Variation in groundwater quality in South Estonian rural areas

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Abstract Since 1991, the groundwater monitoring has been carried out in the experimental watershed of Porijõgi River (276 km²), South Estonia. Water samples to analyze the content of NH₄, NO₂, NO₃, PO₄, total-P, and SO₄ in water were taken from 46 dug wells (with 0.5-10 m depth) four times a year (in spring, summer, autumn and winter). Three main subwatersheds (upper course, forested area and extensive agricultural use, 12 wells; Sipe subwatershed, moderate agricultural use, 11 wells; Vânda subwatershed, intensive agriculture, 23 wells) have been studied. The mean values of NO₃ in groundwater top layer of intensively used agricultural areas exceed the official EC quality standard for drinking water: 50 mg l⁻¹. In the study area, only in forested and extensively used upper course watershed the NO₃ concentration is still low (<15 mg l⁻¹, in some springwater even <1 mg l⁻¹). In the intensively used subwatershed, the maximal values reached 51 and mean values 19 mg NO₃-N l⁻¹. Highly significant correlation (R²=0.78) between the N fertilization intensity and NO₃ concentration in the groundwater top layer was found. In dug wells of the intensively used subwatershed with non-fertilized buffer zones (perennial grasslands with hedges or tree rows), the content has been significantly lower than in those adjacent to fertilized fields. Also, negative correlation between the well depth and NO₃ concentration in water was found. The mean value of SO₄-S concentration is also related to fertilization intensity but is relatively low (<60 mg l⁻¹). Phosphorus concentrations (mean: <0.1 mg l⁻¹) have been found higher under the sandy soils. Despite decreasing NO₃ concentrations in stream runoff of the study area during the last years, there was no decrease of NO₃ in groundwater. As expected, in the water samples of early spring and late autumn NO₃ concentrations have been found significantly higher. Some alternative measures to clean up the contaminated with nitrates drinking water and to control nutrient fluxes in the intensively used watershed (planted soil filters combined with coral chalk filters, macrophyte ponds, reconstructed manure composting sites, buffer zones and buffer strips) have been proposed.

INTRODUCTION

Groundwater quality has become one of the priority problems of environmental protection in many countries with intensive industrial and agricultural development. Various hazardous contaminants, especially pesticides and nitrates enter aquifers by leaching from intensively used agricultural lands. Also, sewage and sludge disposal on lands, leaching from sanitary landfills and solid waste deposits, manure piles and composting places add contaminants to the groundwater (Freeze & Cherry, 1979). In heavily loaded agglomerations of Western Europe, Japan and United States, about 30% of drinking water resources have the NO₃ concentration value higher than the internationally accepted quality standards, i.e. 50 mg NO₃ l⁻¹ according to EC and WHO
European standards, and 10 mg NO$_3$-N l$^{-1}$ according to US EPA standards (Freeze & Cherry, 1979). Despite different activities to improve the drinking water quality like closing contaminated wells, using deeper groundwater layers and cleaning up the water, the nitrate problem is still very serious. For instance, in Nordrhein-Westfalen, Germany, the share of contamination with NO$_3$ (>50 mg l$^{-1}$) groundwater resource has been decreased only from 8.5% in 1979 (Sunkel, 1983) to 8.4% in 1986 (Toussaint, 1989). Also, phosphates, potassium and sulphates could contaminate groundwater leaching from the intensively fertilized arable lands on sandy soils (Toussaint, 1989).

The purpose of this study was to determine dynamics of the shallow groundwater quality in South Estonian rural areas during the transition period from collective to private farming. In this region, the groundwater formed in gravels, sands, and till of the Quaternary era is the main drinking water source. High values of NO$_3$ concentration in the groundwater of the area were investigated already in 1986-1987 (Mander et al., 1989) and also by other researchers. For example, Metsur et al. (1991) investigated that during the period 1987-1990 the share of contaminated with NO$_3$ wells (total number of wells studied was 510) in 6 collective farms of the Tartu county was 57-77%. In about 45% of wells within the large agricultural fields of the Valga county, the NO$_3$ content exceeded 50 mg l$^{-1}$. In some cases the contamination was higher than standards by factor 10 (Veldre et al., 1991). Though the high nitrate values in South Estonia were well-known, the factors influencing the accumulation of NO$_3$ in shallow groundwater, the dynamics and behavior of nutrients in the wells have not been studied before.

**STUDY AREA**

The Porijõgi River drainage basin (276 km$^2$) has been chosen to investigate the groundwater quality dynamics in South Estonia. This area represents reasonably the whole South Estonian landscape. It is located on the border of two landscape regions: the Plain of South-East Estonia and the Otepää Heights (Varep, 1964). The central and northern parts of the Porijõgi River catchment area lie within the South-East Estonian moraine plain in the vicinity of Tartu (58° 23' N; 26° 44' E), the second largest town in Estonia. The absolute altitude of the plateau is from 30 to 60 m, relief is undulated (slopes achieve normally 5-6%) and intersected by primeval valleys, 3-5 km wide and up to 40 m deep, formed by streams during the pleistocene and remodelled by glaciers of the last glaciation. The southern part of the drainage basin lies on the northern slope of Otepää Heights that is formed from moraine hills and kames with extreme variety of glacial deposits. The altitude of this region is up to 120 m, the relative heights reach 30-35 m.

The bedrock of the whole catchment area is formed by red Devonian sandstone (compact sandstone with clay and aleurolite layers of aruküla and burtnieki times) which is covered by loamy sand-till of the Weichselian glaciation or fluvioglacial and glaciolacustrine sands and gravels. The Devonian sandstone lies in the depth from 2 m (in lower course) to 60 m (on the hills of Otepää Heights). The depth of the groundwater top layer is quite different (0.5-20 m) depending on relief and geomorphologic conditions. The upland soils are mostly loamy sands and fine sandy loams with the surface soil organic matter content and pH value in the cultivated fields being 1.6-1.9% and
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5.6-6.5, respectively. On steeper gradients the soils have been eroded. About 50% of this area is used as arable land. In valleys and other lower areas the clay soils and peatlands, partly used as perennial grasslands, dominate. Fertilization rate of intensively managed fields reaches 150 kg N, 70 kg P, and 100 kg K ha⁻¹ year⁻¹. In forested areas (about 25% of the territory), coniferous and mixed forests are most common. In riparian zones, alder forests and willow bushes occur. Bogs covered with pine or birch forests are the most common wetland communities on the watershed borders. The average amount of precipitation during the research period is 780 mm year⁻¹. During the study period the winters have been relatively mild (mean air temperature for the coldest month February has been only approximately -5 °C), snowcover lasted only for few days between November and April.

The Porijõgi River catchment area has been divided into 8 subwatersheds with different land-use structure. Three of them are of most importance for the nutrient cycling study: (1) the Upper course of Porijõgi (10.4 km²) on the slope of Otepää Heights, relatively undisturbed and mostly covered with forests (>75%), (2) the subwatershed of the Sipe stream (9 km²) with medium intensity of agricultural use and wide riparian wetland zone along the stream, and (3) the subwatershed of Vánda ditch (4.2 km²) with intensive agricultural use and more than 65% of arable land.

Detailed description of the land-use structure of the Porijõgi catchment area has been presented earlier (Mander et al., 1989).

MATERIAL AND METHODS

The surface water quality monitoring in the Porijõgi River drainage basin started systematically already in 1987. To have better overview of the nutrient fluxes of this representative basin, investigations of the groundwater quality were added to the research program. First groundwater analyses were made in autumn 1986 and from spring to summer 1987. Water samples were taken from 18 dug wells (10 in the Vánda subwatershed, four in the Sipe subwatershed and, four in Upper course subwatersheds). The data indicated the high NO₃ concentrations of groundwater top layer in fertilized fields (Mander et al., 1989). In systematically organized groundwater monitoring started in 1991, water analyses were made four times a year (in spring, summer, autumn and winter). Altogether 46 dug wells were chosen for the study (incl. those from earlier investigations): 23 in Vánda subwatershed, 12 in Porijõgi Upper, and 11 in Sipe stream subwatershed. All dug wells are in use, they have a clay collar and roof, and there are no significant local point pollution sources (i.e. barns, toilets, manure piles, fertilizer or oil storage sites etc.) in the 50 m radius from the well. Depth of the water table in wells (measured with the 20 m plastic ribbon) varies between 0.5 and 10 m. In each of the dug wells it changes also seasonally depending on precipitation and the water use intensity. The 0.5-1.5 m deep dug wells of the Porijõgi Upper subwatershed represent springs seeping on the forested hillslopes. Dug wells are normally constructed from concrete rings 0.95 m in diameter. The water samples were taken from the layer 1 m below the water surface in 2 liter plastic bottles and transported to the laboratory. All samples from wells were analyzed for pH, NH₄-N, NO₂-N, NO₃-N, PO₄-P, total-P, and SO₄-S. All water analyses were made according to former COMECON- countries standard methods compatible to the international
methods for examination of water and wastewater quality (APHA, 1981). The pH value was measured in the laboratory with glass electrode-pH-meter. Dissolved ammonium-N and sulphate-S were determined colorimetrically, according to Golterman et al. (1978). Nitrite and nitrate were measured in acidified samples by reduction on cadmium amalgam and colorimetry (APHA, 1981). The total phosphorus in water samples was determined colorimetrically after conversion to orthophosphate by wet digestion with acid. Determination of dissolved orthophosphate followed colorimetrically as blue phosphomolybdate, without extraction (Golterman et al., 1978).

Data about fertilization and crops yield have been taken from the annual statistical reports of former collective and state farms locating in the study area. Private farmers were asked at their farms. In the Vânda ditch watershed, main events of fertilization were checked by authors. From the same area, yield samples were taken to determine the C, N, P, and K concentration in biomass. Determination of the nutrient runoff by stream water has been described earlier (Mander et al., 1989). Nutrient input with precipitation has been estimated on the base of investigations made by the meteorological station of the civil airport in Úlenurme locating in 3 km from the center of Vânda ditch subwatershed.

RESULTS AND DISCUSSION

The average values of groundwater quality parameters studied during the period 1986-1993 in the Poriõogi River catchment area are presented in Table 1. It demonstrates typical groundwater situation in South Estonian agricultural landscapes.

<table>
<thead>
<tr>
<th>Subwatershed</th>
<th>pH</th>
<th>NH₄-N</th>
<th>NO₂-N</th>
<th>NO₃-N</th>
<th>PO₄-P</th>
<th>Total-P</th>
<th>SO₄-S</th>
</tr>
</thead>
<tbody>
<tr>
<td>Poriõogi Upper</td>
<td>n 59</td>
<td>59</td>
<td>59</td>
<td>59</td>
<td>46</td>
<td>46</td>
<td>12</td>
</tr>
<tr>
<td>mean</td>
<td>7.1</td>
<td>0.01</td>
<td>0.02</td>
<td>3.7</td>
<td>0.05</td>
<td>0.08</td>
<td>14</td>
</tr>
<tr>
<td>SD</td>
<td>0.2</td>
<td>0.05</td>
<td>0.07</td>
<td>5</td>
<td>0.06</td>
<td>0.08</td>
<td>9</td>
</tr>
<tr>
<td>min</td>
<td>6.7</td>
<td>0</td>
<td>0.004</td>
<td>0.03</td>
<td>0.01</td>
<td>0.02</td>
<td>0.4</td>
</tr>
<tr>
<td>max</td>
<td>7.6</td>
<td>0.05</td>
<td>0.5</td>
<td>18</td>
<td>0.22</td>
<td>0.3</td>
<td>32</td>
</tr>
<tr>
<td>Sipe</td>
<td>n 53</td>
<td>54</td>
<td>54</td>
<td>53</td>
<td>48</td>
<td>48</td>
<td>14</td>
</tr>
<tr>
<td>mean</td>
<td>7.1</td>
<td>0.01</td>
<td>0.01</td>
<td>8.3</td>
<td>0.03</td>
<td>0.06</td>
<td>35</td>
</tr>
<tr>
<td>SD</td>
<td>0.2</td>
<td>0.1</td>
<td>0.1</td>
<td>6</td>
<td>0.06</td>
<td>0.06</td>
<td>21</td>
</tr>
<tr>
<td>min</td>
<td>6.7</td>
<td>0</td>
<td>0</td>
<td>0.1</td>
<td>0.01</td>
<td>0.01</td>
<td>7</td>
</tr>
<tr>
<td>max</td>
<td>7.9</td>
<td>0.4</td>
<td>0.028</td>
<td>24</td>
<td>0.4</td>
<td>0.4</td>
<td>68</td>
</tr>
<tr>
<td>Vânda</td>
<td>n 125</td>
<td>181</td>
<td>183</td>
<td>186</td>
<td>111</td>
<td>111</td>
<td>42</td>
</tr>
<tr>
<td>mean</td>
<td>7.1</td>
<td>0.3</td>
<td>0.04</td>
<td>19</td>
<td>0.04</td>
<td>0.06</td>
<td>62</td>
</tr>
<tr>
<td>SD</td>
<td>0.2</td>
<td>1.71</td>
<td>0.27</td>
<td>13</td>
<td>0.07</td>
<td>0.08</td>
<td>27</td>
</tr>
<tr>
<td>min</td>
<td>6.5</td>
<td>0.01</td>
<td>0.002</td>
<td>2.5</td>
<td>0.01</td>
<td>0.01</td>
<td>10</td>
</tr>
<tr>
<td>max</td>
<td>7.9</td>
<td>15</td>
<td>2.8</td>
<td>51</td>
<td>0.46</td>
<td>0.48</td>
<td>156</td>
</tr>
</tbody>
</table>

n - number of investigations
SD - standard deviation
Nitrogen

The most complicated problem for the whole area is very high concentration of nitrates in the groundwater. In the intensively used Vânda ditch subwatershed the mean value was 19 mg NO$_3$-N l$^{-1}$ which exceeds two times the international quality standard. The highest values exceed even 50 mg NO$_3$-N l$^{-1}$. Even in the forested and extensively used Porijõgi Upper subwatershed the maximal NO$_3$-N concentrations reach the quality standard limit. Also, very high maximum values of toxic NO$_2$-N (up to 0.5 mg l$^{-1}$; quality standard 0.1 mg NO$_2$ l$^{-1}$; Toussaint, 1989) in some wells indicate the fertilizing influences in the subwatershed (Table 1). Ammonium nitrogen was found to be relatively low compared to intensively used agricultural areas in the North and West Estonian limestone plateau (Veldre et al., 1991). However, in the Vânda subwatershed the mean value for NH$_4$-N l$^{-1}$ - 0.3 mg reaches quality standards (0.5 mg NH$_4$ l$^{-1}$; Toussaint, 1989), the maximum values exceed 10 mg l$^{-1}$. Last was observed in the dug well close to one of the manure composting sites (Fig. 3). Relatively high standard deviation of ammonium and nitrite ions may be caused by too little number of analyses, but demonstrate also an agricultural impact.

Fig. 1 Variation of nitrate nitrogen and total phosphorus in the groundwater of Porijõgi River subwatersheds.
There is no correlation between soil structure and nitrate concentration in the groundwater top layer of the study area. Main factor influencing the NO$_3$ dynamics seems to be the agricultural practice, especially manure application and use of mineral fertilizers. Highly significant positive correlation ($R^2=0.78$; 95% significant) between the N fertilization intensity and the nitrates’ concentrations in groundwater was found (Fig. 2). This relationship has been characterized by others authors, mainly in case of pig slurry application (Vetter & Steffens, 1981), cultivation of intensively fertilized crops (Roth & Fox, 1990), and irrigation/fertilization of fields (Weil et al., 1990). In the study area, all these practices were in use.

![Graph](image)

Fig. 2 Correlation between the fertilization intensity (kg N ha$^{-1}$ year$^{-1}$) and nitrate concentration in the groundwater top layer (mg NO$_3$-N l$^{-1}$) of the Porijõgi River catchment area.

A complicated and interesting at the same time question is the temporal and spatial dynamics of nitrates in the groundwater. Different simulation models have been created to describe this process (Freeze & Cherry, 1979; Kinzelbach, 1991). Unfortunately, these models do not describe exactly the changes of nitrate concentrations in the study area of this paper.

Seasonal variations of NO$_3$ concentrations appear only from a known contamination level. So, in the Porijõgi Upper and Sipe subwatersheds no remarkable seasonal fluctuations were observed, but in the groundwater of Vända subwatershed a slight increase of NO$_3$ concentrations in late autumn and winter appears (Fig. 1). At that time from arable lands a lot of nitrogen will be leached. In Table 2, the nitrogen and phosphorus budget of the Vända subwatershed are presented. It demonstrates that during the period 1988-1992 from 22 to 72 kg N ha$^{-1}$ year$^{-1}$ was not removed with the crops. This is approximately on the same level as registered in Danish agricultural landscapes (45-68 kg N ha$^{-1}$ year$^{-1}$), but in Denmark the mean annual fertilization intensity is 60-120% higher than in our study area: 136 kg as mineral N and 53-81 kg as N in animal manure per hectare (Kronvang et al., 1993). One reason for NO$_3$ increasing in winter time may be also the plowing of temporary grasslands in autumn that happened last two years in some fields (Table 3). These areas are an important potential source for groundwater pollution (Cameron & Wild, 1984).

More complicated than to explain the seasonal dynamics is to explain long-term development of the NO$_3$ concentrations. During the last three years a slight decrease of fertilization intensity in the study area was observed. Recently, this tendency has been more intensive for the whole agricultural areas in Estonia. From 1988 to 1991, the N, P, K, and pesticide use was dropped as
Table 2 Nitrogen and phosphorus budget in the Vânda subwatershed.

<table>
<thead>
<tr>
<th>Main flows</th>
<th>N kg ha(^{-1}) year(^{-1})</th>
<th>P kg ha(^{-1}) year(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inputs</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Atmospheric deposition*</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>Fertilization**</td>
<td>109</td>
<td>85</td>
</tr>
<tr>
<td>Outputs</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Plant production***</td>
<td>65</td>
<td>71</td>
</tr>
<tr>
<td>Runoff**</td>
<td>14</td>
<td>12</td>
</tr>
<tr>
<td>Others****</td>
<td>40</td>
<td>12</td>
</tr>
</tbody>
</table>

* - estimated
** - measured
*** - estimated on the base of yield data and mean concentrations in the biomass
**** - accumulation in the soil, ammonia volatilization, denitrification, and leaching into deeper groundwater

Table 3 Variation of the share of grasslands territory in Vânda subwatershed.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>ha</td>
<td>121.9</td>
<td>192.6</td>
<td>171.3</td>
<td>99.3</td>
<td>128.3</td>
</tr>
<tr>
<td>%</td>
<td>36.2</td>
<td>57.3</td>
<td>50.9</td>
<td>29.5</td>
<td>38.2</td>
</tr>
</tbody>
</table>

following: from 101 to 65, from 60 to 45, from 89 to 70, and from 1.8 to 1.3 kg ha\(^{-1}\) year\(^{-1}\), respectively (unpublished official data from the Ministry of Agriculture, Estonia). In spite of the decreasing fertilization intensity, no significant decrease of NO\(_3\) concentrations in the groundwater of the Porijõgi Upper and Sipe subwatersheds was observed (Fig. 1). In the Vânda subwatershed, the fertilization rate did not drop essentially (Table 2). That is why the groundwater is still contaminated with nitrates (Fig. 1). Here, during the 1970's and 1980's a lot of pig slurry was applied (even on the snowcover), as well intensive irrigation, use of mineral fertilizers, and monocultural crop-rotation (mostly barley) dominated. These practices have a long-term influence and could impact the groundwater quality for a longer period. However, it is unknown now, how long time it will take for groundwater in South Estonian moraine landscapes, where loamy tills on compact sandstone with low water permeability rate dominate, to get clean again.

The spatial differences of NO\(_3\) concentrations in groundwater were studied on the base of Vânda subwatershed (Fig. 3). Firstly, in upper parts of the subwatershed (southern and northern parts), where the arable lands dominate, the highest NO\(_3\)-N concentrations (>20 mg l\(^{-1}\)) occur. Secondly, under the permanent grasslands in the north-east and north-west, the groundwater with allowed quality (<10 mg NO\(_3\)-N l\(^{-1}\)) dominates. The lowest values have been found in the floodplain of the Porijõgi River (Fig. 3). The last fact could be explained by higher denitrification ratio of organic soils dominating in the floodplains. Decrease of NO\(_3\) concentrations in groundwater flowing through the riparian forests or wetlands has been mentioned in many case studies (Mander, 1989; Hill, 1990; Weil et al., 1990; Lowrance, 1992). High denitrification potential in surface soils, especially in the organic/mineral
Fig. 3 Isopleths of the mean annual values (1991-1993) of NO$_3$-N concentration in groundwater top layer and a complex of eco-engineering measures to control nutrient fluxes in the Vända ditch catchment area: 1) buffer zones (extensively used grasslands) around dug wells and along the ditch; 2) forest buffer strips on the ditch bank; 3) macrophyte pond with a stand-pipe (siphon) overflow; 4) planted soil filters with coral-chalk filters for cleaning the groundwater; 5) sites for manure composting to be reconstructed; 6) recommended sites for manure composting with an forest buffer strip and clay base; 7) arable land; 8) forest; 9) village; A) mean concentration of NO$_3$-N (mg l$^{-1}$).

soils tarnisition zone and near the field/grassland or field/forest interface, contributes to NO$_3$ disappearance from shallow groundwater. On the slopes of the Aardla primeval valley where the Porijõgi flows (eastern from the Vända subwatershed, Fig. 3), the groundwater seeping creates such transition zone. Denitrification process in groundwater has been analyzed in various studies. Generally, the microbiologically based denitrification capacity depends on two major factors: oxygen lack and carbon availability in water/substrate (Heyder et al., 1985; Gillham, 1991). Because of limited amounts of organic material in deeper layers (in the Vända ditch drainage basin: $>2$ m) from one side and good oxygenation in the upper layer of mineral soils ($<1.5$ m) from the other, the highest local intensity of denitrification is estimated to be in the depth of 1-3 m. In the organic/mineral soil transition zone and in peatlands, the most intensive denitrification occurs in the soil surface layer. That has been described also by other authors (Heyder et al., 1985). However, the depth of the highest denitrification capacity depends on hydrogeological conditions and may be deeper than in our study area (Gillham, 1991; Kinzelbach, 1991). A significant positive correlation between the mean annual NO$_3$-N concentration and the depth of water table in the draw wells ($R^2=0.13$; 95% confidence level) may be
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Table 4 Correlation coefficients and parameters of linear regression equation between the mean annual nitrate nitrogen concentration (y) and the depth of water table in the dug wells (x) of the Porijõgi River watershed.

<table>
<thead>
<tr>
<th>Subwatershed</th>
<th>Number of wells</th>
<th>Correlation coefficient (r)</th>
<th>Intercept</th>
<th>Slope</th>
</tr>
</thead>
<tbody>
<tr>
<td>Porijõgi upper</td>
<td>12</td>
<td>0.34*</td>
<td>2.42</td>
<td>0.5</td>
</tr>
<tr>
<td>Sipe</td>
<td>7</td>
<td>0.12</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Vânda</td>
<td>21</td>
<td>0.34*</td>
<td>15.2</td>
<td>1.77</td>
</tr>
<tr>
<td>Whole watershed</td>
<td>40</td>
<td>0.36*</td>
<td>7.54</td>
<td>1.66</td>
</tr>
</tbody>
</table>

* P<0.05 (characterizes the significance of correlation)

explained by above described denitrification conditions (Table 4). Only in the Sipe subwatershed, where the depth of wells studied varies only from 2 to 5 m, no significant correlation was found.

In the dug wells of the Vânda subwatershed with buffer zones (nonfertilized perennial grasslands with hedges or tree rows around 3 wells in the western and north-western part) the NO₃ concentration was found lower than in those without buffer zones. One explanation for this phenomenon is the intensive NO₃ uptake by tree roots and less input from the grasslands. Additionally, De Wit & Bleuten (1987) described that because of intensive transpiration capacity, small forest within the matrix of intensively used agricultural fields can function as a special "pump" creating a local groundwater stream directed towards the forested area. Thus, buffer zones around wells could improve the water quality in the wells.

Other nutrients

The pH value was found a most stable parameter being neutral and varying only from 6.5 and 7.9. Seemingly, there are no impacts of agricultural activities on this component. Other parameters depend on the intensity of agricultural use.

Phosphates and total phosphorus contents in the groundwater were found relatively low: 0.04-0.05 and 0.06-0.08 mg l⁻¹, respectively. In phosphorus dynamics, the agricultural practices play the important role than the soil structure. In very heterogeneous soil pattern of Porijõgi Upper subwatershed where the sandy soils dominate, the mean phosphorus concentrations were highest of this region: 0.05 mg PO₄-P and 0.08 mg total-P l⁻¹. It differs from results of some other studies. Usually, there is a significant relationship between the soil structure and PO₄ contents in the groundwater top layer (De Wit & Bleuten, 1987). Because of relatively high fertilization rate (27-39 kg P ha⁻¹ year⁻¹) and low uptake by plants (9-15 kg P ha⁻¹ year⁻¹), the phosphorus accumulated in soil, may be washed out in coming years (Table 2). Probably, the high variation of PO₄-P and total-P concentrations is the first sign of this process (Table 1). Also, a slight decrease of phosphorus concentrations in the wells of the whole catchment area in April 1993 was observed (Fig. 1).

Dynamics of sulphate concentrations in the groundwater depends essentially on land-use practices and on fertilization intensity (Table 1). Nevertheless, as was reported by Toussaint (1989), the SO₄-S concentrations did not exceed the drinking water quality standards (350 mg SO₄ l⁻¹).
There is an urgent need to solve the problem of groundwater quality in rural areas of South Estonia. The very high \( \text{NO}_3 \) contents in dug wells that exceed 2-3 times international quality standards create serious health risk to the whole rural population, especially to children. Because of very high prices and limited finances it is impossible for farmers to use principal conventional solutions like construction of new bore-wells to the uncontaminated deeper groundwater layers or connection to the central water supply system. Some farmers are served weekly by special fresh water tanks, but it can not be a long-term solution because of increasing fuel prices. Thus, to clean up the groundwater in-situ, and to control the nutrient fluxes through landscape, a couple of alternative measures has been proposed for the Vânda subwatershed of the Poriõgi River catchment area (Fig. 3).

Among them, a multilayer planted soil filter worked out at the Ecocenter Schattweid, Switzerland for the municipal wastewater treatment, will be used to eliminate nitrates (Heeb & Züst, 1991). To improve the treated water quality to the level of drinking water an additional chalk filter is recommended. The best absorption capacity in that case has been registered by a coral-chalk filter material produced from the marine algae \textit{Lithothamnium calcareum} (originally growing on the coast of Brittany, France) and distributed by the Swedish company BioOrganics Svenska AB. However, many local limestone sorts could be easily used for that purpose.

To decrease nutrient leaching from the soils into groundwater the best management practices are integrated plant production and various crop-rotation systems from organic and biodynamic agriculture. The last ones are based on systematic cultivation of leguminous plants, and they do not accept any use of synthetic pesticides and fertilizers. The advantages of ecological agriculture from the point of view of nutrient leaching and groundwater quality have been shown in different case studies (Bergström, 1987; Toussaint, 1989; and Granstedt, 1990).

An important mean to prevent nutrient input to the groundwater and water bodies is establishment of buffer zones and buffer strips. The best practical examples for groundwater buffer zones with different land-use and management restrictions are known from Germany and Switzerland (Wohlrab, 1976; Toussaint, 1989; Stadelmann et al., 1991). In the Vânda ditch subwatershed, for every dug well with critical water quality such buffer zones of perennial grasslands are recommended (Fig. 3). The groundwater buffer zones were applied in some European countries already in 1970's (Wohlrab, 1976), but the buffer zones along banks and shorelines of water bodies have been accepted from the end of 1980's only in few areas (Mander, 1989). However, the water protection buffer zones as extensively used perennial grasslands combined with forest buffer strips on the stream banks are able to retain and prevent significant amount of nutrients (Mander, 1989; Lowrance, 1992). So, both perennial grasslands as buffer strips and gray alder buffer strips have been proposed for the experimental watershed (Fig. 3). A part of the alder forest buffer-strips has already been planted in 1990-1991 (Mander, 1991).

Similar prevention practices have been foreseen for two big manure composting sites (total area 0.7 ha) that are at present the most important point pollution sources of the Vânda ditch subwatershed. The sites lie relatively close to the stream without any base to prevent the leaching of polluted water into the
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groundwater. Manure piles with no concrete base even when treated biodynamically still give little loss of nutrients into the aquifer (Wistinghausen, 1989). That is why the new recommended manure composting sites have been designed with a clay base and forest buffer strips (Fig. 3).

To retain and transform nutrients in the eutrophicated Vända ditch, a cascade of macrophyte ponds with stand-pipe (siphon) overflow has been designed (Mander et al., 1991).

CONCLUSIONS

The following conclusions have been drawn:
(a) The most critical component in the groundwater of South Estonian agricultural areas is nitrate nitrogen. The mean annual concentration of NO$_3$-N in the intensively used agricultural fields was found 19 mg l$^{-1}$, i.e., twice higher than EC quality standard. Maximal values of NO$_3$-N contents reach 51 mg l$^{-1}$.
(b) A highly significant correlation ($R^2=0.78$) between the fertilization intensity and NO$_3$-N concentration in dug wells was found there.
(c) Higher NO$_3$-N concentrations were found in deeper wells ($R^2=0.13$) and in those located at the parts of the watershed.
(d) In shallow dug wells with nonfertilized buffer zones (perennial grassland with hedges or tree rows of $<100$ m radius around wells), the NO$_3$-N content was lower than in those without buffer zones.
(e) In the groundwater of intensively used fields, the significant increase of NO$_3$-N contents in late autumn / winter occurs. No seasonal dynamics of NO$_3$ concentrations, nor of other nutrients was found in natural and/or extensively used areas.
(f) The values of NH$_4$ and SO$_4$ contents in shallow groundwater were significantly below the limits of quality standards, but had a positive correlation with the fertilization intensity. In the groundwater of sandy areas, the PO$_4$ contents were higher than that under loamy soils.
(g) Despite of slight decrease in fertilization intensity during last three years no essential improvement of groundwater quality was observed.
(h) A complex of alternative measures has been proposed to improve the groundwater quality and to prevent nutrient losses from the drainage basin of a small stream.

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