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Iron and artificial recharge of groundwater

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High iron concentrations may be a problem when groundwater is used for drinking water supply. The objective of this field study has been to clarify the factors that control the presence of iron in the water phase at artificial groundwater recharge with pond infiltration. The iron concentration in water was monitored over a 7-month period at a site with pond infiltration. Water was sampled once a month from 10 to 115 cm depth in the filtersand of the pond, and analyzed for iron. The filtersand was sampled before and after the water sampling period so that accumulation of iron could be quantified. It was found that at the 20 cm level in the filtersand the iron concentration was reduced to a level fully acceptable in drinking water. The analysis of the soil showed that 86% of the accumulated iron was retained in the top 5 cm of the filtersand. According to a budget considering the amount of iron coming to the filtersand and the amount leaving it, 138 g m^{-2} should have been retained during 15 months while 145 g m^{-2} was actually found by analysis. The findings show that the iron retention process was fast, both in time and travel distance. The iron retention efficiency would thus not be improved by a longer residence time. This suggests that even smaller geologic formations may be used for pond infiltration. The results also indicate that the iron retention was a largely abiotic process. The problem with gradually increasing iron concentrations at induced recharge is discussed in view of studies of iron-reducing microorganisms.

Introduction

In the production of drinking water the iron content must be kept low both for aesthetic reasons and for the sake of technical installations in the waterworks and distribution system. When the drinking water source is natural groundwater or groundwater supplemented by artificial pond infiltration or by induced infiltration, iron may be a problem. The groundwater often contains more iron than is aesthetically desirable and is stated in governmental regulations. Swedish regulations allow up to 0.1 mg Fe l^{-1}, but an iron concentration below 0.05 mg l^{-1} is recommended for public waters (Statens Livsmedelsverk 1993). In Finland up to 0.2 mg l^{-1} is allowed in drinking water (Social- och hälsovårdsministeriet 1994). EU states a highest acceptable level at 0.2 mg l^{-1} and sets a guide level at 0.05 mg l^{-1} (Council Directive 1980).

Different methods are applied in groundwaterworks to reduce the amount of iron in the water. The methods are generally based on oxidation of ferrous iron, precipitation and subsequent adsorption of positively charged iron (hydr)oxides onto some kind of adsorbent, e.g. sandfilters. In most cases these methods efficiently remove the iron from the water. In some cases, however, removal seems impossible and may lead to abandoning of the well. This is especially the case when the iron is present in the form of negatively charged iron-organic complexes. The measures that can be taken in these situations are dependent on the ambient conditions in combination with economy and water scarcity/availability. Often there is a problem choosing an appropriate measure because the reason for the elevated iron concentrations has not been defined. Hence, an investigation of the ambient conditions must be a first measure.

It is desirable to simplify the water handling in the waterworks for economical and practical reasons. Thus, when designing waterworks with pond infiltration, the infiltration site, the infiltration conditions and the well sites should be carefully chosen to yield a groundwater containing a minimum of iron. But even if the initial iron content in the infiltration water is retained at an early stage in the infiltration process, the iron concentration may again increase in the aquifer due to a number of factors. Such factors are low pH and redox conditions, low oxygen content, the presence of organic material, the lack of surfaces for precipitation, and influence from iron bacteria. Furthermore, an important explanation for higher iron concentrations in the groundwater than in the infiltration water is the intermixture of natural, iron-rich groundwater.

In wells that are based on induced recharge through the bottom sediments of a lake or river, a gradual increase of iron concentration is an often encountered problem. Months or even years after a new well has been brought into operation, the iron concentration may start increasing towards concentrations that make the water unfit as a drinking water supply. This kind of development is usually not predictable and an improved understanding of these processes would be valuable. If the poor groundwater quality could be improved by some kind of *in-situ* treatment, many latent groundwater resources could be turned into productive groundwater resources.

The main objective of this study is to clarify the factors that control the presence of iron in the water phase during the initial steps of artificial recharge of groundwater for drinking water production — to study the processes taking place in the infiltration zone. More specifically, this means studying the influence of microbial activity, of iron-organic complexes, of an unsaturated zone at basin recharge and of the contact area and residence time on the resulting groundwater quality. An important question is whether the residence times used today could be shortened while maintaining water quality. If so, it would be possible to use even smaller geologic formations for pond infiltration — an increasing demand as many eskers are already utilized or are being excavated for aggregate. The study is part of a larger project which deals with the retention of iron and other processes affecting the water quality in artificial recharge of groundwater (Frycklund *et al.* 1994).

A preliminary field study in Emmaboda in southern Sweden suggested that a large part of the initial iron content was retained in the very top of the filtersand of an infiltration pond (Frycklund and Jacks 1993), supporting the idea that a longer residence time does not improve the iron retention efficiency.

Methods

The concentration of iron in the water and in the filtersand was monitored at a public waterworks where groundwater was artificially recharged through infiltration ponds.

Site description

Järna waterworks, 40 km southwest of Stockholm, was chosen for the study. Samples of water and soil were taken from a 670 m^2 large infiltration pond used for public drinking water production. The pond, which was one of three infiltration ponds in the area, had been excavated during the 1970s in a small glaciofluvial deposit, an esker, which was 13–23 m deep and here exposed along a stretch of 400 m (SGU 1968). A layer of 60–70 cm pre-washed filtersand with a texture of gravelly sand ($d_4 = 0.2$ mm, $d_{10} = 0.3$ mm, $d_{60} = 1.1$ mm, $d_{80} =$ 2 mm, indicating the grain diameter corresponding to the 4, 10, 60 and 80% levels respec-

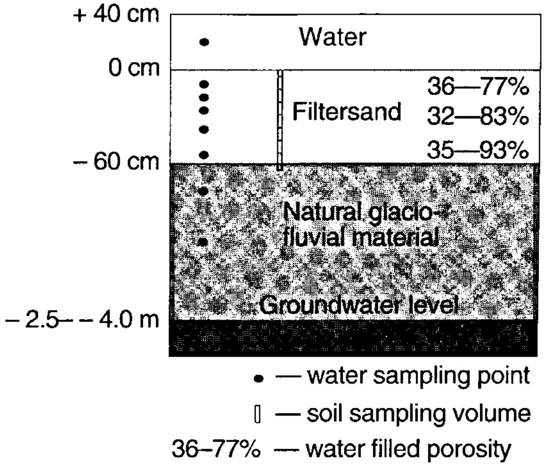

Fig. 1. Water samples were collected from lysimeters at 10–115 cm depth in the filtersand and from the infiltration water. Soil samples were collected from the filtersand down to 70 cm depth. The water filled porosity was calculated from water content and porosity.

tively of the grain size distribution curve, established by sieve analysis) was laid out on top of the natural glaciofluvial material.

Water was pumped from a local creek, Kallforsån, one kilometer north of the infiltration site and spread out over the infiltration pond via a set of aeration stairs in the centre of the pond. 800–2 500 m^3 (average 1 600 m^3) of water was pumped to the infiltration pond typically for 11 hours per day. The large variation in infiltrated volumes is explained by a varying division of water between one, two or three infiltration ponds. At the beginning of the monitoring period all the water was transferred to one pond; later two or three ponds were used for infiltration. The infiltration water was divided evenly between the two or three ponds.

The groundwater level was monitored automatically every half hour for 102 days of the study period. The groundwater table varied during this period between 2.5 and 4 meters below the top of the filtersand. The water filled porosity in the vadose zone, i.e. the percentage of the pores that was filled with water was calculated from the water content and the porosity (Frycklund 1995) (Fig. 1).

Water sampling

Water samples were collected approximately once a month during the period from March to October 1992. Samples were taken from the infiltration water (before infiltration) and further along the water pathways by means of suction lysimeters installed at various depths in the filtersand (Fig. 1). Twenty-two cylindrical, ceramic filters with a pore size of approximately 1 μm, were installed at a depth of 20 cm and spread out over the whole pond area. Furthermore, four profiles of lysimeters were installed along a diagonal line across the infiltration pond at depths between 10 and 115 cm. Unfortunately only three of the lysimeter profiles yielded water. Four of the twelve operating profile lysimeters were made of stainless steel, cylindrical and with a pore size of 2 μm; the remainder were of the same ceramic type as described above.

The water samples were sucked from the lysimeters into an acrylic glass vacuum-bottle and transferred to a sampling bottle.

Samples were also taken from a groundwater tube made of galvanized steel. The tube was situated next to the infiltration pond, 3–4 meters from the edge of the pond. The tube was slotted at the bottom, 5.6 meters below the top of the filtersand.

Each sampling occasion usually lasted for two days, because of the time-consuming sampling procedure.

Water analysis

The water samples were analyzed for iron with atomic absorption spectroscopy. Selected samples were filtered in the laboratory through 0.45 μm membrane and the iron concentration was compared to that of the unfiltered samples. No significant differences were found.

Soil sampling

In order to quantify any accumulation of iron that might occur, samples of the filtersand were taken twice: before and after the water sampling period. Samples were taken both as individual point samples and as composite samples representing the entire pond at a specific depth, down to a depth of 70 cm (Fig. 1). Approximately 60 samples were taken each time.

Soil analysis

The content of iron in the filtersand was determined on the fraction of the soil with a grain diameter < 2 mm or < 1 mm. Iron was extracted with dithionite-citrate (Fe-d) and with pyrophosphate (Fe-p) according to van Reeuwijk (1987) using the Holmgren procedure. Iron was analyzed by plasma emission spectroscopy (Jobin Yvon JY24). The dithionite-citrate extraction is thought to remove the total amount of free iron that is not included in the silicate minerals while pyrophosphate extracts preferably the organically bound iron (Birkeland, 1984).

Analytical and methodological considerations

When the lysimeters were taken out of the pond after the water sampling period, some of them appeared to be cracked. Such lysimeters have been assumed to be intact and yield water representative of the percolating water up until a certain point in time (for more details see Frycklund 1995). The data presented here accordingly only includes the representative water samples.

The possibility that the lysimeters may have acted as iron-filters has been considered. The infiltration procedure included aeration of the water before it reached the filtersand, for which reason saturated oxygen conditions can be assumed. The pH in the water was neutral (Table 1). These conditions, together with the soil-extraction data lead to the conclusion that the iron was present as Fe(III) hydrous oxide. Poorly ordered Fe oxide particles are 4–6 nm in diameter (Bartoli *et al.* 1992 and Carlson and Schwertmann 1987) and may form aggregates of 20–300 nm (Carlson and Schwertmann 1987). This is to be compared to the pore size of the lysimeter filters — 1 000 or 2 000 nm respectively. So the low iron concentration in the lysimeters is not likely to be a result of any filtering effect.

Table 1. Infiltration water chemistry in Järna over the period March–October 1992. Mean value from 7 samples.

	Mean value	Range
pH	6.9	6.4–7.3
HCO_3	43	35–47
Electric conductivity, mS m^{-1}	16.8	15.3–17.7
Mn, mg dm^{-3}	0.03	0.01–0.05
Ca, mg dm^{-3}	17	14–20
TOC, total organic carbon, mg dm^{-3}	7.9	3.8–9.4
Color, mg Pt dm^{-3}	38	20–60

Results and discussion

Water

The average concentration of iron in the water at 20 cm depth in the filtersand was 50–85% lower than in the infiltration water during the measuring period. At 20 cm depth the concentration was 0.03 mg l^{-1} except in March (0.05 mg l^{-1}), while the infiltration water contained 0.08–0.18 mg l^{-1} (Fig. 2). The iron concentration in these soil-water samples seemed rather independent of the variations in the concentration in the infiltration water.

Water samples from the three profiles A, B and C revealed no further reduction of iron concentration with depth, down to 115 cm (Table 2). At 10 cm depth the median iron concentration was 0.03 mg l^{-1}. Ten of the twelve samples from 10 cm depth contained ≤ 0.05 mg l^{-1}. Comparing infiltration water and soil water from the profiles, the iron concentration in the soil water was not reduced to quite the same low level in March and August as on other occasions. The relative reduction was 61% and 78% respectively in August while in March it was 40%, at the 10 cm level. Water from profile B generally contained more iron than did the other waters. The B-lysimeters were of the stainless steel type and the higher iron levels in these water samples may be due to contamination from the iron tubing connected to the lysimeters.

When comparing the small differences between the iron concentrations in the water samples, it must be kept in mind that the analyzed iron concentrations were often close to the detection limit at 0.01 mg l^{-1}. The reproducibility of the analysis is estimated to ± 0.015 mg l^{-1}, for which reason the differences with depth in iron values are not significant for the measuring occasions in May,

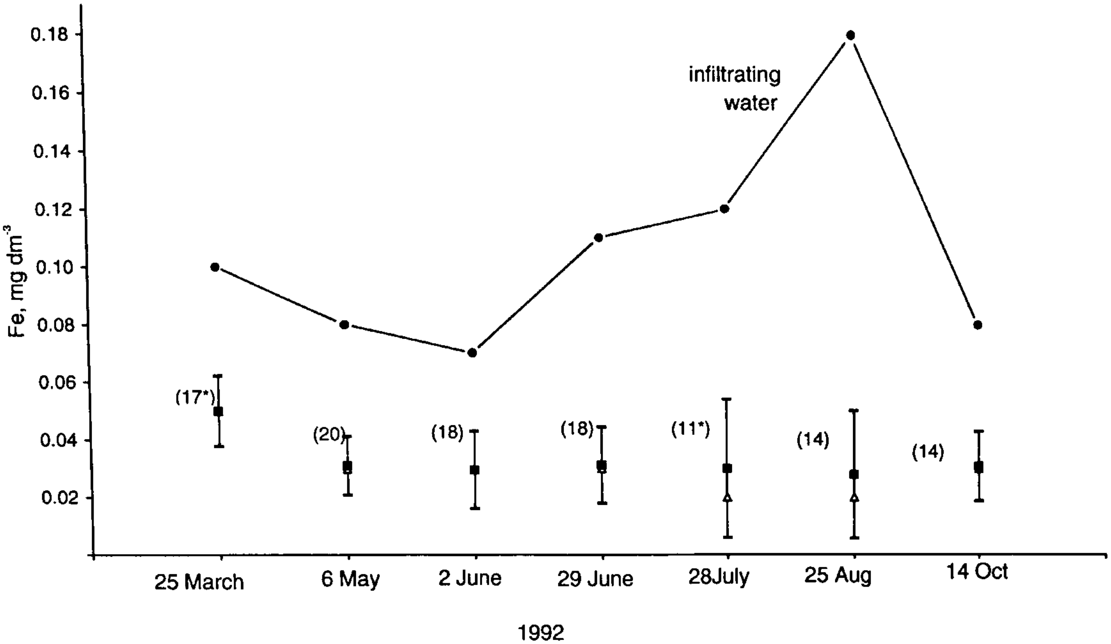

Fig. 2. The mean iron concentration in water samples from 20 cm depth in the filtersand (■), compared to the iron concentration in the infiltrating water (●). The standard deviation from the mean is indicated with vertical bars. Number of samples in brackets, where the * indicates that one outlier has been excluded. Median value for each sampling occasion is indicated with a Δ. The median value coincides with the mean value for 25 March, 2 June and 14 October. The values for the infiltrating water are given for one sample per occasion.

June, July and October. However, a clear general picture appears of how and where the iron retention takes place in the infiltration pond.

The retention of iron from the Järna infiltration water is fast, both in time and travel distance. Thus the zone where most of the iron retention takes place is thin and superficial and the retention would not be improved by further residence

Table 2. The iron concentration in unfiltered water samples from infiltrating water, groundwater and from the three profiles A, B and C in the filtersand during the period from March to October 1992. Groundwater was sampled in the tube situated 3–4 meters from the edge of the pond.

Sampling site, numbers refer to depth in cm	Fe, mg dm^{-3}						
	25 Mar.	6 May	2 Jun.	29 Jun.	28 Jul.	25 Aug.	14 Oct.
Infiltrating water	0.10	0.08	0.07	0.11	0.12	0.18	0.08
A 10	0.06	0.03	0.03	0.03	0.02	0.07	0.05
20	0.14	0.02	0.03	0.02	0.02	0.09	0.04
50	0.08	0.04	0.03	0.03	0.03	0.04	0.03
115	0.04	0.03	0.02	0.04	0.03	0.06	0.04
B 15	–	0.05	0.07	0.07	0.05	0.06	–
30	–	0.05	0.05	0.02	0.03	0.06	0.06
50	–	0.04	0.05	0.05	0.03	0.01	0.03
85	–	0.04	0.04	0.04	0.11	0.12	0.05
C 10	–	–	0.03	0.04	0.01	0.04	0.02
50	0.09	0.03	0.03	0.03	0.03	< 0.01	0.01
115	0.11	0.03	–	–	0.01	0.02	0.02
Groundwater	0.06	0.16	0.64	0.68	0.96	0.88	0.09

time or travel distance. Nevertheless, the production well at Järna waterworks contained high concentrations of iron, on average 0.18 mg l^{-1}, ranging from 0.11 to 0.23 mg l^{-1} during the period from March to October. This was attributed to the intermixture of natural, iron rich groundwater (Frycklund *et al.* 1994).

The iron concentrations in the water samples from the groundwater tube situated close to the infiltration pond are difficult to interpret, since it is not obvious which water they represent (Table 2). On all sampling occasions the groundwater level was 2.9–3.5 m below the top of the filtersand. This groundwater tube is open at 5.6 m below the top of the filtersand. Depending on the local flow pattern here, the tube may represent either young groundwater which has recently percolated from the adjacent infiltration pond or older groundwater coming from the sides, flowing underneath the artificially produced groundwater. These water samples were not filtered, thus the iron content in these samples may also originate from the groundwater tube itself, made of galvanized steel. The general flow pattern in the aquifer changes greatly when the artificial infiltration conditions are changed from the use of one infiltration pond to two or three ponds (Frycklund *et al.* 1994). The local flow patterns have not been determined in such detail that any conclusions can be drawn from the water samples taken from the groundwater tube.

Poorly matched infiltration sites and extraction wells is a common problem, due to insufficient knowledge of the flow paths under different operating conditions. Carefully designed tracer-studies may be useful tools in gaining such knowledge. This could be used to improve the groundwater quality at the artificial recharge scheme.

Soil

The analysis of the soil gave information on the iron oxide precipitates retained on the soil grains. The amount of extractable iron (Fe-d) in the top 5 cm of the filtersand had increased by 140% between the two sampling occasions in 1991 and 1993 (Table 3). At the 5–15 cm level the increase was 10%. Below this level the differences in Fe-d between the two sampling occasions were insignificant. This means that 86% of the accumulation of extractable iron (Fe-d) took place in the top 5 cm and only 14% in the next 10 cm.

According to the Fe-p analysis, about 10% of the retained iron was associated with organic matter in 1991 while in 1993 this percentage had increased to about 20 (Table 3). There were only small differences in this percentage when comparing the top 5 cm with the level 5–15 cm. The increase in iron associated with organic matter corresponds to the increased content of organic carbon in the filtersand, as described in Frycklund (1995).

A tentative iron-budget can be made, based

Table 3. Extractable iron in the filtersand. Fe-d is the iron extractable with dithionite-citrate ("free iron") and Fe-p the iron extractable with pyrophosphate (iron associated with organic matter). Values are presented as % by weight.

	Fe-d, %		Fe-p, %		Fe-p/Fe-d	
Level	1991	1993	1991	1993	1991	1993
0–5 cm	0.154 ± 0.014 *n* = 6	0.366 ± 0.033 *n* = 4**	0.015 ± 0.005 *n* = 30	0.079 ± 0.023 *n* = 30**	0.10	0.22
5–15 cm	0.168 ± 0.008 *n* = 2*	0.186 ± 0.017 *n* =1	0.013 0.011– 0.018 *n* = 5**	0.029 0.012– 0.037 *n* = 5**	0.08	0.16

Note: For the Fe-d analysis the analytical accuracy measured as the average variation between triple analysis of each sample, is indicated. For the Fe-p analysis the range of variation is indicated.

* of which 1 was a composite sample based on 15 samples.

** of which 2 were composite samples each based on 15 samples.

on the amount of iron that had been supplied to the infiltration pond between the two sampling occasions, according to the following equation:

$$\frac{(\mathrm{Fe}_0 - \mathrm{Fe}_{50}) \times V \times d}{670\ \mathrm{m}^2} = \left[\frac{\mathrm{mg\ Fe}}{\mathrm{m}^2}\right],$$

where: Fe_0 = iron concentration in infiltration water, mg l^{-1}, Fe_{50} = iron concentration ion soil water at 50 cm depth, mg l^{-1}, V = daily infiltrated volume of water, m ^{3}, d = number of days.

During this 15-month period, 165 g Fe per square meter had been supplied. After subtracting an estimate of the iron that passed through the filtersand from the supplied amount, it would be expected to find 138 g Fe per square meter accumulated in the filtersand. When the figures in Table 3 are adjusted to the ratio between the surface area of the fraction <1 mm and the surface area of the whole soil, the figures correspond surprisingly well. Based on the analysis of Fe-d in the filtersand, 145 g Fe had been accumulated per square meter.

The results support the conclusions from the water analysis, i.e. that the iron is efficiently retained in the filtersand. They also relieve the concern about any considerable accumulation of iron deeper in the subsoil, caused by the supply of iron with the infiltration water. This implies that the simple maintenance routine applied at the waterworks, consisting of regular removal of the top 10 cm of the filtersand and subsequent cleaning, is a sustainable form of maintenance at this site.

Infiltration conditions

The periodical pumping of water to the infiltration pond resulted in alternation between conditions when the filtersand was submersed, with typically 40 cm deep water, and conditions when the filtersand was exposed to the atmosphere, allowing air to enter the pores. The time lapse between pumping periods was however not long enough to allow the filtersand to dry up: it remained moist throughout. The periodical pumping resulted in a daily fluctuation in the groundwater table of 30 to 60 cm.

The water filled porosity was 32–93% in the filtersand (Frycklund 1995), with the highest values close to the transition zone between the filtersand and the underlying natural material (Fig. 1). All measurements of water content showed that unsaturated conditions prevailed in the filtersand. Fine material was found accumulated in the transition zone (Frycklund *et al.* 1994). The fine material dammed the infiltration water above the transition zone, explaining the observed higher degrees of saturation there.

An indication of the infiltration rate was given by repeated measurements of the water level in the infiltration pond during periods with no supply of infiltration water to the pond. The rate at which the water level was lowered was usually around 40 mm h^{-1}, but on warm afternoons could reach 85 mm h^{-1}. This is a measure of the combined effect of infiltration and evaporation. The variation in rate is explained by *i.* the decrease in water viscosity with increasing temperature and *ii.* the increased evaporation from the free water surface with increasing temperature. The measurements of the water level indicate that the actual infiltration rate was more than ten times lower than the theoretical infiltration capacity, calculated from the texture and porosity of the filtersand. The Fair-Hatch equation (Freeze and Cherry 1979) gives an infiltration capacity of 700 mm h^{-1} at 15 °C for the Järna filtersand, when the effective porosity is estimated to be half the measured total porosity. This value corresponds well to laboratory measurements on the same material, giving a saturated hydraulic conductivity of 800 mm h^{-1} (B. Espeby and B. Nilsson, unpubl.). The discrepancy between these values and the field measurements may be due to accumulation of organic (and inorganic) material in the pores and to the unsaturated flow conditions, both factors reducing the effective porosity even further.

The measured changes of water level in the pond indicate that there was no substantial bypass flow in the filtersand. This conclusion is also supported by the well-balanced iron-budget above.

Iron-organic complexes

Iron-organic complexes in natural waters have been studied by several researchers during the last decades (e.g. Theis and Singer 1974, Wehrli *et al.* 1989, Reckhow *et al.* 1991, Lovley *et al.* 1991, Bartoli *et al.* 1992, Liang *et al.* 1993). The formation of these complexes seem to depend on the strong affinity of the hydroxy-groups on organic molecules for the surface ferric iron atom in ferrihydrite (Cornell and Schwertmann 1979, as cited by Liang *et al.* 1993). Ferrihydrite is a secondary iron oxide with a low degree of crystallinity and variable-charge surface. The structure and formula of the oxide are not fully known or agreed upon, but it clearly has potential to influence a wide range of soil properties. It has a large specific surface area and may for example interact strongly with organic molecules in soils (Childs 1992).

The above mentioned affinity leads to the formation of complexes with iron and organic anions instead of the gradual transition towards the more crystalline hematite through internal rearrangement and dehydration of the ferrihydrite, which would be the normal ageing process.

The adsorptive mechanisms behind the complexation are not yet fully understood although different mechanisms have been proposed. Bartoli *et al.* (1992) describe the complexation process between the humic anion and the positively charged ferrihydrite as adsorption of predominantly electrostatic origin. The iron-organic complexes were found to have a net negative charge. They repelled each other and thus dispersed in the water. The complexes were 7–15 nm in diameter. Others have found the carboxyl and hydroxyl functional groups of natural organic matter to be adsorbed through surface complexation — ligand exchange to iron oxide (Wehrli *et al.* 1989, Gu *et al.* 1995).

Liang *et al.* (1993) found that negatively charged, stable iron-organic colloids with ferric iron, caused increased turbidity in the groundwater. The organic molecules involved in the complexation were natural organic molecules with hydroxy-groups. Non-hydroxy organic acids, such as formic and acetic acid, did not have the complexing ability.

Iron-organic complexes may form with both the reduced (ferrous, Fe-II) and the oxidized form of iron (ferric, Fe-III) (Theis and Singer 1974). Natural organic acids can strongly inhibit the oxidation of Fe-II in an environment where Fe-II alone would not be stable, due to complexation of the iron by the organic acid. A dissolving process is also possible: Fe-II may have a catalytic effect, together with hydroxy and carboxylic groups on natural organic matter, in the dissolution of Fe-III hydroxides. The result will be a negatively charged Fe-III organic molecule dissolved in water (Wehrli *et al.* 1989).

It may be concluded that the type of iron-organic complexes that cause problems in some waterworks are likely to be found when the water contains organic molecules with hydroxy-groups. These are found on aquatic humic substance, hydrophilic acids, carbohydrates and simple alcohols (Thurman 1985).

The content of iron-organic complexes has not been analyzed in the Järna infiltration water. However, the effective iron retention in Järna implies that if there is any iron present in the form of the negatively charged iron-organic complexes described above, the share of the total iron content is minor.

As regards methods for removing the iron-organic complexes from the water phase, Reckhow *et al.* (1991) found that ozone treatment is efficient only for iron-organic complexes where the oxidized form of iron is present and for iron-organic complexes where the organic matter is of non-humic aquatic type. The iron in complexes formed under reducing conditions could not be oxidized and precipitated by ozone.

Certain bacteria might be able to help remove the ferrous iron-organic complexes. The species *Siderocapsa* live in environments where the ferrous iron alone is unstable but is present in iron-organic complexes. *Siderocapsa* use the organic compound for energy and release the ferrous ion, which oxidizes and precipitates as ferric hydroxide on the cell surface (Lundgren and Dean 1979). This species has been found in Finnish infiltration schemes (Hatva 1989).

Bacteria are known to be involved in the precipitation of iron (hydr)oxides (Fenchel and Blackburn 1979, Lundgren and Dean 1979, Hallberg 1988, Hatva 1989). In this study no determination of the presence of bacteria has been made. The invariability with time and temperature in the iron concentration in the soil water, however, indicates that the retention mechanism to a large degree is abiotic. It would be expected that if bacteria were involved to any great extent, the retention would be less effective in March and October when the water and air temperatures were low.

Induced recharge

The problem with gradually increasing iron concentrations in wells based on induced recharge may be better understood in view of studies by Lovley *et al.* (1991). They studied how the reduction of Fe-III coupled to oxidation of organic matter in aquatic sediments was affected by the presence of Fe-III-reducing microorganisms.

The reducing conditions that develop when oxygen and nitrogen is consumed upon the oxidation of organic matter did not themselves result in reduction of Fe-III. Under sterile conditions Fe-III was only slightly reduced at pH 6.7. Only in the presence of Fe-III-reducing microorganisms were the organic compounds readily oxidized with reduction of Fe-III (Lovley *et al.* 1991). This course of events may very likely be taking place in groundwater systems where wells based on induced recharge show a delayed increase in iron concentration; the development of a population of Fe-III-reducing microorganisms is a slow process that will substantially affect the water quality only after some time, with increasing iron concentrations.

Lovley and Phillips (1988) isolated such Fe-III-reducing microorganisms from freshwater sediments. They were identified as rod-shaped organisms, 2 to 4 by 0.5 µm, with an obligately anaerobic respiratory metabolism and were designated GS-15.

As long as ferrous iron-organic complexes are not formed in the aquifer, as described by Wehrli *et al.* (1989), the dissolved iron may be oxidized upon aeration and subsequently precipitated. Theoretically, the biotic dissolution of Fe-III may be prevented by oxygenation of the bottom sediments or the aquifer, although this may be a costly installation. If concentrated zones of infiltration could be localized, such oxygenation would be facilitated.

In Järna the production well yielded groundwater with the highest iron concentrations during periods when the operating conditions at the recharge scheme resulted in a specific flow pattern. This was characterized by a flow of groundwater from a waterlogged area adjacent to a creek. It is most probable that during these conditions a large proportion of the abstracted groundwater was a result of induced recharge from the creek.

Filtersand design

The grain size distribution of the filtersand in Järna seems to be well suited for the retention of iron. The Fe(III) hydrous oxide is retained on the surface of the soil grains, hence the surface area of each cubic meter of filtersand would be a more appropriate parameter than the grain size distribution, as an indicator of the adsorptive capacity of the soil. An estimation of the specific surface area based on the grain size distribution and assuming spherical grains (Hillel 1980), gives that 86% of the surfaces of the filtersand grains are situated on the grains <1 mm. By weight, this fraction only constitutes 5% of the soil. Thus, a small increase by weight of the fraction <1 mm, results in a proportionally larger increase in the adsorptive capacity of the soil, since the adsorption processes take place at the surfaces of the soil grains.

In the choice of textural composition of the filtersand the adsorptive capacity has to be balanced against the hydraulic capacity. This choice also calls for consideration of the need for a thin zone of fine-grained material for the collection of suspended solids, protecting the aquifer from clogging at deeper levels. Furthermore, the textural composition and stratification of fine and coarse layers in the filtersand affect the flow conditions. Unsaturated flow conditions lead to more effective adsorption processes through more physical contact between the water and the soil grains. Saturated flow conditions, on the other hand, result in a larger hydraulic capacity at the expense of the cleaning effect. Systematic studies of these effects have, however, not been made.

It would be valuable if the optimal composition of a filtersand could be evaluated with respect to the aspects above. Column studies under controlled conditions could lead to insights, which could be used for outlining “guidelines for artificial recharge schemes”.

Conclusions

The results from Järna waterworks show that most of the iron supplied to the infiltration pond with the infiltration water was quickly retained in the filtersand. The iron concentration was reduced after 20 cm travel distance in the filtersand, to a level that is fully acceptable in drinking water. However, due to unfortunate flow patterns in the aquifer and/or to an inefficient location of the well, the iron concentration in the abstracted groundwater exceeded stipulated guidelines. This stresses the need for improved knowledge of local flow patterns, when artificial groundwater recharge is to be applied.

The iron retention did not seem to vary with season or temperature. This has been interpreted as reflecting a largely abiotic process.

While the residence time was not important for the efficiency of iron retention, the supply of surfaces for adsorption seemed to be a stimulating factor. Hence, increasing the proportion of fine fraction in the filtersand would promote the adsorptive, cleaning processes. Unsaturated flow conditions would most likely further improve the adsorption, through the good physical contact between the water and the solids. Future column studies could throw more light on these relations.

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