Water quality status and trends in agriculture-dominated headwaters; a national monitoring network for assessing the effectiveness of national and European manure legislation in The Netherlands

J. C. Rozemeijer · J. Klein · H. P. Broers ·
T. P. van Tol-Leenders · B. van der Grift

Abstract Large nutrient losses to groundwater and surface waters are a major drawback of the highly productive agricultural sector in The Netherlands. The resulting high nutrient concentrations in water resources threaten their ecological, industrial, and recreational functions. To mitigate eutrophication problems, legislation on nutrient application in agriculture was enforced in 1986 in The Netherlands. The objective of this study was to evaluate this manure policy by assessing the water quality status and trends in agriculture-dominated headwaters. We used datasets from 5 agricultural test catchments and from 167 existing monitoring locations in agricultural headwaters. Trend analysis for these locations showed a fast reduction of nutrient concentrations after the enforcement of the manure legislation (median slopes of −0.55 mg/l per decade for total nitrogen (N-tot) and −0.020 mg/l per decade for total phosphorus (P-tot)). Still, up to 76 % of the selected locations currently do not comply with either the environmental quality standards (EQSs) for nitrogen (N-tot) or phosphorus (P-tot). This indicates that further improvement of agricultural water quality is needed. We observed that weather-related variations in nutrient concentrations strongly influence the compliance testing results, both for individual locations and for the aggregated results at the national scale. Another important finding is that testing compliance for nutrients based on summer average concentrations may underestimate the agricultural impact on ecosystem health. The focus on summer concentrations does not account for the environmental impact of high winter loads from agricultural headwaters towards downstream water bodies.

Keywords Monitoring · Water quality · Agriculture · Nutrients · Nitrogen · Phosphorus

J. C. Rozemeijer (✉) · J. Klein · H. P. Broers ·
B. van der Grift
Deltares, P.O. Box 85467, 3508 TA Utrecht,
The Netherlands
e-mail: joachim.rozemeijer@deltares.nl

H. P. Broers
TNO Geological Survey of the Netherlands,
P.O. Box 80015, 3508 TA Utrecht,
The Netherlands

H. P. Broers
Critical Zone Hydrology Group, Vrije Universiteit
Amsterdam, De Boelelaan 1085, 1081 HV Amsterdam,
The Netherlands

T. P. van Tol-Leenders
Alterra, Wageningen University and Research Centre,
P.O. Box 47, 6700 AA Wageningen, The Netherlands

B. van der Grift
Department of Innovation, Environmental and Energy
Sciences, Utrecht University, Utrecht, Netherlands

Introduction

The loss of chemicals like nutrients, heavy metals, and pesticides from agricultural fields contaminates groundwater and surface water bodies. This threatens the ecological, industrial, and recreational functioning of these water systems. For example, the enrichment of

ecosystems with nutrients (eutrophication) results in a general loss of biodiversity (e.g. Weijters et al. 2009) and in an increase of toxic algae blooms and hypoxia (e.g. Makarewicz et al. 2007).

Due to a highly concentrated and productive agricultural sector in The Netherlands, nutrient surpluses and leaching are higher than elsewhere in Europe (Oenema et al. 2007; Bouraoui & Grizzetti 2011) and the world (Bouwman et al. 2009). This has caused high nutrient concentrations in groundwater (e.g. Visser et al. 2007), local surface waters (e.g. Mul & Van der Vlies 1999), rivers (e.g. De Wit et al. 2002; Bouraoui & Grizzetti 2011), lakes (e.g. Maasdam & Claassen 1998), estuaries (e.g. Soetaert et al. 2006), and coastal waters (e.g. Skogen & Mathisen 2009). In many cases, nutrient concentrations in surface waters in The Netherlands exceed water quality standards (Oenema & Roest 1998; Oenema et al. 2007). The resulting eutrophication problems are most pronounced in stagnant downstream lakes, wetlands, and near-shore coastal waters (EEA 2001).

To mitigate the environmental issues related to agricultural pollution, the national government in The Netherlands enforced the Dutch Manure Law in 1986 (Dutch Manure Law 1986). In 1991, The Netherlands also adopted the European Nitrate Directive (EU 1991), which regulates the use of nitrogen in agriculture through national action plans. Among other measures, the regulation includes a maximum application of 170 kg N and a ban on fertilizer and manure application outside the growing season. Like in many EU member states, this regulatory policy caused a decrease in nutrient surpluses (Bouraoui & Grizzetti 2011).

The national and European manure legislation involves restrictions that may affect agricultural productivity and profitability. Demonstrating the positive effects of these land use restrictions for ecosystems is crucial to ensure enduring political and social support. However, the effects on nutrient concentrations in surface water appeared hard to point out; the detection of human-induced trends in surface water quality is often hindered by the large weather-induced variability in solute concentrations (Rozemeijer et al. 2010a). In addition, nutrients are notoriously difficult to trace back to one specific source. Monitoring locations are usually situated in the downstream parts of the larger streams to represent a large upstream area with multiple nutrient sources (Bartram & Balance 1996; CIW 2001). Beside agriculture, common nutrient sources include sewage treatment plants, sewage overflows, industrial spills, other urban sources, and atmospheric deposition. In addition, more than 60 % of the surface waters in The Netherlands are influenced by diverted river water from the Meuse and Rhine rivers (Roelofs 1991; Rozemeijer et al. 2012). As a final complication for evaluating manure legislation, temporal and spatial variability of natural background concentrations and biochemical processes also affect nutrient concentrations in surface waters (e.g. Topcu et al. 2011).

In spite of all complications, the exceptionally high density of water quality monitoring locations in The Netherlands still provides opportunities to assess the effects of manure legislation on nutrient concentrations in surface water. About 2500 surface water quality monitoring locations are frequently sampled by local Water Boards. In this study, we combined careful selections from existing regional nutrient monitoring datasets into a national assessment of water quality status and trends in agriculture-dominated headwaters.

The objectives of this study were (1) to combine existing monitoring locations into a national-scale monitoring network for agriculture-dominated headwaters in The Netherlands, (2) to assess the water quality status for nutrients in these headwaters, and (3) to explore whether national and European manure legislation has influenced surface water quality. We used datasets from 5 agricultural test catchments and selected 167 existing monitoring locations in agriculture-dominated headwaters for our national assessments. The monitoring data were tested for compliance with the local water quality standards. In addition, we did a trend analysis using the data of 87 locations with measurements starting in or before 2000. With respect to former research, this paper adds (1) a national assessment of manure policy efficiency in one of the most productive and most densely monitored agricultural regions worldwide and (2) a discussion on how detailed monitoring in test catchments can facilitate regional-scale monitoring data interpretation.

Study area

The Netherlands (ca. 37,000 km^2, 50–54° N, 3–7° E; Fig. 1a) has a semi-humid maritime climate with a yearly precipitation of 500 to 1100 mm and a yearly estimated evaporation of 200 to 800 mm. Average net precipitation varies from +70 mm per month in winter to
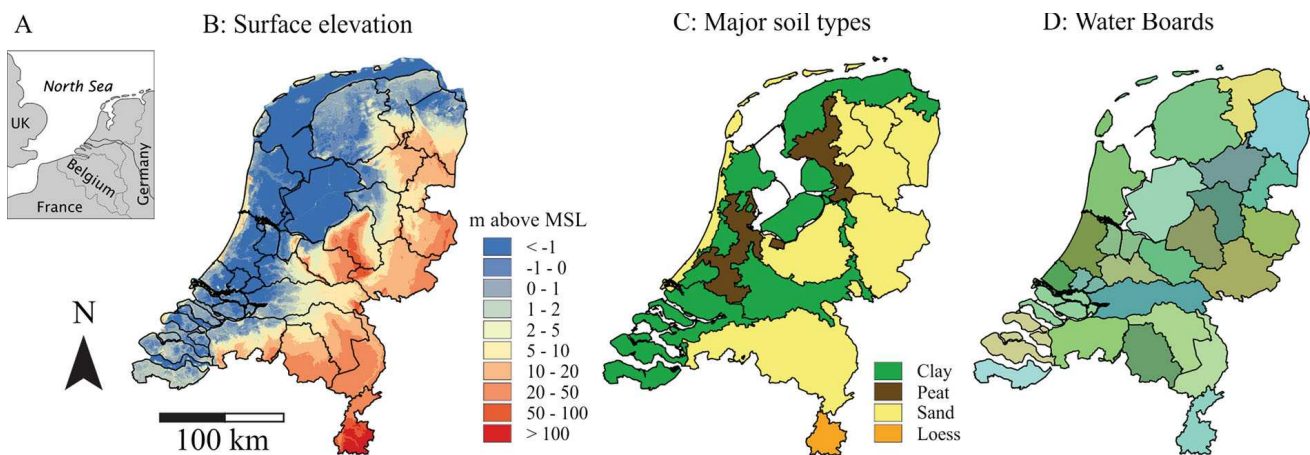
Fig. 1 Study area; **a** location of The Netherlands within Europe, **b** elevation map, **c** major soil types, **d** Water Boards

−20 mm per month in summer. The growing season, defined here as the period with day average temperatures of at least 5 °C, is from March until November. Surface elevations in The Netherlands vary from around mean sea level (MSL) in the western and northern parts up to around 50 m above MSL in the eastern and southern parts (Fig. 1b). The Netherlands is almost completely covered with Pleistocene and Holocene deposits. Pleistocene sandy deposits (aeolian cover sands and fluvioglacial deposits) are at the surface in the southern and eastern parts of The Netherlands (Fig. 1c). The western and northern parts are covered with Holocene fluvial clay sediments and peat soils.

About 50 % of The Netherlands (18,600 km^2) is in agricultural use. Grassland of 9900 km^2 is used for cattle breeding, 7700 km^2 is arable land, and 1000 km^2 is horticulture (CBS 2012). Due to the intensive agricultural land use, nutrient surpluses and leaching in The Netherlands are higher than elsewhere in Europe (Oenema et al. 2007). The N and P surpluses, defined as the input of nutrients (manure and fertilizer) minus the output via crop uptake, are plotted in Fig. 2 for the period 1970–2010. From 1960 to 1985, the amount of livestock in The Netherlands increased, leading to increasing N and P surpluses. For example, the number of pigs increased from 2 to 14 million in this period (Oenema et al. 2007), almost outnumbering the human population. The introduction of maize in 1970, on which a virtually unlimited amount of manure and fertilizer could be applied, allowed for larger cattle densities and a further increase in the N and P surpluses until 1986 (Visser et al. 2007). In 1986, manure legislation (Dutch Manure Law 1986) was enforced in The Netherlands. This law aimed at improving groundwater and surface water quality, mainly by reducing the application of manure and implementing bans on autumn and winter applications of manure. This caused a direct reduction of the N and P surpluses from 1986 onward (Fig. 2). In addition, The Netherlands adopted the EU Nitrates Directive in 1991. This EU legislation forces member states to make action plans to reduce leaching of nitrate to groundwater and surface waters. For 2015, the Dutch national government aims at eliminating the nutrient surpluses (i.e. equilibrium fertilization).

Methods

Data collection

We composed a national monitoring network using existing monitoring locations and water quality time series owned by the 25 Water Boards in The
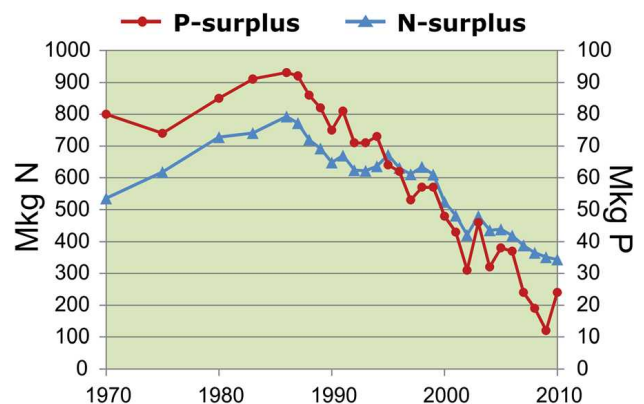

Fig. 2 National N and P surpluses (in million kg) in The Netherlands for 1970–2010 (CBS 2012)

Netherlands (Fig. 1d). In sessions with the local water managers, we selected monitoring locations where agriculture was the main anthropogenic nutrient source in the upstream catchment area. Locations influenced by urban effluent, sewage treatment plants, or industrial spills were excluded. Furthermore, we selected locations with a minimum influence of diverted river water. Especially in artificial and hydrologically complex catchments, this involved consulting the local controllers of small-scale weirs and pumps to recover flow directions and water sources. A final condition for selecting locations was a monthly measurement interval for total nitrogen (N-tot) and total phosphorus (P-tot), which is a common frequency in these surface water monitoring networks. Further details on the selection of monitoring locations were reported by Klein et al. (2012).

The selected monitoring locations typically represent relatively small (10–50 km^2) agricultural headwater catchments. In the sandy southern and eastern parts of The Netherlands (Fig. 1c), most locations are situated in free-flowing, naturally shaped upper parts of stream valleys. In the north and west, locations in artificial polder catchments with controlled water levels are dominant. We selected 167 monitoring locations to assess compliance to environmental standards. Of the selected locations with time series longer than 10 years, 87 were also used for trend analysis. The sampling, conservation, and laboratory analyses were performed by the Water Boards following standardized operating procedures (CIW 2001).

Prior to the data analysis, all individual datasets were combined into one coherent dataset. This involved data inspection and making uniform all units, calculation methods for N-tot and P-tot, and methods of dealing with measurements below the detection limits. In case of inconsistencies and uncertainties, we directly contacted the data providers at the Water Boards for clarification. The dataset was checked for the presence and quality of meta information for each location (location codes, coordinates, descriptions, measurement frequency, start and end dates, location types, and applicable water quality standards). All units were checked and, if needed, changed to milligrammes per litre of N and milligrammes per litre of P for all N and P components. Double measurements (two values for the same location, parameter, and time) were removed. Different causes for these duplicates emerged, and depending on the preference of the data providers, we either used the average value or one of the data values. Several data checks were performed, such as a test on high values below the detection limits, extremely high and low values, and inconsistencies (e.g. PO$_4$ concentration cannot be higher than P-tot). Most inconsistencies were related to N and P components with concentrations close to or below the detection limits. The Water Boards applied different methods for calculating N-tot values from the N components. To obtain consistent N-tot values for our national dataset, we did our own uniform calculation using the provided individual N component concentration values. In most cases (88 %), we could calculate N-tot by summing the separate N-Kjeldahl (N-Kj), nitrate (NO$_3$), and nitrite (NO$_2$) measurements. For 12 % of the measurements, only directly measured N-tot concentrations were available. For P, direct measurements of P-tot were available for all locations. Further details on the data processing and the results of the checks are described in Klein et al. (2012).

In addition to the national dataset, we collected high-resolution temporal and spatial datasets from recent research projects in five agricultural test catchments. The catchments were situated in the peat area (Krimpenerwaard), the clay area (Quarles van Ufford), and the sand area (Schuitenbeek, Drentse Aa, and Hupsel) (see Fig. 4) covering the major soil types of The Netherlands. The objectives of these catchment-scale research projects were to unravel the sources and pathways of nutrients and to identify effective mitigation options. In the Hupsel catchment, near-continuous (15 min interval) data of NO$_3$, P-tot, and discharge were collected in 2007–2008. At the catchment outlet, discharge was measured in a calibrated weir, NO$_3$ with a Hydrion-10 multiparameter probe (Hydrion, The Netherlands) and P-tot by a Phosphax auto-analyzer (Hach Lange, Germany). For more information on the monitoring setup and results from the Hupsel catchment, we refer to Van der Velde et al. (2009, 2010a, b, 2011) and Rozemeijer et al. (2010a, b, c). In the other four catchments, 20–30 water quality monitoring locations were sampled weekly or monthly in the period 2003–2010. At the catchment outlets, discharge was recorded and flow-proportional average weekly samples were collected with auto-samplers. We refer to Woestenburg & Van Tol-Leenders (2011) for details on the monitoring setup and results. For all catchments, the monitoring setup allowed for very accurate load estimates by combining the high-resolution discharge and concentration data.

Compliance testing

Compliance was tested by applying environmental quality standards (EQSs) based on summer average concentrations, as was decided in the implementation process of the EU Water Framework Directive in The Netherlands (e.g. Heinis & Evers 2007). For all 167 locations, we calculated the summer average concentrations for the years 2007 to 2010 by averaging the 6 monthly measurements sampled from April to September. The EQSs are set by the Water Boards based on a national assessment on ecosystem vulnerability (Heinis & Evers 2007) and are water type and location dependent. The most common EQSs for N-tot are 2.4 mg/l for most ditches and channels in the clay and peat regions and 4.0 mg/l for the free-flowing streams in the sandy region. For P-tot, the most common EQSs are 0.15 and 0.22 mg/l, respectively. The EQSs are relatively high, because almost all surface water in The Netherlands are categorized as heavily modified and artificial water systems in EU Water Framework Directive settings. For more details and backgrounds on the EQSs, we refer to Lammens et al. (2008) and Heinis & Evers (2007).

Trend analysis

At 87 of the selected agriculture-dominated measurement locations, monitoring started in or before 2000. These locations were used for trend analysis. At 38 of these locations, monitoring started in or before 1990, 42 locations started between 1991 and 1995, and 7 locations between 1995 and 2000. As a basic principle, we first inferred trends for the individual locations and subsequently combined these individual trends to one aggregated national trend and to trends per soil type. The alternative approach would be to aggregate the concentrations of individual locations to yearly group average or median values and to apply trend analysis to these aggregated concentration levels. The advantage of our approach (first trend analysis, then aggregation) is that the large spatial variability in the absolute concentration levels does not influence the trend analysis results. In addition, this method is less sensitive to missing data periods in the individual time series. This trend analysis approach was also used in studies by Broers & Van der Grift (2004), Visser et al. (2009), and Kronvang et al. (2008).

We used three frequently applied methods for trend analysis: (1) seasonal Mann-Kendall tests (Hirsch & Slack 1984), (2) Theil-Sen robust lines (Hirsch et al. 1982), and locally weighted scatterplot smoothing (LOWESS) trend lines (Cleveland 1979). These methods are all relatively insensitive to extreme values and missing data in the individual time series. The seasonal Mann-Kendall trend test is a robust, non-parametric test on the significance of an upward or downward trend. The test results give the direction of the trend and the probability (p value) that the null hypothesis (no trend) is true. We interpreted a p value lower than 0.05 ($p<0.05$) as a significant trend. The Theil-Sen method is a robust non-parametric trend slope estimator. We used Theil-Sen to estimate the overall slopes in the individual concentration time series. These slopes were aggregated to trend slopes per soil type and a national trend by calculating the average and median Theil-Sen slope. In addition, we calculated the 95 % confidence interval around the average and median Theil-Sen slopes, using the non-parametric approach described by Helsel & Hirsch (1992). The LOWESS trend lines (Cleveland 1979) were used to examine possible changes in trend slopes within the concentration time-series period. LOWESS gives a smoothed curve through the concentration time series.

Results

Concentrations

Figure 3a, b summarizes all N-tot and P-tot concentration data for the 167 selected agriculture-dominated monitoring locations within the period 2007–2010. The data for the clay, sand, and peat regions are shown as separate lines. In addition, summer (April–September) and winter (October–March) concentrations are displayed separately. For reference, the most common EQSs for N-tot (2.4 and 4.0 mg/l) and P-tot (0.15 and 0.22 mg/l) are shown as grey horizontal lines.

For all soil types, the winter N-tot concentrations in agriculture-dominated headwaters were higher than the summer concentrations. The N-tot concentrations in the clay and peat areas were quite similar, while N-tot concentrations in the sand area were distinctly higher in summer and winter than in the other soil types. N-tot concentrations above 10 mg/l occurred in 10–20 % of the measurements in the sand area. For P-tot, highest
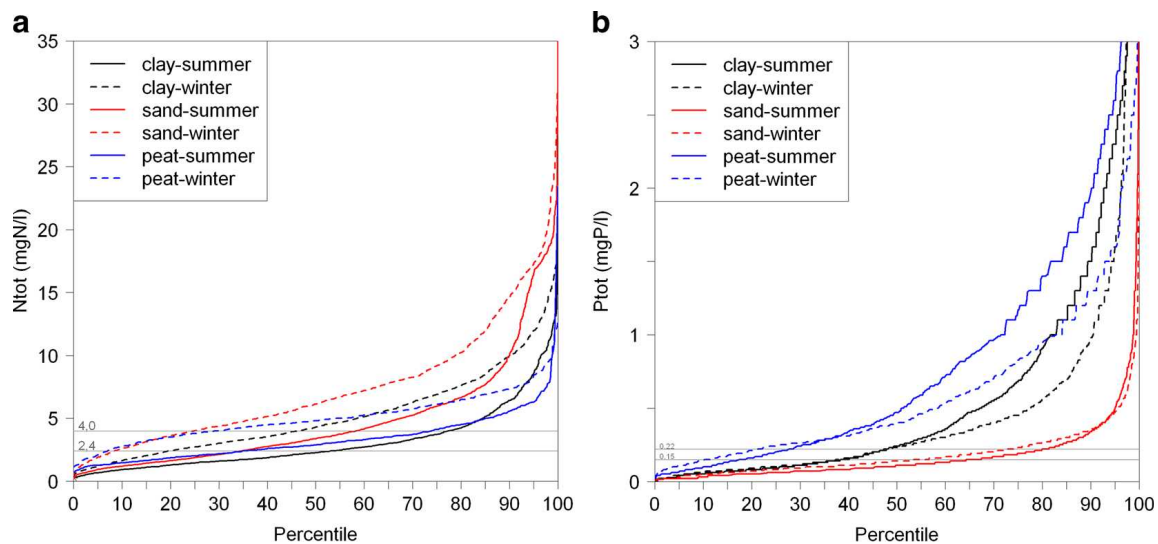
Fig. 3 Cumulative frequency diagrams showing observed N-tot (**a**) and P-tot (**b**) concentrations (2007–2010) in peat, sand, and clay areas in winter and summer. *Grey horizontal lines* indicate the most common EQS values for summer average concentrations

concentrations were measured in the agricultural headwaters in the peat area and the lowest in the sand area. While P-tot concentrations above 0.3 mg/l were quite common in the clay (45 %) and the peat (65 %) areas, only 10 % of the measurements in the sandy region exceed this value. In the peat and clay areas, summer P-tot concentrations were higher than winter P-tot concentrations. For the sand area, no clear distinction between summer and winter concentrations appears from Fig. 3b.

Compliance testing

The results of the compliance testing for all 167 agriculture-dominated locations are shown in Figs. 4 and 5. We present the testing results separately for the years 2007–2010, because compliance varied considerably over these years.

The compliance percentage for N-tot varied from 38 % in 2007 to 61 % in 2009 (Fig. 4). For P-tot, compliance varied between 43 % in 2010 and 55 % in 2007. The combined test shows that between 24 % (2007) and 39 % (2009) of the agriculture-dominated locations complied with both the regional EQS for N-tot and P-tot.

Figure 5 gives a map with the results of the combined compliance test for N-tot and P-tot for 2007–2010. The noncompliant locations were most abundant in the western part of The Netherlands. In the peat area, between 6 and 12 % of the locations complied with both the P-tot EQS and the N-tot EQS. For the clay area, these

percentages were between 24 and 34 %. Especially in the sandy areas, the compliance varied between the years (between 28 and 53 %).

Trend analysis

The results for the trend analysis for 87 agriculture-dominated locations are summarized in Tables 1 and 2 and Fig. 6. Table 1 gives the numbers of significant upward and downward trends. For N-tot, 76 % of the locations were subjected to significant downward trends, and no significant upward trends were found. P-tot concentrations were decreasing significantly on 52 % of the agriculture-dominated locations. Significant upward trends for P-tot were found for 13 % of the locations.

The aggregated results of the Theil-Sen robust lines analyses are shown in Table 2. The median trend slope for N-tot was −0.55 mg/l per decade. For P-tot, the median trend slope was −0.02 mg/l per decade. The relatively high winter N-tot concentrations decline almost twice as fast as the summer N-tot concentrations. For P-tot, the decrease of the summer concentrations is steeper than the decrease of the winter concentrations. The decline of the N-tot concentrations is faster in the sand area (−0.78 mg/l per decade) than in the clay area (−0.43 mg/l per decade). Median slopes for the peat area are not presented in Table 2, because the number of eight monitoring locations was considered too low to estimate accurate median slopes. For P-tot, the decline of the concentrations is faster in the clay area
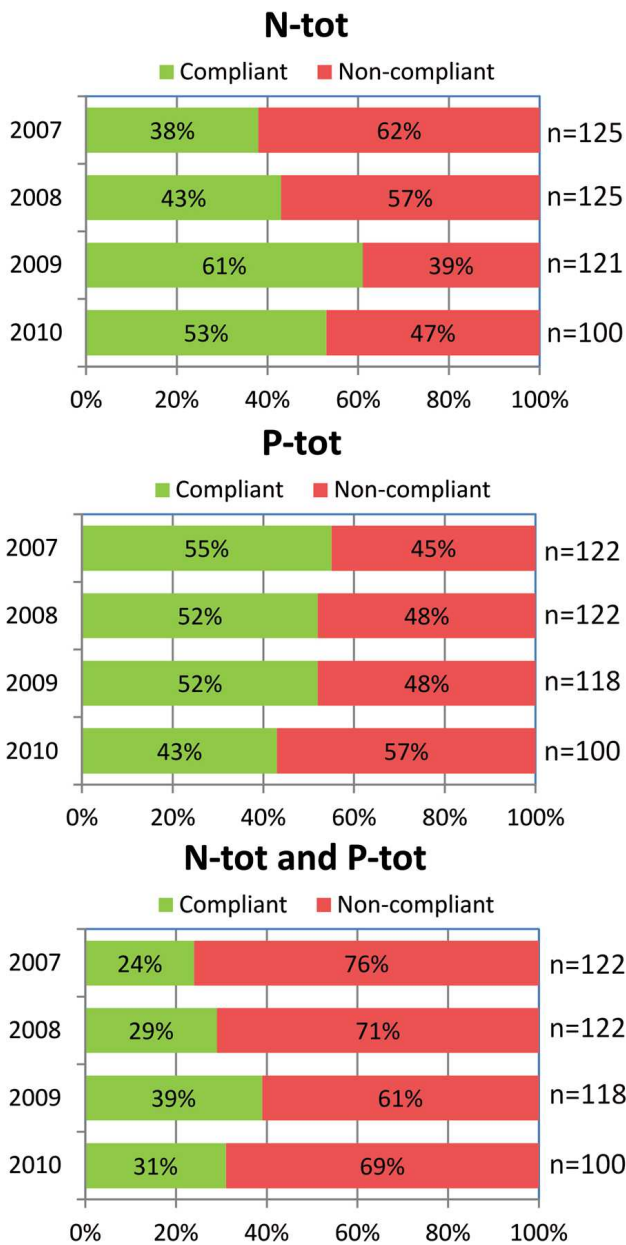
N-tot

■ Compliant ■ Non-compliant

Year	Compliant	Non-compliant	n
2007	38%	62%	n=125
2008	43%	57%	n=125
2009	61%	39%	n=121
2010	53%	47%	n=100

0% 20% 40% 60% 80% 100%

P-tot

■ Compliant ■ Non-compliant

Year	Compliant	Non-compliant	n
2007	55%	45%	n=122
2008	52%	48%	n=122
2009	52%	48%	n=118
2010	43%	57%	n=100

0% 20% 40% 60% 80% 100%

N-tot and P-tot

■ Compliant ■ Non-compliant

Year	Compliant	Non-compliant	n
2007	24%	76%	n=122
2008	29%	71%	n=122
2009	39%	61%	n=118
2010	31%	69%	n=100

0% 20% 40% 60% 80% 100%

Fig. 4 Bar chart of agriculture-dominated monitoring locations that complied (*green*) and did not comply (*red*) to their regional EQS values in 2007–2010. Shown here is the compliance for N-tot, P-tot, and the combination of N-tot and P-tot. *n* gives the number of locations that the compliance testing is based on

(−0.032 mg/l per decade) compared to the sand area (−0.011 mg/l per decade).

Figure 6 gives a visual aggregation of the trend analysis results. In green, we plotted the median Theil-Sen slopes and their 95 % confidence intervals which were also presented in Table 2. The median concentration determines the elevation of the midpoint of the trend line, where the trend lines cross (Fig. 6). Average

Theil-Sen slopes were plotted in blue lines for which the midpoint elevation is equal to the overall average concentration. For both N-tot and P-tot, the average trend line positions are much higher than the medians due to the skewness of the concentration distribution (see also Fig. 3). In addition, the average slopes are steeper than the median slopes due to the skewness of the trend slope distribution. The black lines in Fig. 6 give the aggregated LOWESS trend lines (solid) and the p25 and p75 LOWESS trend lines (dotted). The LOWESS trend lines do not reveal many changes in the aggregated trend slopes over the years. For both N-tot and P-tot, the decrease in concentrations at the agriculture-dominated locations seems to accelerate slightly around 2002.

Agricultural test catchments

The high-resolution concentration measurements at the agricultural test catchment outlets revealed the large temporal variability in nutrient concentrations. As an example, Fig. 7 gives a selection of continuous concentration measurements from the Hupsel catchment outlet. Figure 7a gives the P concentrations of February to May 2008. The arrows indicate a common 1-month grab sampling interval which clearly is not sufficient to capture the large variability in concentrations in surface water. This undersampling results in large uncertainties in estimates of average concentrations and loads, as was shown by Rozemeijer et al. (2010a).

The continuous measurements also gave insight into the concentration response to rainfall events and into the dominant hydrological and chemical processes in a catchment. Figure 7b, c gives the response of NO_3 and P concentrations at the Hupsel catchment outlet to two events (see also Rozemeijer and Van der Velde 2014). NO_3 concentrations repetitively lowered in response to events. At the start of the events, during average flow conditions, the stream discharge mainly consists of relatively young, NO_3-rich shallow groundwater and tile-drain effluent. The rapid response to rainfall events consists of overland flow, interflow, and direct precipitation with low NO_3 concentrations. After some rainfall events during the summer period, the NO_3 concentrations recovered up to a higher level than before the event (e.g. Fig. 7c). This occurred after large summer storms with dry antecedent conditions. These events cause a rise in groundwater levels and even a reactivation of the tile-drain discharge, which leads to increased NO_3 loads towards the surface water. During the discharge peak,

Fig. 5 Compliance to regional EQS values based on summer average concentrations for 4 different years. For each location, *four squares* are shown with the compliance in 2007 (*upper left*) 2008 (*upper right*), 2009 (*lower left*), and 2010 (*lower right*). The agricultural test catchments are indicated from west to east by *1* Krimpenerwaard, *2* Schuitenbeek, *3* Quarles van Ufford, *4* Drentse Aa, and *5* Hupsel
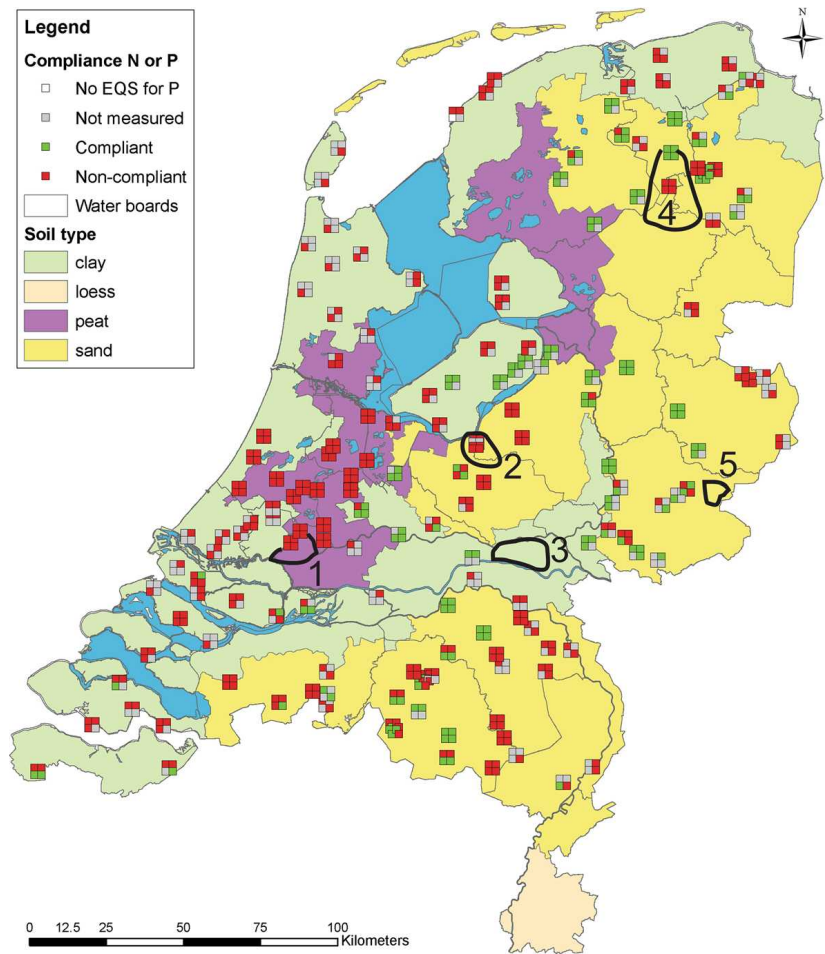
these increased loads are overshadowed by the dilution with quickly discharged precipitation water. After this initial dilution, the increased groundwater levels and tile-drain discharges cause higher NO_3 concentrations than before the event.

Table 1 Aggregated results of the seasonal Mann-Kendall trend tests; number of significant upward and downward trends in the N-tot and P-tot concentrations at agriculture-dominated locations from 1980 to 2010

	Clay	Peat	Sand	Total
N-tot				
Upward ($p<0.05$)	0	0	0	0
Downward ($p<0.05$)	27	5	34	66
No significant trend	8	3	10	21
P-tot				
Upward ($p<0.05$)	2	2	7	11
Downward ($p<0.05$)	21	2	22	45
No significant trend	12	4	15	31

Table 2 Aggregated results of the Theil-Sen robust lines analyses; median trend slopes and confidence intervals for all data, for summer (April–September) and winter (October–March) concentrations, and for the clay and sand areas. For the peat area, the number of monitoring locations was considered too low to estimate accurate median slopes

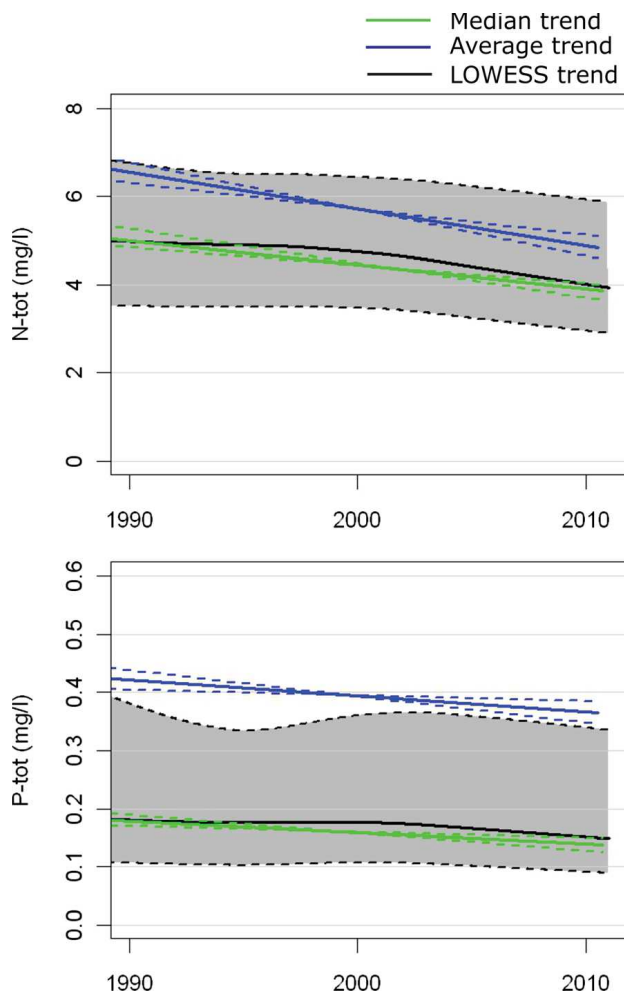|	Median trend (mg/l per decade)	95 % confidence interval (mg/l per decade)
N-tot		
All data	−0.55	−0.77/−0.41
Summer	−0.48	−0.72/−0.34
Winter	−0.89	−1.30/−0.61
Clay	−0.43	−0.67/−0.22
Sand	−0.78	−1.44/−0.52
P-tot		
All data	−0.020	−0.031/−0.001
Summer	−0.025	−0.036/−0.017
Winter	−0.015	−0.027/−0.006
Clay	−0.032	−0.064/−0.018
Sand	−0.011	−0.025/−0.005

Fig. 6 Trends in N and P concentrations over the period 1990–2010. Trends in individual time series were aggregated to deduce a median trend (*green*), an average trend (*blue*), and a LOWESS smooth (*black*) with uncertainty band (25 and 75 percentile) around the median using LOWESS smooths (*grey*)

second event, the maximum P concentration occurred during the first event. The high P concentrations during the first discharge peak can be explained by the large supply of transportable P that has accumulated during the dry period before these events. At the start of the second event, a large proportion of this transportable P has already been removed from the catchment during the first event. The accumulation and flushing process explains the short peaks in P concentrations in response to events. For a more elaborate review on the driving processes of short-scale temporal variations in solute concentrations and the consequences for monitoring, we refer to Rozemeijer and Van der Velde (2014).

The high-resolution discharge and concentration data also allowed for accurate load estimates for all agricultural test catchment outlets (Woestenburg & Van Tol-Leenders 2011). In contrast to concentrations, loads can be used to assess the impact of agricultural catchments on nutrient concentrations in downstream receiving surface water bodies. As an example, Fig. 8 gives the cumulative yearly N and P loads from the Drentse Aa catchment in 2003 to 2010 (Roelsma et al. 2011). The plots show a large variability in total yearly loads between the years. Furthermore, all individual cumulative load curves are steep during the winter period and much flatter during summer. This indicates that most nutrient export (circa 80–90 %) from these catchments towards downstream water bodies occurs during the winter period (November–March).

Discussion

Compliance testing

In this study, we evaluated the national and European manure legislation by assessing the water quality status for N-tot and P-tot concentrations in agriculture-dominated headwaters in The Netherlands. We found that up to 76 % of the 167 agriculture-dominated monitoring locations do not comply with either the EQS for N-tot or for P-tot. These results indicate that further reductions of nutrient losses are needed to reach compliance with the EQSs in agricultural headwaters in The Netherlands.

The compliance testing results varied considerably over the years 2007–2010 for both N and P, while no major changes in legislation, land use, and nutrient surpluses occurred during this period. These variations
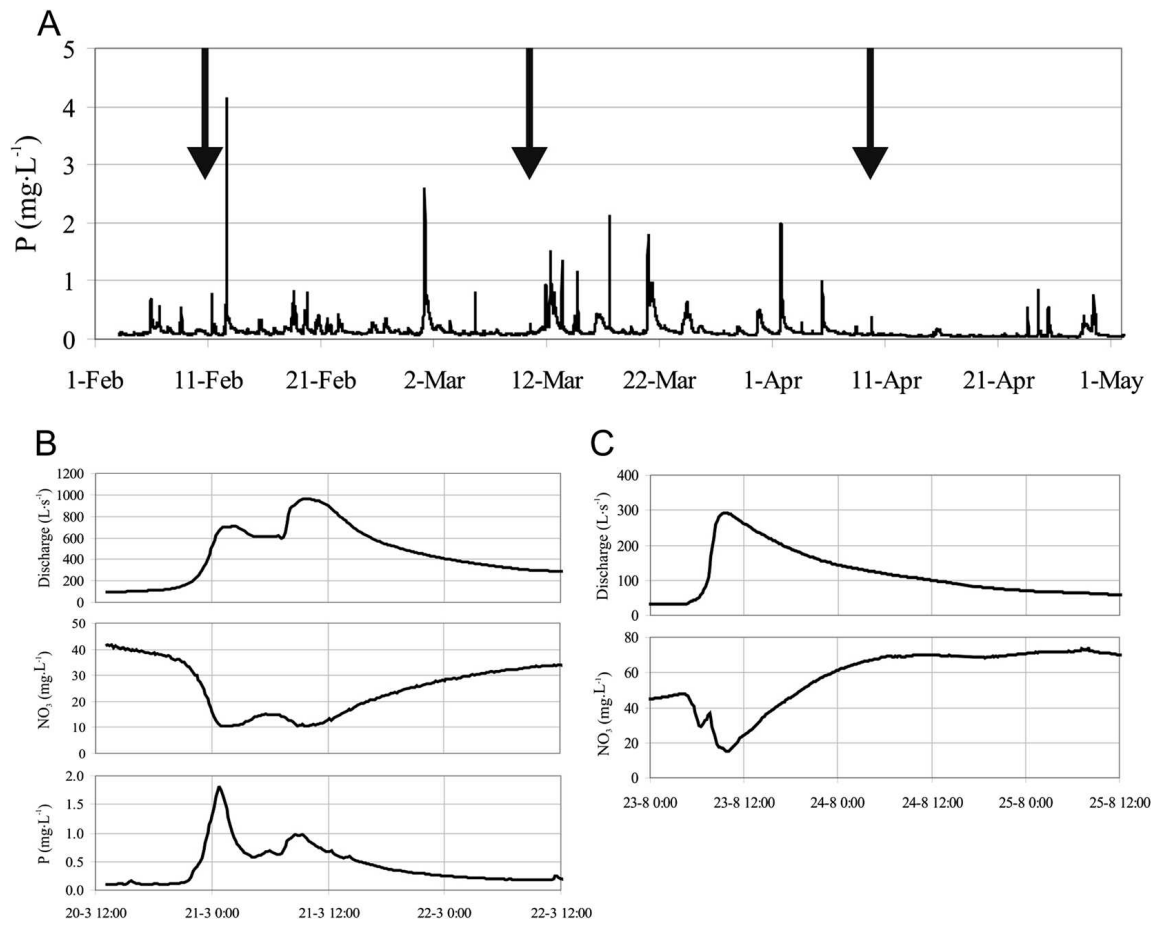
In contrast to NO$_3$, the P concentrations in the Hupsel catchment structurally peaked during rainfall events (Fig. 7a, b, Rozemeijer and Van der Velde 2014). This is related to the large amounts of P-rich Fe- and Al-hydroxides at the ditch bottoms and inside tile drains that we observed in the catchment. During steady flow conditions, these 'iron curtains' build up when anoxic Fe-rich groundwater seepage gets oxidized in the drains or the ditch and adsorb P ions from discharged groundwater. When the water flow velocities in the tile drains and ditches increase, this particulate P is detached and transported downstream. During extremely wet conditions, overland flow also contributed to the P peaks. The process of P accumulation and flushing is strikingly demonstrated by the stacked event on 21 March 2008 (Fig. 7b). Despite the higher peak discharge for the

Fig. 7 Examples of high-resolution water quality data from the Hupsel catchment. **a** continuous P-concentration measurements for February to May 2008 with *arrows* indicating a common monthly sampling scheme. **b** Discharge and P and NO₃ concentration response to a stacked rainfall event at 21 March 2008. **c** Summer storm event at 23 August 2007 with higher NO3 concentrations after a dilution dip. Adapted from Rozemeijer and Van der Velde (2014)

in nutrient concentrations and loads are related to weather variability as was observed from high-resolution data (e.g. Figs. 7 and 8). Most Dutch farmers apply the maximum allowed amount of N and P regardless of
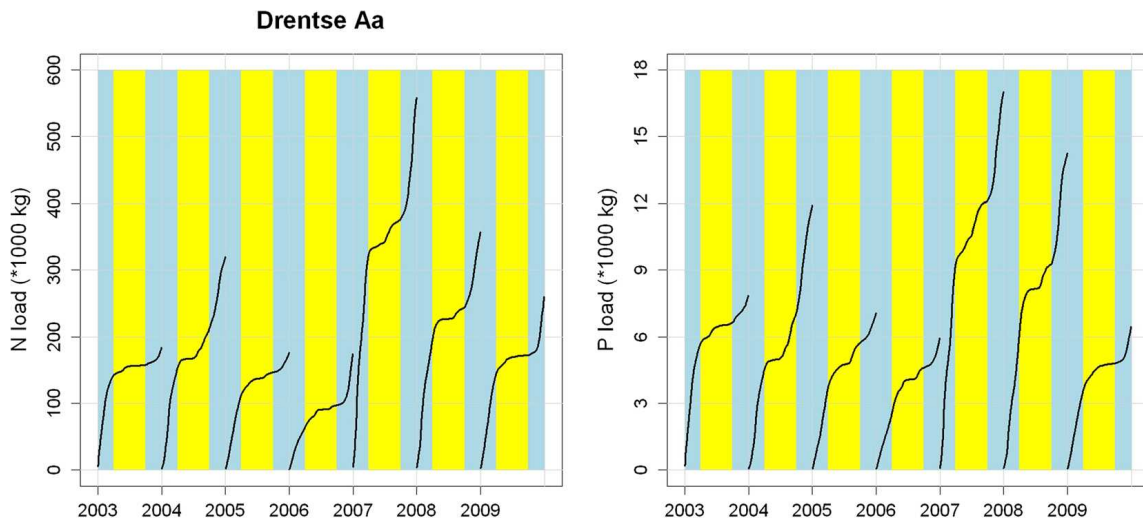

Fig. 8 Cumulative yearly N and P loads from the Drentse Aa test catchment. The blue bands indicate the winter period (November–March) and the yellow bands indicate the summer period (April–October) (from Roelsma et al. 2011)

weather conditions. However, wet conditions and a larger contribution of shallow flow routes to surface waters can mobilize the legacy stores of N and P from agricultural soils (Rozemeijer & Broers 2007). The large weather-induced variability is important to consider when interpreting the common water quality status assessments for individual years. To prevent misjudgments and inappropriate management decisions, water quality status assessments should be based on several years of data.

A further drawback of compliance testing using summer average concentrations is that most nutrient export from agricultural catchments occurs during the winter period (Fig. 8). Summer leaching of water and solutes from agricultural fields is limited in The Netherlands due to a precipitation deficit (i.e. more evapotranspiration than precipitation). During summer, the surface water system is mainly fed by deeper groundwater, which is less affected by recent agricultural activities (Rozemeijer & Broers 2007). In most catchments, the winter inputs from agriculture are dominant and summer concentrations of N-tot and P-tot are lower than the winter concentrations (Fig. 3). In the peat and clay catchments, however, a larger contribution of P-rich groundwater, together with the release of P from bottom sediments during anoxic conditions in dry and warm periods, leads to higher concentrations in summer (Fig. 3, see also Woestenburg & Van Tol-Leenders 2011). As summer concentrations in agricultural headwaters are mainly controlled by deep groundwater and bottom sediment chemistry, focusing on summer concentrations in agricultural headwaters is not appropriate for evaluating the impact of recent agricultural activities on downstream water quality and ecology. Still, the legitimate EQSs for agricultural headwaters are based on summer average concentrations as was decided in the implementation process of the Water Framework Directive in The Netherlands (e.g. Heinis & Evers 2007). As a consequence, compliance monitoring in The Netherlands focuses on the local ecological effects of nutrient availability in the growing season at the monitoring locations itself. Nutrient transport from agricultural headwaters to downstream water bodies is not considered. This is a major shortcoming, as eutrophication problems are most pronounced in stagnant downstream water bodies (EEA 2001). Detection and mitigation of large winter loads may therefore be more beneficial for ecosystem health than focusing on summer concentrations and local ecology in the headwaters.

The monitoring of summer average nutrient concentrations in downstream water bodies may be sufficient to detect emerging eutrophication problems. However, this monitoring is not sufficient to evaluate the effectiveness of nutrient management policy on nutrient exports and on downstream stagnant water body ecology, where winter-imported nutrients play a crucial role in summer eutrophication. The same dominance of winter nutrient exports from agricultural catchments was reported for various studies in humid regions (e.g. Petry et al. 2002; Tappin et al. 2013).

The choice of sampling frequency has significant implications for compliance testing and load estimates (see also Cassidy & Jordan 2011; Wade et al. 2012). The common monthly measurement frequency causes large uncertainties in the estimated summer average concentrations that are used for the compliance testing. To illustrate this uncertainty, we used the continuous concentration time series from the Hupsel catchment for April to October 2008. In 100,000 realizations, we randomly subsampled six snapshots from the continuous data. Figure 9 shows the range of possible summer average concentrations that may be estimated from the same concentration pattern. Given this large range, compliance of individual locations with the applicable EQS is partly coincidental. In our national assessment, the
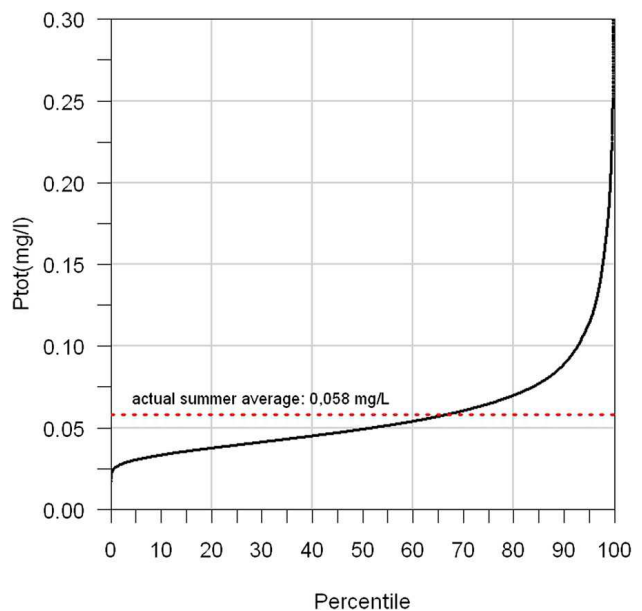

Fig. 9 Distribution of average summer P-tot concentration estimates from random subsampling of six snapshots from a continuous concentration dataset for May to September. The figure shows that common monthly measurement intervals may result in under- or overestimated summer average concentration estimates, which are used for EQS compliance testing

effects of the uncertainty for individual monitoring locations are probably reduced by aggregating the monitoring results and presenting group statistics.

Continuous monitoring using sensors or auto-analyzers or measuring average concentrations using passive samplers could reduce the uncertainties in the estimates of average concentrations and loads. A comparison of load estimates based on conventional grab sampling, continuous monitoring, and time-averaged passive sampling is given by Rozemeijer et al. (2010a). A general review on the driving processes of short-scale temporal variations in solute concentrations and the consequences for monitoring is given in Rozemeijer and Van der Velde (2014).

Trend analysis

Our trend analyses yielded downward-aggregated trends for both N-tot and P-tot concentrations from 87 trend locations in agriculture-dominated headwaters. Nutrient concentrations in agricultural headwaters decreased with median slopes of 0.55 mg/l per decade for N-tot and 0.020 mg/l per decade for P-tot. Van Puijenbroek et al. (2014) recently studied trends in ditches, canals, rivers, and lakes in The Netherlands between 1990 and 2010. Unlike our agriculture-specific headwaters, these waters were impacted by multiple sources. Still, Van Puijenbroek et al. (2014) found similar downward trends for N-tot and P-tot. However, they observed a stabilization of nutrient concentrations since 2004, while this study showed that the downward trends persist in the agriculture-specific headwaters (Fig. 6). The similarity in trend slopes suggest that agricultural inputs dominate nutrient concentrations in downstream surface waters. However, proving the causality between agricultural headwater nutrient exports and downstream water body concentrations and ecology is not possible using these national-scale datasets. To distinguish agricultural loads from other sources would need detailed monitoring of these water bodies and their tributaries and probably a water and nutrient balance or water and solute transport model to relate the input loads from different sources to concentrations in downstream waters. Sewage water treatment plant emissions are another important N-tot and P-tot source from which the emissions reduced during 1990–2010 (Van Puijenbroek et al. 2014). Another complicating factor is that many downstream surface waters in The Netherlands are influenced by the inlet of diverted river water from the Rhine and Meuse rivers (Roelofs 1991; Rozemeijer et al. 2012).

Using a water and nutrient transport model, Oenema & Roest (1998) predicted the effects of manure legislation in The Netherlands on P and N emissions from agriculture to surface water between 1985 and 2008. They predicted a decrease in N emissions by 38 % between 1985 and 2008, while the P emissions were not expected to decrease on the short term. Although these results are not directly comparable, Oenema & Roest (1998) seem to have slightly overestimated the reduction in N emissions and underestimated the reduction in P emissions.

Our results indicate a relatively quick water quality response to the manure legislation enforced in 1986 to reduce agriculture-related eutrophication of water resources. In a comparable study in Denmark, Kronvang et al. (2008) also found that N-tot concentrations in surface water in agricultural catchments started to decrease in the years directly after the implementation of national manure legislation. For Sweden, Kyllmar et al. (2006) also found decreasing trends in N-tot and P-tot concentrations and related this to recent changes in agricultural land use.

The relatively quick response of surface water quality to manure legislation contradicts with several studies that report long time lags between changes in land use and water quality effects in surface water (e.g. Howden et al. 2011). These delays are attributed to long mean transit times of infiltrated water through the subsurface and to the continuing contribution of legacy stores of N-tot and P-tot that have accumulated in soils over several decades of intensive agricultural land use (Stalnacke et al. 2003). In some cases, however, the detection of significant trends is hindered by natural interannual weather-related variability, combined with too short time series (Raïke et al. 2003). In addition, many surface waters are also affected by nonagricultural sources of nutrients, like sewage treatment plants, sewage overflows industrial spills, other urban sources, and atmospheric deposition.

The different response times of catchments can be explained within the conceptual framework proposed by Rozemeijer & Broers (2007). This concept states that stream water quality is the result of a varying mixing ratio of groundwater from different depth levels. Manure legislation will initially affect nutrient concentrations in near-surface flow routes towards the surface water system. Especially in lowland catchments with

shallow groundwater levels, near-surface flow routes may deliver an important fraction of the total water discharge and contaminant loads towards the surface water system. In many lowland agricultural areas, artificial drainage further enhances runoff through near-surface flow routes by tapping the upper groundwater. For example, a field study by Rozemeijer et al. (2010b) showed that subsurface tile drains contributed 80 % of the yearly water discharge and 90 % of the total nutrient and heavy metal loads from an agricultural field towards the surface water system. Several other studies also report accelerated transport of agrochemicals and reduced subsurface residence times caused by artificial drainage (Heathwaite & Dils 2000; Jaynes et al. 2001; Stamm et al. 2002; Frey et al. 2012). A large contribution of near-surface flow routes may explain the non-delayed responses to manure legislation in areas like The Netherlands (this study) and Denmark (Kronvang et al. 2008).

Conclusions

Trend analysis showed that water quality in agriculture-dominated headwaters in The Netherlands has improved over the period 1990–2010. The results show a relatively quick response to the introduction of manure legislation in 1986. Still, in 2007–2010, up to 76 % of the agriculture-dominated water quality monitoring locations in The Netherlands did not comply with either the EQS for N-tot or for P-tot. Results from detailed catchment-scale pilot studies improved our understanding of weather-induced year to year differences and other uncertainties that are involved in a regional-scale water quality status and trends assessment. An important observation is that testing compliance for nutrients based on summer average concentrations may underestimate the agricultural impact on ecosystem health of downstream water bodies. The focus on summer concentrations does not account for the environmental impact of high winter loads from agricultural headwaters towards downstream water bodies.

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