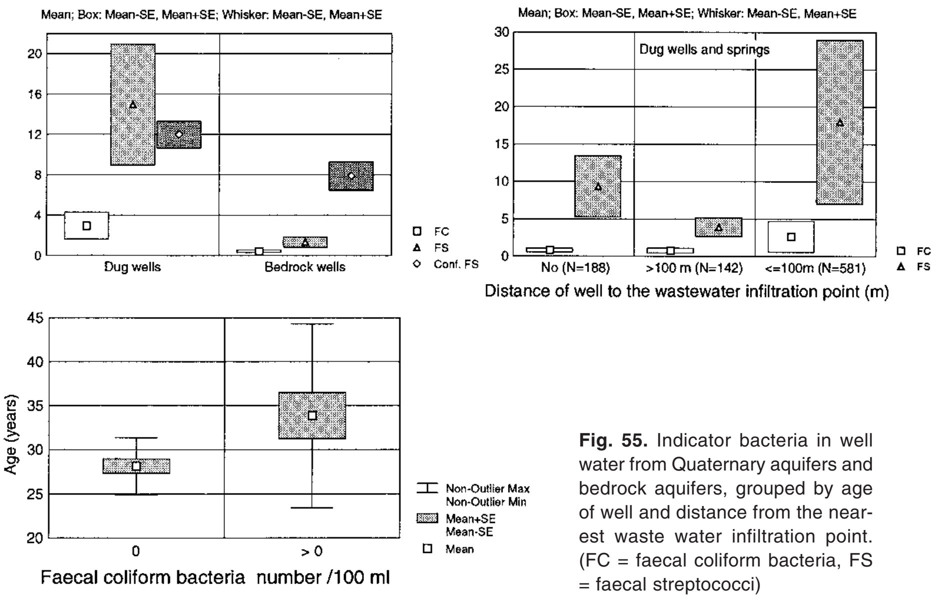


Fig. 55. Indicator bacteria in well water from Quaternary aquifers and bedrock aquifers, grouped by age of well and distance from the nearest waste water infiltration point. (FC = faecal coliform bacteria, FS = faecal streptococci)

4.2.3 Indicator values of bacterial groups

Coliform bacteria (total coliforms) have long been used as suitable microbial indicators of drinking water quality, largely because they occur in higher numbers than pathogens and are easy to detect and count in water samples. Most coliform bacteria belong to the genera *Escherichia*, *Citrobacter*, *Enterobacter*, *Klebsiella*, *Rahnella* and *Hafnia*, but the group is heterogeneous and also includes bacteria which can be found in both faeces and other environments such as nutrient-rich water, soil and decaying vegetation. This group also contains species that are rarely, if ever, found in faeces (Lahti 1991, WHO 1996, Macler and Merkle 2000). Niemelä and Niemi (1989) and Niemi *et al.* (1997) have shown that total coliform bacteria are not suitable as indicators of faecal contamination in surface water, because of the frequent occurrence of environmental coliform organisms. In the present well water samples, total coliforms usually seemed to indicate intrusion of surface water into the wells, as also observed earlier by Lahti (1991).

The group of faecal (thermotolerant) coliforms comprises the genus *Escherichia* and, to a lesser extent, certain species of *Klebsiella*, *Enterobacter* and *Citrobacter*. Only *E. coli* is specifically of faecal origin (WHO 1996). According to Niemelä and Niemi (1989), a simple faecal coliform count works well to demonstrate faecal contamination in our subarctic pristine surface water environments. Niemi and Niemi (1991) noted that in faecal coliforms ditches, brooks and natural ponds in pristine and agricultural areas were generally composed of presumptive *E. coli,* and essentially the same situation was found here. Niemi and Niemi (1991, 2000) have shown that the cleaner the environment is, the more suitable faecal coliforms are as a measure of the occurrence of presumptive *E. coli* and thus as an indicator of faecal contamination. As *E. coli* is found in large numbers in the faeces of humans and of nearly all warm-blooded animals; it serves as such as a reliable index of recent faecal contamination of potable water (WHO 1996). The rest of the faecal coliform bacteria in household wells may be of other origin, e.g. *Klebsiella,* which is found on wood (Martikainen 1987). The best indicator of septic contamination thus seems to be *E. coli,* and to a lesser extent the faecal coliform bacteria, as also mentioned earlier by WHO (1996). Even *E. coli* has not proved to be a perfect indicator of the hygienic quality of household water, however, because according to Korhonen *et al.* (1996) and Heinonen-Tanski *et al.* (1999), species of potential enteric pathogenic bacteria have been found in well water even when coliforms were lacking.

The term ‘faecal streptococci’ (nowadays enterococci) refers to those streptococci generally present in the faeces of humans and warm-blooded animals. Taxonomically, they belong to the genera *Enterococcus* and *Streptococcus.* They may generally be isolated from the faeces of humans and animals, but certain species occur primarily on plant material, too (WHO 1996, Niemi and Niemi 2000). Only 20 % of the faecal streptococci identified in the present work were *Enterococcus* species indicative of faecal origin, and faecal streptococci were sometimes detected in well water samples that contained no coliform bacteria. This may have been because the bacteria existing in the sample were of plant or insect origin (Lahti 1991).

4.3 Classification of the well water

The well water samples studied here were divided into geochemically homogeneous groups using K-means Cluster Analysis (Rock 1988) in order to identify variables which would be useful for classifying the wells. The data were log-transformed (except pH) and standardised prior to the analysis in order to equalise the scales. The samples formed seven clusters, as described in Table 27, the number being selected so that each variable appeared in approximately one cluster. The properties of the wells in the clusters are described using information from cross-tabulations.

Most of the wells included in cluster I (pH) are bedrock wells (60.5 %), and the common types of bedrock in this cluster are S-type granite and diorite. Most of the wells (78.3 %) are in good condition and are not penetrated by surface water, and the pH, alkalinity and F concentrations of their water are generally high. The geographical distribution of the wells in this cluster is given in Fig. 56.

The concentrations of NO_3, Cl, SO_4 and K are generally high in the water of wells belonging to cluster II (nitrate). A large proportion of the wells dug into silt (38 %), till (35 %) or sand/gravel deposits (33 %) belong to this cluster, but also as much as 20 % of all the bedrock wells. The wells usually extend into coarse-grained deposits, most of them are located on arable land or in yards, and more of them are poorly constructed than in the first cluster. Nitrate seems to be the most common quality problem affecting household wells, especially in Central and Eastern Finland (Fig. 56).

The members of cluster III (low in electrolytes) are often shallow dug wells or spring wells having till, sand/gravel or silt as the surrounding surface sediment. Only a few wells in this cluster are in clay-covered areas. Other sand deposits, fluvial or glaciofluvial deposits, morainic landforms and ice-marginal end moraines are typical locations, while the common bedrock type is granite or rocks of the TTG series or gneiss/schist group. This cluster includes many wells situated in forests, far away from any possible polluting sources, and is dominated by examples from inland areas, and particularly from the northern parts of Finland (Fig. 56). The wells are typically located on eskers and the Salpausselkä end moraine complexes.

About 20 % of the wells in till, sand/gravel and clay areas belong to cluster IV (aluminium), in which the water typically has a high concentration of aluminium and to a lesser extent a high $KMnO_4$ consumption. Littoral deposits, cover till, glaciofluvial deposits and Litorina clays are most common types of surrounding deposit. There are surprisingly large numbers of wells in forest locations (23.3 %), indicating other origins for the aluminium rather than the clay particles present on arable land. A notable proportion of the wells in the rapakivi area (36.6 %) and about 22 % of those in other granite bedrock areas belong to this cluster, as can be seen in Fig. 56, where wells in the southern parts of Finland are shown to predominate.

The 30 heavily contaminated wells in cluster V are equally distributed over all parts of the country (Fig. 56). Only one contaminated well is situating in forest, while the others are on arable land or in yards.

The water of the wells in cluster VI (dissolved) is generally high in electrolytes. Bedrock wells (54.6 %) and wells cutting into confining clay beds predominate. Alluvial sediments and Litorina and other clay deposits are the typical sediments into

Table 27. K-means clusters, numbers of wells in each cluster and variables with high values in a certain cluster.

Cluster	Name	N	Variables with high values in a cluster
I	pH	152	pH, alkalinity, F, (Na)
II	Nitrate	404	NO_3, (K, Cl, SO_4)
III	Low in electrolytes	234	-
IV	Aluminium	229	Al, ($KMnO_4$ consumption)
V	Contaminated	30	NH_4, NO_2, K, Mn, Cl, EC, alkalinity, total hardness, colour, $KMnO_4$ consumption, NO_3, Na, SO_4, Fe,
VI	Rich in electrolytes	130	Na, Cl, EC, alkalinity, pH, total hardness, F, Mn, SO_4
VII	Turbid	206	Colour, turbidity, Fe, Al, Mn, NO_2, $KMnO_4$ consumption

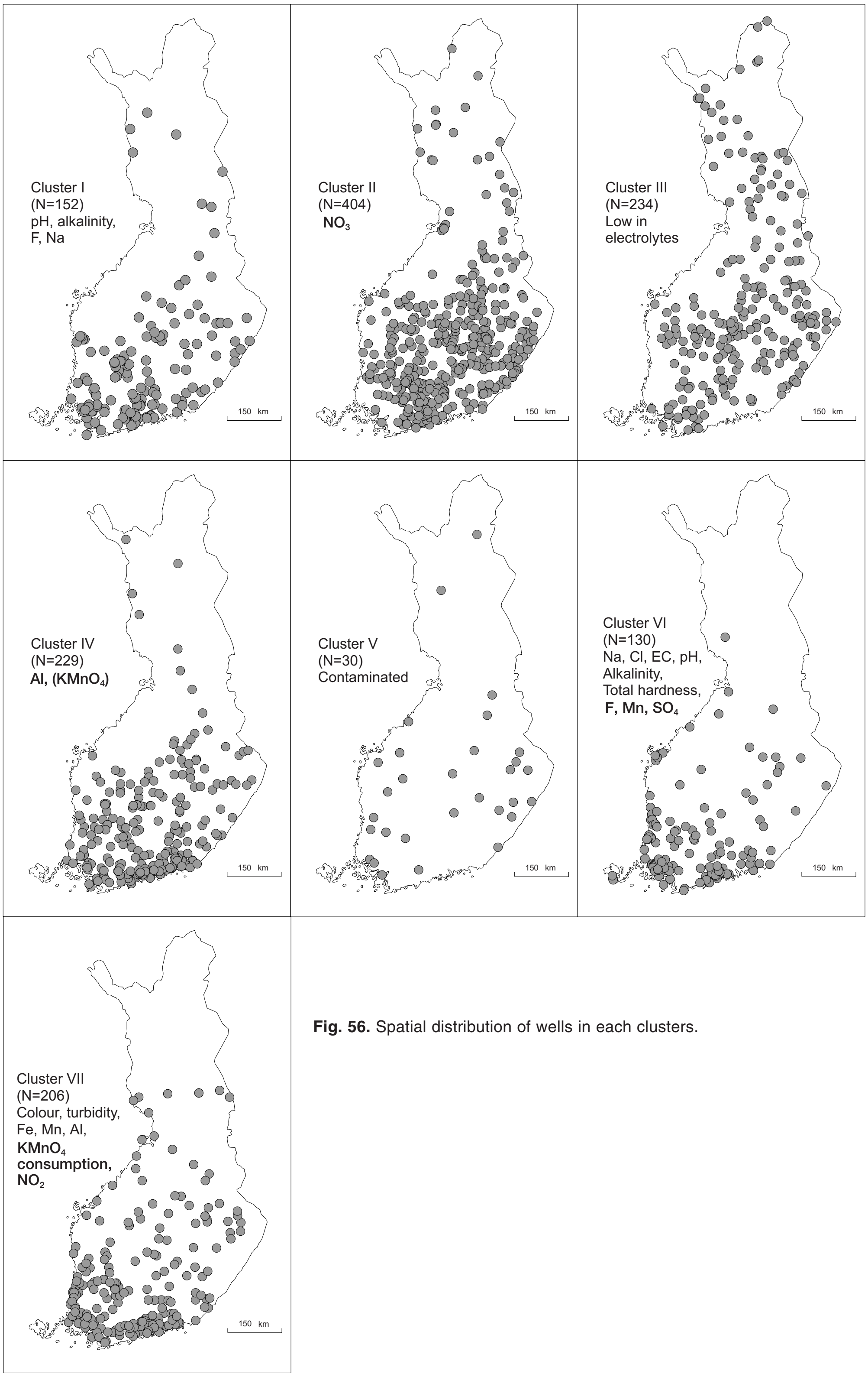


Fig. 56. Spatial distribution of wells in each clusters.

which the shallow wells have been dug. The most common bedrock type is S-type granite. Most of the wells are situated in the south-western part of Finland, which is covered by clay deposits (Fig. 56).

Cluster VII (turbid) includes wells in clay and till areas. Litorina and other clay deposits and cover till are main types of sediment represented. A large proportion of the wells in Litorina clay areas (41.6 %) belong to this cluster, and the predominant locations are on arable land or in yards. There are a lot of poorly constructed wells and ones which can be penetrated by surface water. The distribution of the wells in this cluster follows the coastline of southern Finland (Fig. 56).

The major disadvantage of this K-means cluster method is its inflexibility, i.e. it assigns each sample unambiguously to one cluster and is unable to deal with compositional overlap, a common phenomena in hydrogeochemical data of this kind. The classification can be used, however, to describe the wells in each cluster as either good ones (clusters I and III), moderately good ones (cluster VI) and ones that have quality problems (clusters II, IV, V and VII).

4.4 Water composition and cumulative contributing factors

4.4.1 The cumulative effects model

The present research has established a realistic and statistically valid representation of the quality of well water in Finland and has helped to identify factors which may affect its quality. These factors operate on various scales and affect individual wells differently according to the preponderance of one factor over another. Each factor also interrelates with all the others. The variables describing water quality in wells can nevertheless be related to their predominant scales, as indicated in Fig. 57.

4.4.2 Extra-regional scale

The database on the present wells includes measurements or observations indicative of whether the wells extend into the bedrock or into surficial sediments, the nature of the rock, the composition of atmospheric deposition, the distance from the present-day seashore, the Quaternary geological

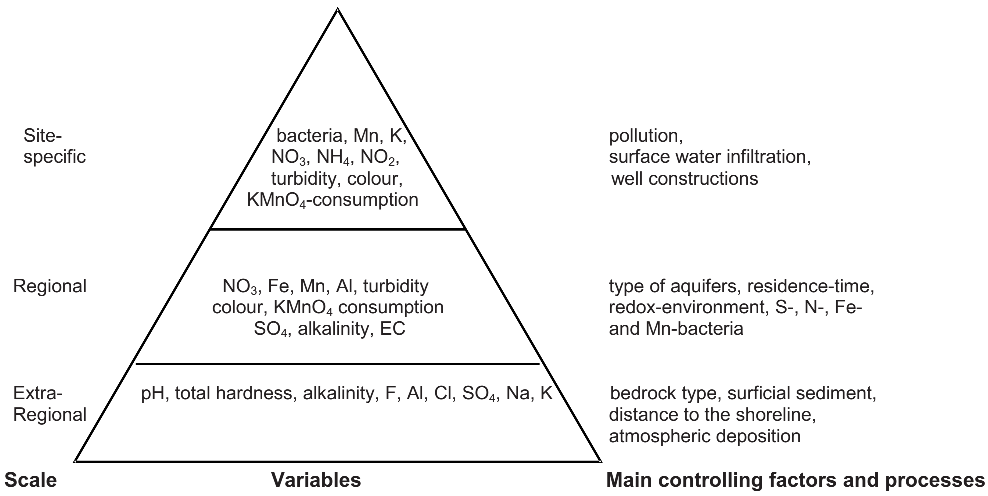


Fig. 57. Water quality variables and the main controlling factors and processes, described in the form of a cumulative effects model.

history of the well site, and other extra-regional determinants that may confer a 'natural' geochemical signature on the water. In order to assess the geological factors acting on the quality of the water, a number of specific chemical variables are taken as critical, notably F, Al, Cl, SO_4, Na, K, alkalinity, pH and total hardness.

One of the variables that most clearly represents the geological layer is fluoride, which is strongly connected with the geochemistry of the bedrock. This applies equally well to overburden wells related to rapakivi areas, as shown recently by Lahermo and Backman (2000). Aluminium is partly related to fluoride, because it is derived from Al-F complexes. The presence of fluoride is the main cause of Al mobilisation, as concluded by Hem (1992). On the other hand, especially in acid sulphate soils in western Finland, high aluminium concentrations are connected with the geochemistry of clay, as observed earlier by Palko (1994).

Sulphate concentrations in Finnish wells seem to be controlled by many extra-regional factors: sulphur deposition, primary sulphide minerals in the bedrock and till, and sulphides/sulphates in postglacial clay and silt deposits dating back to past marine phases of Baltic Sea. The main sources of sulphates in well water are sulphide minerals in the bedrock or unlithified sediments. There are high sulphate concentrations in the groundwater of black schist areas, and the spatial distribution of sulphate seems to follow the sulphur concentration in the fine fraction of glacial till. The presence of acid sulphate soils also explains the high sulphate concentrations in the coastal areas of Finland in particular. When the sulphate content is attributable to the geological environment, it is usually connected with high Ca, K, Mg, Mn, Na concentrations (Åström and Björklund 1995) or Fe concentrations (Loukola-Ruskeeniemi 1999). The fairly strong statistically significant correlations ($\rho > 0.5$, $p < 0.01$) between sulphate and total hardness (Ca, Mg), Cl, K and Na support the idea of a largely geological origin for the sulphate in well water. The correlation between sulphate and nitrate is weak, which confirms that less of the sulphate is of atmospheric origin. On the other hand, the sulphate concentrations in the well water were generally higher in 1990 than in 1958, which indicate the effect of acid rain, especially in the southernmost parts of Finland.

The chloride in well water seems to be predominantly of marine origin, either via salts in precipitation, leaching of fossil salts from the aquifer or intrusion of current sea water. Other possible minor sources may be anthropogenic, e.g. fertilisers and road salts, or lithological (Banks *et al.* 1998b, Lahermo *et al.* 1990).

Sodium also seems to be a good indicator of a marine effect. The most important exchange reactions in the unsaturated zone are the water-softening reactions, in which calcium and magnesium are exchanged for sorbed sodium as the groundwater moves through clayey material (Domenico and Schwartz 1998, Mather 1997, Toth 1999). This explains the rise in the sodium concentration of well water with increasing reaction time and also means that sodium is not as conservative an indicator of a marine effect as is chloride.

The pH and alkalinity of well water also seem to be controlled mainly by the groundwater retention time. This is reflected in the higher values found in drilled bedrock wells than in dug wells. As carbonate rocks are not well represented in the data, the effect of the bedrock chemistry could not be assessed. The effect of atmospheric acid deposition can be seen in a slight decrease in alkalinity values (between 1958 and 1990). However, Backman *et al.* (1999) and Soveri *et al.* (2001) discovered that the trend towards groundwater acidification altered in 1990 and pH and alkalinity have not longer been decreasing. In the light of these observations it might be assumed that the affect of acid deposition on the quality of well water will diminish.

4.4.3 Regional scale

Aquifer systems in Finland are much less broad in extent than the geological factors discussed above, and can be considered regional to local in scale. They are shown as the middle layer in the theoretical pyramid. In order for the data set to be discriminant at a hydraulic or regional level, the constraints were that it had to include observations or measurements aimed at determining the aquifer type, i.e. shallow or deep, in sand, gravel or in till or covered by clay. Consequently, the nature of the aquifer in which each well was located was determined or deduced. Analytical variables such as EC, alkalinity, turbidity, colour, $KMnO_4$ consumption, Fe, Mn, SO_4 and NO_3 played a major role in detecting the level of such effects.

The finer the soil, the longer the residence time, the more the soil processes alter the quality of the infiltration water and the greater the eventual impact on groundwater quality. This can be seen in EC, which increases with decreasing grain size of the soil in the aquifers. Small clay mineral particles and colloids increase the colour and turbidity of well water, and the chemical composition of clay minerals can also be seen reflected in groundwater quality in the form of high Al, Fe and Mn concentrations in clay-dominated areas. High $KMnO_4$ values in clay-covered aquifers reflect the high organic content of clays.

The occurrence of iron and manganese in groundwater is principally affected by the oxygen balance, which depends on the geological structure and flow patterns of the aquifer. The groundwater in clay-covered aquifers and in some bedrock fissures and fractures has been found to be deficient in oxygen, mainly due to the long retention time of the water. The decomposition of organic matter and the oxidation of iron, manganese and nitrogen compounds consume oxygen, so that the groundwater becomes reducing with respect to these compounds (Lahermo 1970, Hatva 1989, Hem 1992, Backman *et al.* 1999). As Litorina clay, a factor operating on an extra-regional scale, contains large amounts of organic matter, it affects the oxygen demand and thus also the state of oxidation of the aquifer, and thereby the iron and manganese concentrations in well water.

The relative concentrations of nitrogen compounds depend on the degree of nitrification and state of oxidation. The high NO_3 /NH_4 ratios and lack of nitrate in clay-covered aquifers and bedrock aquifers indicate lower redox potentials than in other shallow aquifers. Clay soils also absorb nitrogen (Seppänen 1987) and prevent nitrate-rich water from entering the aquifer. This partly explains the low nitrate concentrations in clay-covered aquifers relative to coarse-grained, more permeable aquifers.

In addition to the state of oxidation, microbiota may be seen to affect iron, manganese, nitrogen and sulphur compounds. The total numbers of microbes in soil and groundwater samples depend on the amount of nutrients, pH, temperature, redox state etc. (Hem 1992, Heinonen-Tanski 1987, Seppänen 1987). The effect of microbial activity on sulphate concentrations in water is difficult to estimate, but sulphide oxidation is one of the most important acid and sulphate-producing reactions in geological systems (Domenico and Schwartz 1998, Toth 1999). Reducing conditions in clay-covered aquifers, especially in areas with acid sulphide soils, can partly prevent the sulphate concentration in the groundwater from increasing any further, as sulphides can no longer be oxidised.

4.4.4 Site-specific scale

As noted, even in a similar geological context and with comparable hydraulic or aquifer-dependent effects, water may still differ in quality from one well to another. This is mainly due to local or highly site-specific factors such as the overall maintenance of the well, its depth and age, present land use near the well and pollution derived from animal stables, septic tanks etc. These factors, where they apply and exercise an extensive influence on water quality, will mask the natural variation caused by the regional and extra-regional factors. Intermediate between regional and site-specific factors are 'land-use' effects such as agricultural practices or local industrial influence. These are included here among the site-specific factors, although they could just as well be placed in the middle layer of the pyramid. In order to detect such effects, variables such as nitrogen-compounds and coliform bacteria were selected for discussion here.

The redox environment affects primarily the state of nitrogen oxidation, but the origin of the nitrogen is anthropogenic and predominantly site-specific. The effect of atmospheric deposition is marginal. There are no lithological sources of nitrogen compounds (Rönkä 1983, Hem 1992), and a large part of the nitrogen in groundwater seems to be of agricultural origin. The main sources of nitrogen in well water are waste from animal husbandry, agricultural practices such as fertilisation and human settlement. Nitrate has a moderate correlation with potassium in highly permeable aquifers ($\rho = 0.55$, $p < 0.01$), pointing to potassium nitrate fertilisers as its origin. The nitrite and ammonium in well water are suggestive of fresh contamination.

The effects of human activity on well water quality can be seen in a higher ion content. Thus the concentrations of almost all ions were lower in the wells situated in forests, far away from human activity, than in the other wells. The microbiological quality of the water in the forest wells was also somewhat better than elsewhere.

The existence of microbiological indicator bacteria in well water reveals either intrusion of surface water into the wells or faecal pollution originating from humans, domestic animals or wild animals. There is a greater risk of bacterially unsafe water in the case of dug wells than in drilled wells, in wells situated near points of waste water infiltration and in old wells as compared with young ones. Pollution sources (cowsheds, piggeries, outdoor toilets or arable land) near a well will increase the risk of bacterial pollution.

In addition to being a redox-sensitive variable, manganese seems to be connected with site-specific contamination. This may be related to microbial activity in surface and waste water.

5 Conclusions

5.1 Major findings and conclusions

1) Most of the domestic wells (80 %) were dug wells or captured springs, and the rest were wells drilled into the bedrock. Average age of the wells in 1990 was 27 years. The oldest were stone-lined wells, their average age being 52 years, while that of the concrete ring wells was 27 years, and that of the wells drilled into bedrock about 15 years. Most of the wells (72 %) were shallow, extending to depths of less than 8 metres, while the average depth of the dug wells was 5.8 m.
2) Mapping of the geological settings of the Finnish wells showed that the most common type of surface sediment around them was till. Only 15 % of the wells had been sunk into sandy or gravel deposits, and in almost as many cases the aquifer was covered by clay. Half of the dug wells were in cover moraine, 5 % in drumlins or other morainic landforms and only 34 in ice-marginal end moraines. About 14 % of the dug wells passed through confining clay layers, which in a half of the cases were Litorina clays. About 6 % of the wells were located in glaciofluvial gravel or sand deposits and a few percent more represented littoral deposits. Only a few wells (24) were in alluvial or other undefined sand deposits. Most of the wells were situated in areas dominated by either gneiss/schist or TTG-series rocks, one third in granitic bedrock areas and only 5 % in areas composed of intermediate or felsic volcanics. There were only 15 wells in claystone or sandstone areas.
3) Only one third of the wells (37.2 %) fulfilled all the hygienic and technical requirements and recommendations for drinking water. Over 60 % of the wells had one or more water quality problems and there was a negative correlation between water quality and the age of the well. Household water fulfilling all health-related requirements was in use in 63 % of the households. Nitrate was the most common individual water quality defect, explaining one quarter of the health-related quality problems. The nitrate problem was most serious in the wells dug into coarse-grained, highly permeable aquifers. Total coliform and faecal coliform bacterial counts were too high in almost as many cases as nitrate, but high number of total coliforms and faecal coliforms were detected together in only 8 % of the wells, which confirms the finding that wells having multiple water quality problems are rare. High fluoride concentrations were twice as serious health problem in bedrock wells than in dug wells. Also, in 45 % of the wells the water was not technically or aesthetically satisfactory, the main problems being high iron concentrations and high colour and turbidity values. Dug wells and springs entailed more problems in meeting the technical-aesthetic target values for all variables other than manganese, chloride and sodium concentrations. Wells in clay-covered areas were clearly the most problematic as far as the technical-aesthetic quality objectives were concerned. The wells in coarse-grained deposits were better, although aluminium concentrations were too high in 13.5 % of these.
4) The water quality variables were divided here into four main types according to their statistical distribution. Type I comprised normally or log-normally distributed variables (pH, EC, alkalinity, total hardness, turbidity, Cl, Fe, SO_4, and K), type II variables characterised by a skewed distribution and high kurtosis (colour, $KMnO_4$ consumption, NO_3, Al and Na), type III variables with many analyses below the detection limits (Mn and F) and type IV indicator variables (NO_2, NH_4 and bacterial counts). Since normal distributions could not be obtained for the variables, non-parametric and robust statistical methods were preferred and log-transformed data were used throughout the statistical treatment when using robust parametric tests such as Principal Component Analysis.

5) Differences in the chemical composition of the groundwater were observed between the bedrock wells and dug wells, pH, conductivity and hardness usually being higher in the bedrock groundwater than in the water from Quaternary aquifers, indicating more mineralised water with longer residence times. The water in bedrock wells was also enriched in F and Na. In the case of hardness and Mn, Cl and SO_4 concentrations similar distributions were noted for these two main well types, possibly on account of the lithological sources and the presence of recent and relict marine salts in the bedrock waters, and also as a result of anthropogenic contamination of wells in shallow dug wells. The variables Fe, K, turbidity, colour, $KMnO_4$ consumption and Al exhibited higher values in the water from Quaternary aquifers than in that from bedrock aquifers. Most of these variables indicate the influence of surface water rich in organic matter and/or suspended clay particles. Bacterial counts and the concentrations of nitrogen compounds were generally higher in the water from dug wells than in the bedrock wells, indicating the high vulnerability of shallow groundwater.
6) Correlation analysis and principal component analysis led to the differentiation of five factors contributing to the conclusions reached in the survey. These correspond to a combination of specific geological, hydrogeological and local factors which together are manifested as cumulative effects operating at particular locations. The strongest modifier of well water quality seems to be an extra-regional or regional factor termed 'salinity' (EC, total hardness, Cl, SO_4, K and Na), which appears to represent a mainly extra-regional, 'geological' factor connected with the residence time of the water in geological material. Some of the elements involved in this factor can be of atmospheric, marine or anthropogenic origin. The second factor, partly regional and partly site-specific, explains the effect of dissolved organic matter and the 'redox environment' (turbidity, colour, $KMnO_4$ consumption, Al, Fe and Mn), and a third is the extra-regional 'pH-factor' (pH, alkalinity and F) reflecting buffering capacity. The site-specific 'pollution factor' (total coliform bacteria, faecal coliform bacteria and faecal streptococci) and 'contamination factor' (NO_2, NH_4 and Mn) seems to be of minor significance.
7) The microbiological indicators (total coliform bacteria, faecal coliform bacteria and faecal streptococci) were relatively independent of the water-soluble components. Total coliforms had significant, although weak positive correlations with faecal coliforms, faecal streptococci, $KMnO_4$ consumption, aluminium and colour, and faecal coliforms and faecal streptococci were also weakly correlated with $KMnO_4$ consumption and aluminium. These correlations suggest that the occurrence of bacteria is connected with the leaching of organic matter into the wells. Total coliform bacteria were detected in over 60 % and faecal coliform bacteria in 10 % of the wells. The proportion of *E. coli* among the faecal coliforms was as high as 82 %, confirming the usefulness of this category for indicating faecal contamination in Finnish wells. Faecal streptococci were detected in one third of the wells, but the proportion of confirmed faecal streptococci in their count was only 20 %, which means that the bacteria found in the sample may have been partly of plant or insect origin. The existence of bacteria is more to be expected in dug wells than in bedrock wells, and particularly in wells which are near waste water infiltration point. An increased risk of bacteria in the water showed a positive correlation with the age of the well, and dug wells or springs having thermotolerant coliforms were more often situated less than 100 m from an outdoor toilet, arable land or a cowshed. The present results show that the hygienic quality of well water cannot be assessed without the detection of indicator bacteria.
8) K-means clustering analysis led to a grouping of the water samples into seven homogeneous clusters, each characterized by a dominant set of quality indicators. These clusters were designated by the following names: pH, Nitrate, Low in electrolytes, Aluminium, Contaminated, Rich in electrolytes and Turbid. The method appeared to be capable of providing an identification system for achieving an understanding of the reasons for quality problems in individual wells.
9) The magnitude of the seasonal changes in well water quality appears to be highly dependant on site-specific factors such as the construction and depth of the well, as the natural variability related to climatic factors such as precipitation, evaporation and meltwater from snow was

masked by anthropogenic influences and surface water infiltration. As many as 27 % of the well users had observed seasonal changes in water quality, most of these (17 % of all users) reporting that the water was of poorer quality in spring than at other seasons. The water analyses confirmed this. The minimum values for alkalinity, electrical conductivity and hardness in spring indicate dilution of the well water by large amounts of infiltrating water from melting snow, ground frost and the rain. The occurrence of the highest values for $KMnO_4$ consumption in spring reflect surface water infiltration through the casing. The fact that the fluoride content was generally highest in summer indicates that effective dissolving of fluoride happens in spring, driven by the low pH values of the precipitation and meltwater. There are less total coliform bacteria in spring, however, because the sources of contamination are not active. The highest average count of faecal coliforms was detected in summer, maybe because of cattle grazing.

10) Long-term changes are reflected in a lowering of alkalinity, and to a lesser degree of pH, a reduction in iron content and in an increase in chloride and sulphate concentrations and $KMnO_4$ consumption. The decrease in alkalinity and increase in sulphate and electrical conductivity refer to extra-regional acidification processes, while the notable decreases in iron and chloride concentrations and in $KMnO_4$ consumption are to greater or lesser extents indications of changes in site-specific factors such as the depth or construction of the wells. Such variations are not so marked on a national scale as had been assumed in the light of a preliminary investigation into well water acidification, although they fluctuate in magnitude from one part of the country to another.

11) The values obtained for pH, electrical conductivity, alkalinity and hardness and the concentrations of most ions (Al, Na, SO_4, F, Mn and Fe) were usually highest in the fine-grained sedimentary aquifers, due to the fact that the solubility of minerals always increases with decreasing hydraulic conductivity, i.e. longer reaction time. The fact that the concentrations were lowest in small lee-side sand and gravel deposits reflects the short reaction time of the groundwater. Concentrations were also generally low in glaciofluvial deposits and ice-marginal end moraines, except in the case of nitrate content, which was highest in these coarse-grained formations and the lowest in the clay-covered aquifers.

12) The oxidation-reduction environment in the aquifer has a major effect on well water quality. The state of oxidation of nitrogen compounds and the occurrence of iron and manganese in groundwater is principally affected by the oxygen balance, which is controlled by the geological structure and flow patterns of the aquifer. The groundwater in clay-covered aquifers and in deep-seated bedrock fissures and fractures is frequently reducing with respect to iron, manganese and nitrogen. The relative concentrations of nitrogen compounds (NO_3 /NH_4 ratios) indicate differences in the redox environment in which separate the clay-covered aquifers and bedrock aquifers from the other shallow aquifers. The nitrate deficiency observed in reducing aquifers indicates the reduction of nitrate to nitrite and ammonium. The environment can also be unsuitable for nitrification due to a lack of oxygen and specific bacteria while nitrogen is still in the form of ammonium or nitrite. Manganese ions are more stable in resisting oxidation than those of ferrous iron, a situation which is seen here in the occurrence of Mn in solution even in well water characterised by only a moderate oxygen content.

13) Some of the wells were properly constructed, in good condition and free of surface water infiltration. If also the redox state in an aquifer is good, the water quality in the associated wells will be affected mainly by extra-regional factors, the importance of which can vary spatially, as seen in the values for chemical variables such as F, Al, SO_4 ,Cl, Na, K, Mn, Fe, alkalinity, pH and total hardness. There are wells in clay-covered areas in the coastal areas of Finland in particular where regional factors, notably the redox condition of the aquifer, have a predominant effect on water quality. Analytical variables such as turbidity, colour, $KMnO_4$ consumption, Fe and Mn, sulphate and nitrate, will then play the main role in detecting the level of that effect. There were many wells, however, where a site-specific factor such as surface water penetration because of poor construction or a nearby pollution source was so essential that it masked all the other water quality indicators.

14) All the layers of factors are represented to various extents in each well. Well owners can modify the site-specific factors, of course, and ameliorate their effects by keeping their wells in a good state of repair, addressing problems of insufficient aeration and eliminating possible sources of pollution nearby. Such measures can affect the microbiological quality of the well water, the amounts of nitrogen compounds contained in it, its turbidity, $KMnO_4$ consumption and in part its aluminium, Fe and Mn concentrations, but it is impossible to influence the state of oxidation in the aquifer or any of the extra-regional factors such as the chemical composition of the soil and bedrock.
15) Well water is vulnerable to site-specific factors such as surface water infiltration into the well, and the variation in quality variables in individual wells is likely to be greater than the seasonal variation in the aquifers. This means that a sample from a well is not especially representative of the water quality in the aquifer, but merely denotes the quality of the well itself.

5.2 Recommendations

There is an increasing risk of poor quality drinking water in the case of wells situated on arable land or in farmyards in the vicinity of a cowshed, piggery, outdoor toilet or road. If the well is old and its structure is leaking, the possibility of poor quality water will increase considerably. It is therefore strongly recommended that well owners should take care of the condition of their wells in order to avoid infiltration by surface water, and that new wells should be sunk as far from sites of human activity as possible. The quality of the water obtained from old wells in particular should be analysed regularly.

Wells in highly permeable aquifers (sand and gravel deposits) should be carefully protected from the influence of fertilisers and agricultural and human waste water in order to avoid contamination with nitrogen compounds and bacteria.

The results of water quality analyses should be considered as a whole in order to form a complete picture of the contributing factors. A single analysis of one variable does not necessarily reveal much about the general quality of the water in the well. At the same time, additional knowledge about the construction of the well should be used when assessing water quality problems and the reasons for them. All the basic anions (HCO_3, Cl, SO_4, NO_3) and cations (Ca, Mg, Na, K) should be included in list of the analyses in order to make geochemical observations and interpretations possible.

Seasonal variability can be observed in the quality of well water, particularly with respect to bacterial counts and certain indicators of surface water infiltration. Especially in shallow wells (< 5 m) of poor construction, water samples should also be taken in spring in order to form a worst-case picture of water quality, and not only in winter, when the water quality is probably at its best.

Yhteenveto

Suomen haja-asutuksen vesihuolto nojautuu pääasiassa yksittäisten talouksien käytössä oleviin pienkaivoihin. Vedenhankinnan valtakunnallisessa pitkän tähtäimen suunnitelmassa hyväksytyt tavoitteet ovat taata koko väestölle hyvänlaatuisen talousveden saatavuus kohtuullisin kustannuksin ja torjua vedenottopaikkoihin kohdistuvat ympäristöriskit. Tämä vaatii pienkaivojen osalta riittävästi tietoa niistä tekijöistä, jotka kunkin kaivon osalta vaikuttavat sen veden laatuun. Tässä tutkimuksessa tarkasteltiin Suomen haja-asutusalueen pienkaivojen veden laatua ja siihen vaikuttavia tekijöitä hyödyntäen 1421 kaivon tutkimusaineistoa. Kaivot on valittu siten, että ne painottuvat käytön mukaan edustaen mahdollisimman hyvin kaikkia Suomessa jatkuvassa talousvesikäytössä olevia pienkaivoja.

Suurin osa tutkimuksessa mukana olleista kaivoista on maaperään kaivettuja matalia betonirengaskaivoja, joiden keskisyvyys on 5.8 m tai lähdekaivoja, joiden keskisyvyys on vain 1.6 m ja loput kallioporakaivoja. Kaivojen keskimääräinen ikä oli tutkimusvuonna 27 vuotta. Kaivoista tutkittiin yleiset vedenlaatutekijät ja tarkasteltiin vuodenaikojen, kaivon ympäristön geologisten olosuhteiden, kaivotyypin, kaivon syvyyden, kaivorakenteiden, kaivon kunnon, meren läheisyyden sekä mahdollisten likaajien vaikutusta veden laatuun käyttäen pääasiassa epäparametrisia tilastollisia analyysimenetelmiä kuten klusterianalyysiä ja pääkomponenttianalyysiä.

Tutkittujen kaivojen vedestä vain 37.2 % täytti kaikki talousvedelle asetetut laatuvaatimukset ja tavoitteet. Moniongelmaisia kaivoja, joiden vesi toisin sanoen ei täyttänyt niin terveydellisin kuin teknis-esteettisinkään perustein määrättyjä raja-arvoja oli 276. Kaivoja, joiden vedessä oli ainoastaan terveydellisiä laatuongelmia oli 251, ja kaivoja, joiden veden laadussa ainoastaan teknis-esteettiset laatutavoitteet eivät täyttyneet oli 366. Kaivovesi oli heikkolaatuisinta rannikolla ja parasta Pohjois-Suomessa. Raja-arvot ylittävä nitraattipitoisuus, koliformisten bakteerien kokonaismäärä ja lämpökestoisten koliformisten bakteerien esiintyminen sekä korkea fluoridipitoisuus olivat yleisimmät kaivoveden terveydelliset laatuepäkohdat. Teknis-esteettisiä laatuongelmia oli 45 % tutkimuskaivoista. Ongelmia oli tavallisimmin väriluvun, rautapitoisuuden, sameuden, alumiinipitoisuuden, mangaanipitoisuuden, kaliumpermanganaatin kulutuksen tai pH:n suhteen.

Maaperään kaivettujen kaivojen ja toisaalta kallioon porattujen kaivojen veden laatua verrattiin keskenään. Kallioporakaivojen vesissä on yleensä enemmän liuenneita aineita kuin niitä matalampien maakaivojen vesissä johtuen veden pidemmästä viipymästä. Veden pH, sähkönjohtavuus ja alkaliniteetti sekä fluori-, natrium-, mangaani- ja kloridipitoisuudet ovat keskimäärin korkeampia kallioporakaivoissa kuin maakaivoissa. Kaivoveden sameus, väriluku, $KMnO_4$-luku, sekä alumiini-, rauta ja kaliumpitoisuus puolestaan on keskimäärin suurempi maaperään kaivetuissa kaivoissa kuin kallioporakaivoissa johtuen osittain orgaanista ainesta ja savipartikkeleita sisältävän pintaveden pääsystä mataliin kaivoihin. Myös typpiyhdisteitä ja indikaattoribakteereita esiintyy maakaivojen vedessä yleisesti osoittaen niiden likaantumisherkkyyttä. Kun verrataan kaivoveden laatua sosiaali- ja terveysministeriön asettamiin yksittäisten kaivojen talousvesiä koskeviin laatuvaatimuksiin havaittiin, että porakaivojen veden terveydellinen laatu on yleisesti ottaen parempi kuin maakaivojen lukuun ottamatta fluoridipitoisuutta. Myös teknis-esteettiset laatutavoitteet täyttyvät useammin kallioporakaivoissa kuin maakaivoissa lukuun ottamatta mangaani-, kloridi- ja natriumpitoisuuksia.

Kaivon geologisen ympäristön vaikutusta tarkasteltiin ryhmittelemällä kaivot ympäristön vallitsevan pintamaalajitteen, muodostumatyypin ja kallioperän kivilajin perusteella. Kaivoista 77 % on maakaivoja ja vastaavasti noin 23 % kallioporakaivoja. Maakaivoista puolet on kaivettu pohjamoreeniin ja vain 6 % on varsinaisissa moreenimuodostumissa kuten drumliineissa ja lisäksi pieni osa sijoittuu reunamuodostumiin. Vain n. 18 % maakaivoista sijaitsee karkearakeisissa jäätikköjokikerrostumissa tai rantakerrostumissa ja saman verran on kaivettu savipeitteisiin pohjavesiesiintymiin. Sora- ja hiekkakerrostumissa olevissa kaivoissa on veteen liuenneiden aineiden kokonaismäärä pienin kasvaen seuraavassa järjestyksessä: jäätikköjokikerrostumat < rantakerrostumat < reunamuodostumat < pohjamoreeni < moreenimuodostumat < jokikerrostumat < savipeitteiset kerrostumat. Savipeitteisten alueiden kaivovedet ovat usein sameita, värillisiä, rauta-, mangaani- ja alumiinipitoisia. Osoittautui kuitenkin, että ympäristön savipeite suojaa kaivoa nitraattikuormitukselta ja kohonneet nitraattipitoisuudet ovatkin yleisesti karkeiden sora- ja hiekka-alueiden kaivovesien ongelma. Kallioperän litologia vaikuttaa sekä kallioporakaivojen että maakaivojen veden koostumukseen. Selvimmin erottuvat omana ryhmänään rapakivialueen kaivot, joissa fluoridi- ja alumiinipitoisuudet ovat korkeammat kuin muissa kaivoissa. Myös sulfidimineraaleja sisältävä kallioperä heijastuu kaivoveden koostumukseen. Mustaliuskealueella olevien kaivojen veden pH ja alkaliniteetti on keskimääräistä alhaisempi ja niissä on runsaasti liuenneita aineista kuten rautaa ja sulfaattia.

Meren läheisyys näkyy analyysituloksissa kaivotyypistä riippumatta veden SO_4-, Cl- ja Na-pitoisuuksissa ja sähkönjohtavuudessa, jotka kaikki ovat keskimäärin sitä suurempia, mitä lähempänä rannikkoa kaivo sijaitsee. Rannikkoalueen eniten liuenneita aineita sisältävät vedet sijoittuvat pääosin Litorina-savikon peittämälle alueelle. Näillä alueilla kaivovedet ovat muita, myös muita savialueita, värillisempiä, sameampia ja sisältävät enemmän orgaanista ainesta, sulfaatteja, klorideja, fluoridia, natriumia, kaliumia, rautaa ja alumiinia.

Mikrobiologisten indikaattoribakteerien esiintymistä kaivovedessä tarkasteltiin erikseen, koska ne ovat suhteellisen riippumattomia muista vedenlaatumuuttujista. Koliformisten bakteerien kokonaismäärä korreloi kuitenkin heikosti kaliumpermanganaatin kulutuksen, alumiinipitoisuuden ja värin kanssa, mikä kertoo yhteydestä orgaanista ainesta sisältävän pintaveden pääsyyn kaivoon. Koliformisia bakteereita havaittiin 60 % tutkimuskaivoista ja lämpökestoisia koliformisia bakteereita 10 % kaivoista. Lämpökestoiset koliformiset bakteerit varmistuivat 82 %:sesti ulosteperäistä saastumista osoittaviksi *Escherichia coli* -bakteereiksi. Fekaalisia streptokokkeja havaittiin kolmanneksessa tutkimuskaivoista, mutta vain vajaat 20 % näistä varmistui ulosteperäistä saastumista osoittaviksi enterokokeiksi. Vanhoissa kaivoissa, matalissa rengaskaivoissa ja kaivoissa, jotka sijaitsevat lähellä ulkokäymälää, peltoa, navettaa tai jäteveden imeytyspaikkaa on kohonnut riski bakteerien esiintymiselle.

Kaivoveden vuodenaikaisvaihtelua pyrittiin arvioimaan vertaamalla kolmen eri vuodenaikana suoritetun näyttenoton tuloksia. Luontainen pohjaveden laadun vuodenaikaisvaihtelu on niin pientä, että se peittyy kaivovesien kyseessä ollen muiden vaikuttavien tekijöiden, ennen kaikkea pintavesivaikutuksen, alle. Kaivon käyttäjistä 27,4 % ilmoitti havainneensa eroa veden laadussa eri ajankohtina. Kaivon käyttäjistä 17 % ilmoitti veden olevan keväisin muuta ajankohtaa huonompaa, mikä näkyy jonkin verran myös analyysituloksissa. Kaivovedessä esiintyi kuitenkin keskimäärin eniten ulosteperäistä saastutusta osoittavia bakteereita kesällä johtuen todennäköisesti karjan laiduntamisesta kaivon läheisyydessä.

Vertaamalla tämän tutkimuksen kaivojen veden laatua 1950-luvun lopulla tehdyn talousvesitutkimuksen (Wäre 1960) tuloksiin voitiin havaita tiettyjä kaivovesien laadun pitkäaikaismuutoksia: alkaliniteetti, pH ja rautapitoisuus ovat nykyään keskimäärin hieman alhaisempia, sähkönjohtavuus, kloridi- ja sulfaattipitoisuus puolestaan hieman korkeampia kuin 1950-luvulla.

Johtopäätöksenä voidaan todeta, että kaivoveden laatuun vaikuttavat useat tekijät, joita voidaan tarkastella eri mittakaavassa. Laatutekijät voidaan jakaa karkeasti laaja-alaisiin geologisiin tekijöihin, alueellisiin pohjavesiesiintymäkohtaisiin tekijöihin ja paikallisiin, kaivokohtaisiin tekijöihin. Perusta kaivoveden laadulle on se geologinen ympäristö, jossa kaivo sijaitsee. Osa kaivoveden laatuominaisuuksista liittyy selkeästi kaivon geologiseen ympäristöön (mm. pH, alkaliniteetti, kovuus, F, Al, SO_4, Cl, Na, K, Mn ja Fe pitoisuudet), osa laatuominaisuuksista on riippuvainen pohjavesiesiintymän hapetus-pelkistysoloista (sameus, väriluku, Fe, Mn, SO_4, typpiyhdisteet) ja osa laatuepäkohdista johtuu pintavesien pääsystä kaivoon tai likaavasta toiminnosta kaivon lähistöllä (mikrobiologinen laatu, typpiyhdisteet, sameus, kaliumpermanganaatin kulutus, Fe, Mn, Al). Kaivovedet ovat herkkiä paikallisille vedenlaatuvaihteluille, eivätkä pienkaivot ole siten parhaita näytteenottopaikkoja kuvaamaan pohjaveden laatua akvifereissa. Kaivon omistaja voi vaikuttaa suuresti paikallisiin tekijöihin huolehtimalla kaivorakenteiden kunnosta, estämällä pintavesien pääsy kaivoon, huolehtimalla kaivon ilmanvaihdosta ja siitä, että kaivo sijaitsee mahdollisimman kaukana sitä likaavista tekijöistä kuten karjasuojista, käymälöistä, jäteveden imeytyspaikoista, pellosta tai tiestä.

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I dedicate this thesis to my family.

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Maps and map serials used in the study

(GSF = Geological Survey of Finland, NLS = National Land Survey of Finland)
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Journal articles

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Korbut V.V. [Корбут В.В.] 1989. Nest building of hooded crows. 1. Utilisation of different substrata. *Zool. Zh*. 68: 88--95. [In Russian with English summary].

Book chapters

Tamminen P. & Starr M. 1990. A survey of forest soil properties related to soil acidification in Southern Finland. In: Kauppi P., Anttila P. & Kenttämies K. (eds.), *Acidification in Finland,* Springer, Berlin, pp. 237–251.

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