Trends in nutrient concentrations in Latvian rivers and the response to the dramatic change in agriculture

P. Stålnacke, A. Grimvall, C. Libiseller, M. Laznik, I. Kokorite

Abstract

In recent years, the use of fertilisers in the Baltic countries (Estonia, Latvia, and Lithuania) has decreased at an unprecedented rate. The import of mineral fertilisers and feed stuff became almost non-existent, and extensive slaughtering of livestock reduced the amount of manure. In Latvia, the purchase of mineral fertilisers decreased by a factor of 15 between 1987 and 1996 and the number of livestock decreased with a factor of almost 4 during the same time period. Such abrupt and comprehensive changes in land use have never before occurred in the history of modern European agriculture.

Here, the impact that this dramatic reduction has had on concentrations of nutrients in Latvian rivers is examined. To discern temporal changes, statistical analyses were undertaken on time series of nutrient concentrations and relationships between concentrations and runoff at 12 sampling sites in ten Latvian rivers covering drainage areas from 334 to 64,000 km².

Considering the study period 1987–1998, only four of the 12 sites showed statistically significant downward trends (one-sided test at the 5% level) in the dissolved inorganic nitrogen (DIN = NO₃-N + NO₂-N + NH₄-N) data. There are probably two main explanations for the weak DIN trends. Firstly, long water-transit times in the soilwater and groundwater may have caused substantial time lag between changes in input and output of nitrate in the studied catchments. Secondly, the loss of DIN might have been dominated by mineralisation of large pools of organic nitrogen that have accumulated over several years. These inferences are supported by (i) a hydrograph recession analysis and (ii) indications of DIN transformation processes, presumably denitrification, in smaller streams and channels, based on measurements in small agricultural catchments (1–4 km²) in Estonia and Latvia.

Formal testing of trends in phosphorus data revealed that marked drops occurred in riverine concentrations at six sites in 1987–1998. A joint analysis of concentration time series for all sampling sites for 1987–1998 showed weak statistical significance for downward trends in NH₄-N, NO₃-N, and DIN (p ≈ 0.04) and substantial significance for PO₄-P (p < 0.01).

Thus, the extensive decrease in agricultural intensity that began in the early 1990s has led to only a slow and limited (especially regarding nitrogen) response in Latvian rivers. The difference noted between nitrogen and phosphorus also suggests that factors other than reduced fertiliser application influenced the inertia of the water quality response.

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Key words: nutrient concentrations; agricultural change; Latvia; rivers; statistical analysis; runoff.
Our findings, along with those obtained in similar studies, show that large cuts in nutrient inputs do not necessarily cause an immediate response, particularly in medium-sized and large catchment areas.

**Keywords:** Nitrogen; Phosphorus; Agricultural rivers; Trend analysis; Agricultural change; Latvia

1. Introduction

Large losses of nutrients from agricultural soils are often caused by intensive use of fertilisers, especially in situations when fertiliser use exceeds the nutrient requirements of the crops. This has been clearly demonstrated in plot experiments (e.g. Brink, 1983; Andersson, 1986; Neill, 1989; Liang et al., 1991), and catchment studies (Dillon and Kirchner, 1975; Hill, 1981; Rekolainen, 1989; Zákóvá et al., 1993). There is a substantial amount of data suggesting that agriculture or specific farming practices can have a marked impact on the output of nutrients (particularly nitrate) from entire river basins. In several investigations (e.g. Sharpley and Withers, 1994; Hill, 1978; Kauppi, 1978; Hill, 1981; Fleischer and Hamrin, 1988; Loigu, 1989; Rekolainen, 1989; Grimvall and Stålnacke, 1996), it has been found that concentrations of nutrients in river water were strongly correlated to the percentage of agricultural land in the studied basins. Moreover, examination of long time series of nutrient concentrations in European rivers has revealed upward trends in nitrogen levels, which coincide with the increased use of nitrogen fertilisers and intensification of animal production, whereas temporal trends in phosphorus (e.g. PO₄-P) seem to be associated to a lesser degree with agriculture (Schröder, 1985; Behrendt, 1988; Behrendt and Böhme, 1993; Grimvall, 1990; Tsirkunov et al., 1992; Zákóvá et al., 1993; Procházková et al., 1996; Grimvall et al., 2000; Tumas, 2000). However, Tomlinson (1970) examined 15-year-long water quality records in England and discovered upward nitrate trends in only six of 18 rivers, despite a substantial increase in fertiliser application during the studied time period. Thomas et al. (1992) investigated seven small watersheds in Kentucky (USA) and observed that doubling the rates of fertiliser application after 18 years did not change the nutrient loads in the waters. Keeney and DeLuca (1993) scrutinised records of nitrate concentrations in a river in Ohio (USA) and reported that the annual load of nitrate had changed very little from 1945 to 1980, and that the temporal trends that were revealed seemed to be related to agricultural activities in general rather than the application of fertilisers in particular. An analysis of nitrogen loadings to Lough Erne in north-west Ireland showed no temporal trend in the inorganic nitrogen inputs over a 25-year period, despite increased loadings from diffuse agricultural sources (Zhou et al., 2000). Similarly, Stow et al. (2001) recently reported that they were surprised over the lack of correlation between substantial increases in the non-point source loadings and loads of nitrogen in the mouth of the Neuse River in North Carolina.

It is noteworthy that all of the studies cited thus far have dealt with the possible impact of increased emissions from agricultural sources on riverine loads or concentrations. Only a few river-basin-scale studies have been conducted to elucidate the influence of rapid land-use changes and decreases in fertiliser application. Pekarova and Pekar (1996) reported that nutrient concentrations in surface waters in the Ondava River (1089 km²) in Slovakia have decreased after a general and substantial reduction in the use of fertilisers. Furthermore, Mander et al. (2000) found downward trends in one small agricultural catchment in Estonia (258 km²), and considerable decreases in nutrient concentrations in river waters have also been observed in the large Tisza River (157,000 km²) in Hungary (Oláh and Oláh, 1996) and in the Elbe River (Hussian et al., 2003). In contrast, no downward trends were found by Procházková and co-workers (1996) in the Vltava River (12,900 km²) or by Berankova and Ungerman (1996) in the Morava River in the Czech Republic, or by Tonderski (1997) in the Vistula (194,000 km²) and Oder (108,000 km²) Rivers in Poland in 1989–1995, despite dramatic changes in agriculture in the mentioned countries.
Likewise, Tumas (2000) noted only weak downward trends in Lithuanian rivers (1130 km²), regardless of a decrease in fertiliser application equivalent to the reduction in Latvia. Furthermore, the same tendency has been documented in case studies performed in large river basins in North America (Hetling et al., 1998).

In the former Soviet Union, Latvian agriculture was specialised in meat and milk products, and considerable import of mineral fertilisers and animal feed was a dominant aspect of the production system (Ozolins, 1994). When the Baltic countries became independent in 1991, there was a sudden decrease in demand for Latvian farm products, which led to rapid restructuring of Latvian agriculture. The import of mineral fertilisers and feedstuff became almost non-existent, and extensive slaughtering of livestock reduced the amount of manure. Since that time, application of both commercial fertilisers and manure has declined at an unprecedented rate. The data presented in Table 1 show that, from 1987 to 1996, the purchase of mineral fertilisers decreased by a factor of 15, although the amounts of fertilisers that were actually spread on the fields may have changed more gradually. The table also shows that the number of livestock decreased by a factor of almost 4 from 1987 to 1998. Such abrupt and comprehensive changes in land use have never before occurred in the history of modern European agriculture.

This raises the question whether the dramatic reduction in the use of fertilisers over the past few years has caused downward nutrient trends in receiving waters. In the present study, time series of nutrient concentrations measured in rivers located in catchment areas comprising substantial proportions of agricultural land are statistically evaluated. Considering that natural variation in runoff can have an extensive impact on observed concentrations and riverine loads, we paid special attention to possible temporal changes in concentration–runoff relationships.

2. Study area, database and data handling

2.1. Study area

Five agriculturally dominated river basins, with a total drainage area of more than 120,000 km², were statistically analysed for nutrient trends (Table 2). The largest of these is the catchment of the Daugava River (including the tributaries Aiviekste and Dubna), which comprises 84,000 km² at the water quality monitoring site farthest downstream (No. 1; Fig. 1) and includes territory in five different countries: Russia (21%), Belarus (38%), Latvia (28%), and Lithuania and Estonia (13% together). The other four river basins are located almost entirely in Latvia (95% of the area), and they surround the Lielupe (including the tributary Berze), Gauja, Salaca, and Venta Rivers.

The climate in the study area is humid and moderately mild due to the impact of Atlantic air masses, and differs little between the investigated catchments. The only exception to this is the upper part of the Daugava basin (in Russia and Belarus), where the climatic conditions are slightly more continental in character. The annual mean temperature in the area is approximately 6 °C, with a monthly minimum mean of −9 °C in January and maximum mean of +17 °C in July (Jansons et al., 2002). The vegetation period (air temperature > +5 °C) begins on about 15 April and lasts 180–200 days. There is a west-to-east gradient in annual precipitation: from

<table>
<thead>
<tr>
<th>Year</th>
<th>N fertiliser consumption (kg/ha arable land)</th>
<th>P fertiliser consumption (kg/ha arable land)</th>
<th>Livestock (No. of heads)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1987</td>
<td>127</td>
<td>30.4</td>
<td>1,616,100</td>
</tr>
<tr>
<td>1988</td>
<td>117</td>
<td>27.8</td>
<td>1,589,900</td>
</tr>
<tr>
<td>1989</td>
<td>83</td>
<td>28.6</td>
<td>1,597,800</td>
</tr>
<tr>
<td>1990</td>
<td>68</td>
<td>26.6</td>
<td>1,556,950</td>
</tr>
<tr>
<td>1991</td>
<td>66</td>
<td>10.0</td>
<td>1,494,300</td>
</tr>
<tr>
<td>1992</td>
<td>39</td>
<td>7.5</td>
<td>1,231,900</td>
</tr>
<tr>
<td>1993</td>
<td>23</td>
<td>2.1</td>
<td>736,000</td>
</tr>
<tr>
<td>1994</td>
<td>17</td>
<td>1.3</td>
<td>605,000</td>
</tr>
<tr>
<td>1995</td>
<td>7</td>
<td>0.5</td>
<td>592,600</td>
</tr>
<tr>
<td>1996</td>
<td>8</td>
<td>1.5</td>
<td>557,700</td>
</tr>
<tr>
<td>1997</td>
<td>N.D.</td>
<td>N.D.</td>
<td>520,700</td>
</tr>
<tr>
<td>1998</td>
<td>N.D.</td>
<td>N.D.</td>
<td>475,000</td>
</tr>
</tbody>
</table>

Sources: FAO statistics and Statistics Latvia (various years). N.D., no data available.
900 mm near the coast to less than 600 mm in the eastern parts of the study area in Belarus and Russia. Annual rainfall in Latvia is generally about 700 mm and exceeds evaporation (usually approx. 450 mm), which results in water losses through percolation that are particularly pronounced during spring and autumn (Jansons et al., 2002). The precipitation during winter normally forms a stable snow cover that lasts

The numbers used to designate the sampling sites refer to those indicated in the map in Fig. 1. Sources: LHMA (areas) and CORINE (land cover).

Table 2
Investigated rivers and sampling sites, and areas of associated catchments

<table>
<thead>
<tr>
<th>Sampling site no.</th>
<th>River</th>
<th>Location of sampling site</th>
<th>Drainage area (km²)</th>
<th>Agricultural land (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Daugava</td>
<td>Lipshi (mouth)</td>
<td>84,100</td>
<td>52</td>
</tr>
<tr>
<td>2</td>
<td>Daugava</td>
<td>Daugavpils</td>
<td>64,600</td>
<td>58</td>
</tr>
<tr>
<td>3</td>
<td>Aiviekste</td>
<td>Mouth</td>
<td>9160</td>
<td>48</td>
</tr>
<tr>
<td>4</td>
<td>Dubna</td>
<td>Mouth</td>
<td>2780</td>
<td>57</td>
</tr>
<tr>
<td>5</td>
<td>Lielupe</td>
<td>Kalnciems</td>
<td>16,500</td>
<td>46</td>
</tr>
<tr>
<td>6</td>
<td>Tulja</td>
<td>Zoseni</td>
<td>334</td>
<td>29</td>
</tr>
<tr>
<td>7</td>
<td>Iecava</td>
<td>Mouth</td>
<td>1010</td>
<td>39</td>
</tr>
<tr>
<td>8</td>
<td>Lielupe</td>
<td>Jelgava</td>
<td>12,000</td>
<td>63</td>
</tr>
<tr>
<td>9</td>
<td>Berze</td>
<td>Dobele</td>
<td>723</td>
<td>74</td>
</tr>
<tr>
<td>10</td>
<td>Gauja</td>
<td>Carnicava (mouth)</td>
<td>8900</td>
<td>38</td>
</tr>
<tr>
<td>11</td>
<td>Salaca</td>
<td>Salaca (mouth)</td>
<td>3570</td>
<td>37</td>
</tr>
<tr>
<td>12</td>
<td>Venta</td>
<td>Kuldiga</td>
<td>8320</td>
<td>48</td>
</tr>
</tbody>
</table>

The numbers used to designate the sampling sites refer to those indicated in the map in Fig. 1. Sources: LHMA (areas) and CORINE (land cover).

Fig. 1. Map showing the twelve investigated sampling sites.
145–150 days (Jansons et al., 2002). Summer (June–August) is the dampest season, with an average of 300 mm of rain. The excess precipitation generally results in annual runoff of approximately 250 mm, hence drainage is an important precondition for agricultural operations and rational farming. More precisely, artificial drainage is done to extend the vegetation period by allowing an up to two-week earlier start of the period of soil preparation and sowing in the spring, and also facilitating the harvesting of crops in late summer and autumn. In Latvia, 60% of the total area of agricultural land (i.e. 1.6 million ha) is artificially drained (Jansons, personal communication), achieved to 95% by use of subsurface tile drains.

The political and subsequent economical changes in the early 1990s in the studied countries have had a profound affect on the entire agricultural sector. In addition to the decrease in use of fertilisers (Table 1), the production of meat, milk, and eggs in Latvia declined by 40–50% from 1989 to 1994 (Löfgren et al., 1999), and an equivalent reduction occurred in the Russian part of the Daugava basin (Lääne et al., 2002).

According to the national soil classification system used in Latvia, the main soil types in the country are gleyic clay loam and sandy clay loam, both of which are relatively poor in natural fertility, and more than 50% of the soils are podzolic (Jansons, personal communication). Consequently, the mean content of organic matter in Latvian soils is relatively low (i.e. 1.8% organic C) and has decreased in 24% of the surveyed area (Latvian Environment Agency, 2002), most likely due to the decreased use of manure. Furthermore, 23% of the soils are acid (pH < 5.6) and require liming, but the rate of liming has also dropped drastically: the area treated in 1995 was only 2% of that treated in 1990 (UN-ECE, 1999). Some areas consist of fertile soddy carbonate-rich soils, which are characterised by a high pH (7.5–7.9) and a substantial content of organic matter. Less than 7% of the soils are productive luvisols (Zemgale) (UN-ECE, 1999). Essentially the same soil characteristics prevail in Belarus, with the addition of peaty soils in some areas (Fejes et al., 1997). In one of the few published articles concerning the content of major plant nutrients in top soil in our study area, Jansons et al. (2002) recently found that the levels in Latvia are fairly good: P–Al ranged from 8 to 11 mg per 100 g and N from 0.08 to 0.15%.

Considering lithotectonic classification, the study area is part of the Eastern European Platform (Reimann et al., 2000). Throughout Latvia, Quaternary deposits generally cover Devonian sediments to form what is known as a sub-Quaternary surface (Atlas of Geography of Latvia, 1999; http://www.geology.wmich.edu/santis/gas/regional.html). Paleozoic rocks such as sandstone, limestone, and dolomite cover most of the territory (Reimann et al., 2000), and crystalline rocks are found in very deep layers. There are some regional differences, for instance the presence of Permian and Carboniferous limestone and dolomite bedrock in the southwest. The thickness of the quaternary sediment layer varies from a few metres to 300 m in the north-eastern part of the country.

The three large hydropower dams (Kegums, Riga, and Plavinas) located in the lower reaches of the Daugava River constitute an important feature of the study area. The total capacity of all three dam reservoirs together is 1.01 km³, with a surface area of 101.9 km². The Kegums Reservoir has an area of 24.8 km² and its full capacity is 0.16 km³; this reservoir is situated farthest upstream of the three dams, in the vicinity of the Aiviekste tributary (sampling site No. 3; Fig. 1). The Riga Reservoir has an area of 42.2 km² and a full capacity of 0.34 km³, and is located immediately upstream of the most downward sampling site on the Daugava River (site No. 1; Fig. 1). The Plavinas Reservoir has an area of 34.9 km² and a full capacity of 0.51 km³, and it is situated between the other two reservoirs.

2.2. Sampling sites and water quality data

Time series of nitrate (NO₃-N), ammonia (NH₄-N), dissolved inorganic nitrogen (DIN; sum of NO₃-N, NO₂-N, and NH₄-N), phosphate (PO₄-P), and total phosphorus (TOT-P) concentrations and data on runoff for major Latvian rivers were examined based on data from the Latvian Hydrometeorological Agency. NO₃-N was not statistically analysed for trends due to very low concentrations (normally less than 0.015 mg l⁻¹; Table 3). The data covered the
period 1987–1998 and were based on 12 monitoring stations in ten rivers (see Table 2 and Fig. 1) to represent both small and large catchments with substantial areas of agricultural land. One of the sampling sites (No. 6 in Fig. 1) was selected to represent a catchment with relatively less intensive agriculture.

In general, monthly time series of concentration values were available for the entire period 1987–1998, although approximately bimonthly sampling was introduced in 1995 at four of the studied sites (Nos. 3, 4, 7, and 9 in Fig. 1). TOT-P data were available for only six of the 12 sites. The runoff data were recorded on the sampling occasions.

During the studied period, PO4-P concentrations in the samples were determined according to a photometric procedure in which ammonium molybdate, ascorbic acid, and potassium antimonide are added to the sample. More precisely, TOT-P was determined by digestion with peroxydisulphate (APHA METHOD 4500-P B.5), and PO4-P was assayed by the molybdenum blue method (APHA METHOD 4500-P E). The concentration of nitrate (NO3-N) was determined by first reducing nitrate to nitrite and then applying a photometric procedure using Griess reagents (APHA METHOD 4500-NO3.E). The concentration of NH4-N was measured by the indophenol blue method (APHA METHOD 4500-NH3.F), and ISO 6777:1984 standard was used for determination of NO2-N.

Samples from sites 2 and 4 were analysed at a district laboratory in Daugavpils until 1997. All other samples were analysed at a national laboratory in Riga, which has been accredited by the Latvian National Accreditation Body since 1998. From 1992 to 1994, water samples were also transported to the Department of Water and Environmental Studies at Linköping University for chemical analysis of nutrients. These intercalibrations did not discern any major discrepancies in NO3-N or PO4-P concentrations.

3. Statistical methods

3.1. A univariate test for temporal trends in riverine concentrations

A variety of statistical procedures have been developed to distinguish between random fluctuations and more persistent temporal changes in environmental quality. Relatively recently, Esterby (1996) reviewed both univariate and multivariate procedures, and several trend tests have also been described in statistical textbooks (e.g. Gilbert, 1987; Helsel and Hirsch, 1992). A modified version of the Mann–Kendall test was in our study used to analyse long-term changes in nutrient concentrations. The Mann–Kendall test is a non-parametric tool used to detect monotone trends in time series. In this case it had been adapted to account for
seasonal variation (Hirsch et al., 1982; Hirsch and Slack, 1984).

For a time series observed over \( p \) seasons during \( n \) years, the Mann–Kendall statistic \( S_g \) for season \( g \) is defined as the sum of all signs of differences

\[
S_g = \sum_{i<j} \text{sgn}(X_{ig} - X_{jg}),
\]

where

\[
\text{sgn}(y) = \begin{cases} 
1, & \text{if } y > 0 \\
0, & \text{if } y = 0 \\
-1, & \text{if } y < 0
\end{cases}
\]

\( S_g \) is under the null hypothesis of no trend asymptotically normally distributed with mean 0 and variance \( \sigma_{gg} = n(n-1)(2n+5)/18 \).

The Hirsch–Slack test (HS) is subsequently defined as the sum of the Mann–Kendall statistics for all seasons

\[
S = \sum_g S_g, \quad g = 1, \ldots, p
\]

and is asymptotically normally distributed with mean 0 and variance

\[
\text{Var}(S) = \sum_g \sigma_{gg} + \sum_{g<h} \sigma_{gh}, \quad g, h = 1, \ldots, p
\]

where \( \sigma_{gh} \) denotes the covariance between the Mann–Kendall test statistics for seasons \( g \) and \( h \). A consistent estimator \( \hat{\sigma}_{gh} \) of \( \sigma_{gh} \) has been derived by Dietz and Killeen (1981).

Corrections of these test statistics in the presence of missing values or ties has been described by Hirsch and Slack (1984).

### 3.2. A test for trend with correction for influencing variables

Prevailing weather conditions often cause natural fluctuations in the nutrient concentration time series, which may impede detection of an existing trend (Stålnacke and Grimvall, 2001). To account for such fluctuations, it is essential to include explanatory (i.e. meteorological or hydrological) variables in the analysis. Here, this is accomplished by computing the conditional distribution of the Hirsch–Slack statistic \( S'_g \) for the response variable, given the statistic \( S'_q \) for the explanatory variable (Libiseller and Grimvall, 2001). This test is referred to as the Partial Hirsch–Slack (PHS) test. In the case of one response and one explanatory variable, this distribution is asymptotically normal with expectation

\[
E(S'_r|S'_q = s) = \frac{\sigma_{rq}^s}{\sigma_{qq}}
\]

and variance

\[
\text{Var}(S'_r|S'_q = s) = \hat{\sigma}_{rr} - \left( \frac{\hat{\sigma}_{rq}}{\hat{\sigma}_{qq}} \right)^2 \hat{\sigma}_{qq}
\]

where \( \hat{\sigma}_{rr} \) and \( \hat{\sigma}_{qq} \) denote the variance of \( S'_r \) and \( S'_q \), respectively, and \( \hat{\sigma}_{rq} \) is the covariance between the test statistic of the response variable and the covariate.

This procedure can be generalised to allow for more than one explanatory variable. We used water discharge as such a variable in our study. The PHS statistic has been shown to be powerful in situations with monthly and quarterly data, provided there are at least ten years of data (Libiseller and Grimvall, 2002).

### 3.3. Testing for trend simultaneously at several monitoring sites

Further testing for trends can be conducted at several sampling sites simultaneously (Loftis et al., 1991). Provided that the response and the explanatory variable have similar dependence structure at all sites existing trends are then more likely to be detected. The procedure for testing at more than one site is identical to that for including several seasons. The Hirsch–Slack statistics are first computed separately for the different sites and are then summed to one joint statistic (JHS). Likewise, the variance is computed as the sum of all variances and covariances of the test statistics for the different sites. If it is necessary to correct for an influencing variable, the JHS statistic for the response variable can be conditioned on the JHS of the explanatory variable, using the PHS technique described above (JPHS). This procedure
is applicable when time series at several sites have similar trend patterns, in which case it will more reliably determine the trend in a larger area, for example an entire catchment.

4. Results

4.1. Exploratory analysis of mean concentrations

The median water discharge values and nutrient concentrations are given in Table 3. The highest nutrient concentrations occurred in the Lielupe River basin, which is the most intensively farmed area in Latvia (Laznik et al., 1999). This is exemplified by the NO$_3$-N and DIN concentrations in the basin, which were >1.70 mg NO$_3$-N l$^{-1}$ and >2 mg DIN l$^{-1}$, respectively. In addition, the highest concentrations of PO$_4$-P (0.069 mg l$^{-1}$) and total phosphorus (0.082 mg l$^{-1}$) were found in the Lielupe River basin. Relatively high levels of phosphorous were also noted at Daugavpils on the Daugava River (0.058 mg l$^{-1}$ PO$_4$-P and 0.070 mg l$^{-1}$ TOT-P). The lowest concentrations of both nitrogen and phosphorus were found in the Salaca, Gauja, and Tulija Rivers.

4.2. Exploratory analysis of time series of water discharge and nutrient concentration data

Time series of monthly water discharge data for Latvian rivers during the period 1987–1998 are presented in Fig. 2. A characteristic feature that can be discerned is that the highest water discharge generally occurred in spring. However, the rivers in the western part of Latvia, such as the Venta, also exhibited relatively substantial water discharge during autumn. The time series do not show clear long-term trends in water discharge, with the exception of tendencies towards elevated water discharge in the Lielupe and Venta Rivers in the late 1980s.

The time series of monthly concentrations of DIN (Fig. 3) indicated downward trends at only a few sites (e.g. in the Lielupe River), and decreased concentrations can be seen starting in the early 1990s at the site furthest downstream on the Daugava River (Lipshi). Time series show pronounced seasonal fluctuations in DIN, with the lowest levels in the summer period, and the highest concentrations occurring from late autumn to spring.

Time series of monthly concentrations of PO$_4$-P are illustrated in Fig. 4, and they show more pronounced downward trends, for example in the Daugava, Lielupe, Iecava, and Venta Rivers. Seasonal patterns are not as clear for phosphates as for nitrogen.

4.3. Exploratory analysis of relationships between nutrient concentrations and runoff

Fig. 5 illustrates typical relationships between monthly concentrations of nutrients and water discharge. Greater correlation is apparent between nitrates and water discharge than between PO$_4$-P and water discharge. For nitrates, the explained variation $R^2$ varied from 0.45 for the Salaca River to 0.07 for the Daugava River. The $R^2$ values for the Berze, Venta, Gauja, and Dubna Rivers were relatively high (> 0.30). The correlation between phosphates and water discharge was weak: $R^2$ varied from 0.19 for the Venta River to 0.0004 for the Daugava River. Even though the correlation between nutrient concentrations and water discharge was sometimes weak, this simple exercise clearly justified that water discharge must be taken into account when analysing trends in nutrient concentrations.

4.4. Tests of significance of temporal trends

Table 4 summarises the results obtained using the HS and PHS methods for trend analysis of the period 1987–1998. The HS test revealed statistically significant ($p < 0.05$) downward trends in water discharge in four cases. Analysis of temporal changes in NH$_4$-N revealed only three statistically significant trends, two of which were the upward trends detected in the Daugava (Lipshi sampling station) and the Tulija River. The PHS test revealed only one statistically significant upward trend, in the Tulija River. A downward trend was found in the Iecava River, which is located in the Lielupe River basin. The results for NO$_3$-N indicated significant downward trends ($p < 0.05$) at four of the 12 monitoring stations: the site on the Venta
Fig. 2. Time series of monthly water discharge in Latvian rivers, 1987–1998.
Fig. 3. Time series of monthly concentrations of inorganic N (DIN) in Latvian rivers, 1987–1998.
Fig. 4. Time series of monthly concentrations of PO₄-P in Latvian rivers, 1987–1998.
River, both sites on the Lielupe River, and the site (Lipshi) on the Daugava River. The PHS test detected three statistically significant downward trends, in the Lielupe, the Iecava, and the Venta River, as well as one upward trend at Daugavpils on the Daugava River. Considering DIN, we found four significant trends, all of which were negative.

For PO₄-P, the results of the HS test were statistically significant in six cases (see Table 4), all showing downward trends; the greatest significance

Table 4
Tests of significance of temporal changes in data on water discharge and nutrient levels in 1987–1998

<table>
<thead>
<tr>
<th>Station</th>
<th>Q</th>
<th>NH₄-N</th>
<th>NO₃-N</th>
<th>DIN</th>
<th>PO₄-P</th>
<th>TOT-P</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>HS</td>
<td>PHS</td>
<td>HS</td>
<td>PHS</td>
<td>HS</td>
<td>PHS</td>
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Results that are statistically significant at the 5% level (one-sided test) are highlighted in bold-face type. HS, Hirsch–Slack test; PHS, Partial Hirsch–Slack test.
was found for the Daugava and Lielupe basins. The PHS test confirmed these findings: analysis of the total phosphorus trend revealed a significant trend in two of the six tests, one in the Daugava River and the other in the Lielupe River.

4.5. Joint analysis of concentration time series for all sampling sites

Inasmuch as the rivers are situated close to each other and are therefore probably subject to the same changes in the environment, we also applied the trend test simultaneously to all sampling sites during the period 1987–1998. The Tulja River was excluded from this analysis due to the limited number of observations on discharge that were available for that sampling station, and because, in contrast to the other rivers, it represented a catchment with little agricultural activity and would therefore have impaired the outcome.

Table 5 shows the results of the joint trend analysis. The joint Hirsch–Slack (JHS) test indicated a negative trend for both NO₃-N and DIN. Correcting these test statistics for the influence of discharge lowered the test statistics (JPHS), which nevertheless remained significant ($p \leq 0.04$). A clear negative trend was revealed for PO₄-P even after the correction ($p < 0.01$; JPHS). The same could be expected for TOT-P, but the obtained results were ambiguous, since observations were missing for five of the 11 sites.

In the case of ammonia, it seems that the persistent trend in discharge had concealed an existing negative trend. Due to the high negative correlation between discharge and NH₄-N, this trend was revealed only after the test statistics were corrected for the covariate (JPHS).

5. Discussion

It is essential that we know how long it can take to detect the response of a river system to changes in agriculture, because such information is needed to allow environmental authorities and decision and policy makers to establish realistic goals (Stow et al., 2001). In 1988, the Ministers of Environment of the Baltic countries declared that by 1995 the nutrient loads to the Baltic Sea should be reduced by 50% from the levels that prevailed in 1987 (Helcom, 1988). A recent evaluation of the extent to which that objective had been achieved indicated that the gross loads from agriculture (i.e. emissions at sources) had decreased by more than 50% between the late 1980s and 1995 in both Latvia and Russia (Lääne et al., 2002). The results of our study clearly show that these reductions are not yet appreciably detectable in the river systems. This applies in particular to nitrogen: only eight out of a total of 36 tests for the various nitrogen compounds gave statistically significant evidence of downward trends at the 5% level, and only three of the eight downward trends represented nitrate (Table 4). The relatively limited number of downward trends in nitrogen in our investigation is somewhat remarkable, considering the highly dramatic changes in agricultural practices and the prominent role that farming has played in the catchments we examined.

There are several explanations for the weak or absent nutrient trends. It is widely accepted that the magnitude and temporal variability in nutrient concentrations in agriculturally dominated rivers are regulated by, and dependent on, a number of factors, including the following: (1) the soil nutrient pool and farming practices such as cropping systems, long-term fertilisation intensity, and soil cultivation techniques; (2) hydrological pathways and their influence on retention; (3) temporal variability in flow conditions; (4) in-stream, riverine, and lake retention. The effects of these factors, as well as the influence of other sources, are discussed below in relation to the temporal nutrient trends discerned in our study.

<table>
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<tr>
<td>TOT-P</td>
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</table>

JHS, Joint Hirsch–Slack test; JPHS, Joint Partial Hirsch–Slack tests for trend detection in the presence of covariates.

\* Analysis based on only six rivers (HS for Q: −2.1).
5.1. Soil nutrient pools, mineralisation, and farming practices

In the Baltic countries (Estonia, Latvia, and Lithuania), there was a two- to four-fold increase in the use of commercial fertilisers from 1960 to 1980, and a peak was reached in 1987 (Tsirkunov et al., 1992). During the same time period, there was also a substantial increase in the use of manure due to the establishment of large farms focused on milk and meat production. Examination of water quality data for Latvian rivers has indicated that there was a marked increase in the concentrations of nutrients in the 1960s and a somewhat less pronounced rise in the 1970s and early 1980s (Tsirkunov et al., 1992; Matisone and Laznik, 1994). Compared to the Nordic countries, at the end of the 1980s the average use of fertilisers in the Baltic countries had reached the same level or was higher, while crop yields were only about half (Löfgren et al., 1999). This means that nutrient utilisation was relatively inefficient, clearly indicating a high potential for losses to air and water or accumulation of nutrients in the soil. Estimation of nutrient surpluses on a national level in Latvia has shown that the nitrogen and phosphorus excesses were 57 and 29 kg ha\(^{-1}\), respectively, in 1989, and these values dropped to below zero as early as 1994 (Löfgren et al., 1999). This dramatic change was due mainly to the limited use of mineral fertilisers, as exemplified by the reduction in nitrogen application from almost 130 kg ha\(^{-1}\) in 1987 to less than 10 kg ha\(^{-1}\) in 1995 (Table 1). This is further supported by soil-surface balances for selected agricultural fields in Latvia, which were calculated as the difference between nutrients applied in fertilisers (commercial and manure) and nutrients removed in crop yields; such computations indicated deficits of up to 40 kg N ha\(^{-1}\) and 15 kg P ha\(^{-1}\) for the period spanning from the beginning of the 1990s to 1996 (Vagstad et al., 2000). These values can be compared with the annual mineralisation of 50–200 kg N ha\(^{-1}\) in the soil N pool (Löfgren et al., 1999). Thus it is likely that the loss of nitrogen was regulated primarily by mineralisation of large pools of organic nitrogen and soil organic matter that have accumulated over several years prior to the dramatic change in agricultural input, and this may largely explain the weak downward nitrogen trends reported in our study. The great potential of the soil to deliver mineralised nitrogen from the pool of organic N is further implied by the results of field experiments performed in Sweden (Gustafson et al., 1998), and this is strengthened by the study conducted by Vagstad et al. (2000), which focused on nutrient dynamics in small agricultural catchments in Latvia and showed that soil N mineralisation—not the applied fertilisers—was the main source of leached N. In the Netherlands, Oenema and Roest (1998) proposed that predicted changes in the mineralisation and immobilisation of nitrogen in soils would delay and disproportionately reduce the impact of a reduction in the N surplus. Furthermore, these investigators foresaw that, despite an 82% reduction in the P surplus, no reduction in P emissions would take place within 13–15 years, partly because soil P would continue to increase, albeit at a much slower rate. The importance of accumulation of phosphorous in soils and the influence of this on long-term P losses has been stressed in other studies as well (e.g. Sharpley and Withers, 1994; Foy and O’Connor, 2002). Moreover, an assessment done in the Rhine and Elbe basins indicated that changes in the N and P soil surface balance are not immediately reflected by changes in N and P emissions to the surface waters, at least not within a five-year time frame (De Wit and Behrendt, 1999). A study of ten agricultural fields in Sweden has shown that zero and negative P-balance estimates for the soils over a ten-year period did not reduce the transport of phosphorous in the rivers (Uleñ et al., 2001).

The economic recession in the Baltic states and Russia also led to a substantial abandonment of agricultural land. For example, the area of unsown agricultural land in Latvia increased from 37% in 1989 to more than 50% in 1994 (Löfgren et al., 1999), and in the Russian part of Daugava basin, the amount of arable land has decreased by 9% (Lääne et al., 2002). It appears that this trend is continuing, and in 2001 a half million hectares of agricultural land was in fallow in Latvia, due partly also to changes in land ownership (Jansons, personal communication). The tremendous potential for nitrogen mineralisation in unfertilised bare fallow is also well documented. A study carried out at Rothamsted (UK) showed that it took almost 50 years for nitrate losses to diminish by 50% (Addiscott, 1988). It is assumed that phosphorus
losses in the Baltic Sea region are influenced to a lesser degree by mineralisation and are instead due mainly to erosion of particles on and in the soil profile (e.g. Rekolainen and Leek, 1996). Soil erosion is affected primarily by soil tillage and the cropping system: losses of soil are lower from unploughed or grass-covered land than from scarified ground or fields covered with winter grain (Sibbesen et al., 1994), even though freezing of plant residues on fertilised grasslands or catch crops can lead to substantial surface losses of P (Uhlen, 1989; Bechmann et al., 2003). In our study area, the decrease in cereal crop land accompanied by an increase in sown grass lands and unsown-unploughed land (Löfgren et al., 1999) may partly explain the observation that the P concentrations in rivers declined more rapidly than the N levels. In addition, some parts of the unsown land had no covering remnant crops and were therefore categorised as essentially bare fallows.

5.2. Hydrological pathways and their impact on retention processes

Depending on the hydrological pathways along which nutrients are transported, retention processes may significantly reduce the concentrations of these substances before they reach a surface-water recipient (e.g. a first-order stream). In other words, the amount of nutrients that actually reach the surface water is strongly influenced by the pathways the compounds follow from the soil profile and root zone (i.e. gross losses) to the surface-water recipient. If a soil–water system is characterised by slow-flow processes (as is the case in groundwater-fed rivers), smaller loads are likely to be detected in the receiving surface waters due to increased losses through denitrification, dilution or longer water transit times in groundwater. This may result in measured net losses in stream water that are much smaller than gross losses from the root zone. In a study conducted in Denmark (Grant et al., 1997), N losses from the root zone in sandy soils were estimated to 124 kg ha\(^{-1}\), whereas the load at the catchment outlet was only 12 kg ha\(^{-1}\). Hence, it may be difficult to detect the effect of gross reductions in streams and rivers, particularly if there in addition is considerable inertia or a time delay in the system. Conversely, if a soil–water system is characterised by fast-flow processes (e.g. when flow through groundwater systems are of little importance), a relatively large portion of the gross nutrient losses will be delivered to the surface water. Thus the net losses measured in stream water will be nearly equivalent to the gross losses from the root zone, which, of course, means that the net losses will be high if there are substantial gross losses of nutrient loads. Consequently, in such a system, it is likely that a reduction in gross losses from the root zone will have a relatively rapid effect on the measured load of nutrients in surface waters. In Latvia, about 50% of the annual run-off is generated from snowmelt in spring, 30% comes from rainfall events, and 20% originates from groundwater discharge (Jansons et al., 2002). However, there are some spatial differences, as illustrated by this example: groundwater discharge can contribute up to 40% of the runoff to rivers in the highlands of central Latvia, whereas the corresponding value is <10% in the south-western lowlands (the Venta and Liepāja basins, Fig. 1; Vodogrečky, 1969). Base-flow calculations performed by applying the method proposed by the Institute of Hydrology (1993) to data from three agricultural catchments in central lowland Latvia further confirmed that groundwater played only a minor role, contributing 10–20% of the runoff (Vagstad et al., 2001). Considering these arguments, it can be expected that, in areas with low groundwater recharge, reduced nutrient loadings (e.g. brought about by decreasing surpluses or implementing other mitigation strategies) will have a fairly rapid impact on water quality. Notably, the three significant downward trends in nitrate-N observed in our study (Table 3) were all detected at sites where contribution from groundwater is expected to be low. The rather weak nitrate trend in the Gauja River (Table 3) may have been due to the relatively higher level of groundwater recharge (35–40%; Vodogrečky, 1969).

Long transit times in soil water and groundwater may also cause a substantial time lag between changes in input and output of nutrients in the studied drainage basins. A hydrograph recession analysis has shown that water residence times are much longer in
agricultural basins in Estonia and Latvia than in comparable areas in Norway (Deelstra et al., 1998). This is partly due to differences in the spacing of lateral tile drains: these conduits are typically 10–15 m and in some cases more than 30 m apart in Latvia (Jansons et al., 2002), whereas they are approximately 6–8 m apart in Norway (Deelstra et al., 1998). Furthermore, many of the tile drains in Latvia are in poor condition (i.e. they are clogged or contaminated), because maintenance has been neglected due to the economical recession that began in the agricultural sector in the early 1990s. There are no large-scale or nationwide statistics on the conditions and performance of the tile drains, but it is assumed that a substantial number of these outlets do not function optimally (Jansons, personal communication). With that in mind, and considering the fact that almost 80% of the agricultural land in Latvia are tile drained and cereal yields are low (i.e. 2000 kg ha\(^{-1}\)), it is probable that conditions in agricultural soils in Latvia are favourable for nitrogen retention, especially via denitrification. In addition, anaerobic conditions may inhibit the nitrification of ammonia and lead to evaporation of this gas, and it is possible that this process occurs in some parts of Latvia where the soil pH is close to 8 (Jansons et al., 2002).

5.3. Temporal variability in flow conditions

The studied rivers have fairly similar hydrological regimes that are characterised by pronounced spring flow with a prominent peak in April (Fig. 2). Roughly 45% of the annual river flow occurs during the period March to May. This is particularly prominent in the Daugava River due to snowmelt in the somewhat hillier regions of Russia and Belarus. Rainy periods in autumn can also contribute to the annual river flow on occasion.

The trend in P\(_{O_2}\)-P was observed mainly during winter and spring (not shown), when water discharge is normally at the highest level, which indicates decreased P losses from the soils rather than reduction of emissions from point sources (i.e. municipal wastewater treatment plants). A drop in such emissions would have led to decreased P concentrations primarily during low water flow, when dilution is at a minimum. This finding is supported by similar results published by Smith et al. (1994).

The time series plots shown in Fig. 3 also indicate that there was substantial variability in the observed data, which may have concealed less pronounced downward trends. Natural variability, such as annual irregularity in water discharge, may obfuscate any reductions in loads or concentrations that occur in response to nutrient management actions (Stow et al., 2001), which would be especially noticeable for diffuse sources (Foy and Bailey-Watts, 1998). In addition, it is widely recognised that specific hydrological events can have a large impact on riverine loads of phosphorus (Øygarden, 2000) and thereby introduce spurious trends in time series of phosphorus data. However, variability in water discharge was taken into account in the methods we used to test statistical trends (see Sections 3.2 and 3.3), therefore, it is unlikely that our results were to any noticeable extent jeopardised by irregularities and time trends in flow.

5.4. In-stream, riverine, and lake retention

Phosphorous in rivers and streams are retained by adsorption onto streambed sediments, sedimentation, and via uptake by algae and aquatic macrophytes. The impact of instream physical and biological processes on regulation of P fluxes through river systems is still poorly understood. Nonetheless, according to Hill (1981), as well as other investigators, the adsorption onto bottom sediments is considered to be the major mechanism of P retention. Recently, Schulz et al. (2003) emphasised the importance of macrophytes in this context, estimating that these plants accounted for 25% of the phosphorus retained in a German river. In addition, Imhoff (1989) arrived at an estimate of 50% for instream P retention in the Ruhr River. However, it has also been showed that the main channel of a river can act as a source rather than a sink. Svendsen et al. (1995) studied a Danish lowland stream and found a negative annual P retention due to resuspension of retained material during high flows and also to stream bank erosion. On the other hand, these investigators also noted that retention of dissolved reactive phosphorous (DRP) represented up to 60% of the input of DRP into the stream channel, most likely through uptake of
phosphorous by benthic and pelagic algae instead of aquatic macrophytes. In our study area, part of our research group has previously performed a spatial analysis of water quality data from various sampling sites along a stretch of the Daugava river, and the results suggested that the riverine retention of P was rather low (Stålacke et al., 2002).

Considering nitrogen, there is a growing body of evidence indicating that retention, especially through denitrification, may be an important process in many rivers (Holmes et al., 1996; Howarth et al., 1996; Sjodin et al., 1997). However, little is known about the magnitude of the denitrification that occurs in running waters (Sutton et al., 1995; Chadwick et al., 1999). The other processes responsible for N retention in rivers and streams are organic matter burial in sediments, sediment sorption, and plant and microbial uptake, for example through assimilation by algae and aquatic macrophytes (Billen et al., 1991). Haag and Kaapenjohann (2001) recently reviewed the literature and reported that the retention of nitrate-N in rivers is probably in the range of 1–5%, although values of <20, 20, and 30% have also been published by Seitzinger et al. (2002), Hill (1997), and Billen et al. (1991), respectively. In the Nordic–Baltic region, instream and riverine retention is assumed to be low. Jansson and co-workers (1994b) observed less than 3% retention along a small stream in southern Sweden, and Arheimer (1998) performed model calculations for the whole of southern Sweden and noted an instream retention in the range of 2–7%. In the Kasari River in Estonia, instream retention has been found to account for less than 10% of total N retention (Lidén et al., 1999). Moreover, in our study area, we have previously seen signs of nitrate transformation processes (presumably denitrification) in smaller streams and channels in Latvia and Estonia (Stålacke et al., 1999). More precisely, it was reported that nitrate levels in the main channel outlets of small agricultural catchments were only 30–65% of the concentrations recorded detected in the lateral tile-drain outlets within the same catchment. This means that nitrogen retention in smaller agricultural (i.e. first-order) headwater streams and agricultural surface-water may partly explain the weak nitrogen trends in the rivers investigated in the present study. When it comes to river-scale retention, our earlier spatial analysis of water quality data from various sampling sites along the Daugava River indicated relatively low rates in the range of 1–5% for N (Stålacke et al., 2002). Both the observation of higher retention in smaller streams and the data indicating fairly low retention in the river network agree with findings reported by Alexander et al. (2000), showing that nitrogen retention is inversely related to channel size, that is, it declines rapidly with increasing size of the river channel.

It is well known that, in addition to streams and rivers, lakes and reservoirs can act as efficient traps and/or sinks for phosphorus in particular, which can cause a substantial delay between decreases in emissions or loads upstream and significantly noticeable improvements downstream (e.g. at the mouths of rivers). Nitrogen retention in lakes draining to the northern part of the Atlantic Ocean has been reported to range from 20 to 80% (Howarth et al., 1996), and it is assumed that such retention is generally high in lakes and reservoirs in the Nordic–Baltic region. Jansson and co-workers (1994b) proposed that productive lakes may remove up to 50% of the total N input. However, budget calculations (input/output load estimates) for the three hydropower reservoirs in the lower reach of the Daugava River have indicated insignificant nutrient retention ranging from −10 to 20% for nitrogen and from −10 to 10% for phosphorus (Stålacke et al., 2002). A plausible explanation for this is the short water-residence time in the reservoirs (water exchange 33, 27, and 19 times per year in the respective reservoirs) in combination with the high runoff during snowmelt in spring.

5.5. Other explanatory factors

The downward trends we found in PO_4-P were not generally accompanied by corresponding trends in nitrate (Table 4), which suggests causes other than reduced application of fertilisers. One such possible explanation is the reduced emissions from slaughter houses and animal farms after the dramatic decrease in livestock in 1992–1993, since manure usually tends to be relatively richer in P than in N compared to mineral fertilisers (Sharpley and Withers, 1994). It is also conceivable that the 50% decrease in waste-water discharges (mainly P and organic matter) from 1991 to 2000 (Juhna and Klavins, 2001) had an impact on P
levels. The greatest decline in phosphorus occurred between 1991 and 1995, a period of waning industrial production. In 1995, emissions from point sources to the studied river basins accounted for 1–3% and 11–35% of the total riverine loads of nitrogen and phosphorus, respectively (Helcom, 1998). The highest relative phosphorus emissions were found in the Gauja basin (35%). Reduced emissions of phosphorus and organic matter from point sources can, under certain conditions, reduce the denitrification capacity in rivers (Chesterikoff et al., 1992). We do not have access to any documentation of deficits in dissolved O$_2$ in the water column or the upper sediments in the studied rivers. Nevertheless, it cannot be ruled out that the weak nitrogen trends can also be linked to the more rapid changes in phosphorus emissions and concentrations of phosphorus compounds in the rivers.

As discussed above, internal losses from the river bed may also have an important impact in this context and therefore conceal some trends.

The statistically significant upward trend in nitrate at the uppermost sampling site on the Daugava River is also noteworthy and unexpected. More specifically this site (No. 2; Fig. 1) mainly represents the source emissions that originate from the Russian–Belarussian part of the Daugava River basin, thus we anticipated that it would exhibit the same trend pattern as found for Latvia, in light of reports of generally decreased gross emissions from agricultural sources, improved sewage-treatment facilities and, closing down of industries in Russian and Belarussian river basins (Lääne et al., 2002).

5.6. Final remarks

Based on the discussion presented above, it can be concluded that there are three main explanations for the nutrient trends, especially in regard to nitrogen:

1. The loss of inorganic nitrogen may have been regulated primarily by mineralisation of large pools of organic nitrogen that have accumulated over several years.
2. Long water-transit times in the soil water and groundwater have most likely caused a substantial time lag between changes in the input and the output of nutrients in the studied catchments.
3. High nitrogen retention in smaller agricultural (i.e. first-order) headwater streams and agricultural surface-water channels.

The difference noted between the nitrogen and phosphorus trends also suggests that factors other than reduced fertiliser application influenced the inertia of the water quality response. In other words, it cannot be ruled out that the more rapid phosphorous response in the rivers was also due to other aspects related to agriculture, such as decreased areas of cultivated land, as well as to reductions in point source emissions.

6. Conclusions

The main conclusions that can be drawn from the present results are as follows:

- Despite an unprecedented decrease in the application of mineral fertilisers and manure in Latvia since the late 1980s, we found very little evidence that this change in agricultural practices has influenced the riverine concentrations of nitrogen. More precisely, considering all 12 of the sampling sites we investigated, only three showed downward trends in nitrate, and only four displayed downward trends in inorganic nitrogen.
- We detected downward trends in phosphate concentration at half of the studied sites. Nevertheless, it cannot be ruled out that this was due to factors other than decreased agricultural emissions.

Acknowledgements

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