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Hydrology is the key factor for nitrogen export from tile-drained catchments under consistent land-management

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Abstract

In this study, 14 years of climate, stream flow, land management, nitrate-nitrogen (NO_3^--N) load and concentration data were analyzed to identify potential drivers for NO_3^--N losses at two tile-drained catchments under cropland use in northeastern Germany. Mean ($\pm$standard deviation) annual NO_3^--N concentrations were 9.7 ± 2.9 (drainage plot) and 6.8 ± 2.4 mg l^{-1} (ditch catchment), while mean annual NO_3^--N loads amounted to 22 ± 16 and 20 ± 16 kg ha^{-1}, respectively. Significant positive relationships between annual discharge and annual NO_3^--N losses underlined the importance of hydrologic conditions on NO_3^--N export mechanisms. No direct relationships were found between N soil surface surpluses and NO_3^--N losses. Any possible impact of N soil surface surpluses on NO_3^--N export rates was overridden by the hydro-meteorological conditions in the catchment. Positive correlations between the climatic water balance and NO_3^--N losses suggest that agricultural catchments with similar characteristics as ours may face—without countermeasures—increased N losses in the future as regional climate projections predict wetter winters in the coming decades. Our analysis has further shown that effects of land management strategies aiming at reducing N losses into surface waters might only become visible with a delay of years or even decades.

1. Introduction

Eutrophication due to the excess input of the major nutrients phosphorus (P) and nitrate (NO_3^-) is still a major concern for many surface water bodies in Europe. Several legislative initiatives at European Union (EC 1991, 2000) or at regional level (OSPAR 1992, HELCOM 2007) demand measures to achieve a good ecological status for inland and coastal waters. While the input of P has decreased substantially over the last three decades, mainly due to technical improvements of wastewater treatment plants, there is little progress for NO_3^- in Germany (UBA 2018). There is a general weak tendency of decreasing NO_3^- concentrations in German rivers, but concentrations in the coastal areas still remain on an unacceptably high level (BMUB 2017). The slow decrease of NO_3^- concentrations was mainly attributed to a decreasing tendency of N surplus between 1990 and 2014 due to reduced N fertilization, while N export from

the fields has gradually risen (BMUB 2017). There is also a tendency of decreasing NO_3^- concentrations in several major European river basins such as the Rhine, Odense or Thames, while other rivers such as the Loire exhibit an increasing trend although N surpluses generally decreased in all investigated catchments (Bouraoui and Grizzetti 2011). The different catchment responses were partly explained by different lag and storage times of NO_3^- both in unsaturated and saturated zones of soils and in aquifers (Bouraoui and Grizzetti 2011). A recent study for the Mississippi river basin suggested a strong legacy effect for nitrate, which means that N has accumulated in the soil for many years and that it will take decades until N concentration in the rivers decrease even if no N fertilizers are being applied anymore (van Meter *et al* 2016). Even for smaller catchments, the situation is neither clear, and relationships between N balances and N concentrations in surface waters are often difficult to find. In a review investigating

small agricultural catchments in the Baltic Sea region, 10 of 14 catchments showed no significant relationships between the annual N balance and N concentrations (Bechmann *et al* 2014). The reason for these weak relationships was primarily explained by long but uncertain lag times (Bechmann *et al* 2014).). At the parcel scale, where soil properties and flow path are comparatively homogenous, clear effects of N surpluses on N losses via drainage water could frequently be established (e.g. Jaynes *et al* 2001, Kladivko *et al* 2004, Vetsch *et al* 2019).

However, not only lag times make it hard to figure out causal relationships between N balances and NO_3^- concentrations in surface waters. The situation is complicated by the strong impact of hydrometeorological conditions on catchment response of solutes. Precipitation and discharge can be good predictors for NO_3^- concentrations. Regression models suggested that a large proportion of the variation in daily NO_3^- concentrations at the catchment outlet may be explained by a combination of precipitation and discharge (Feng *et al* 2013). Often, direct relationships exist between discharge and NO_3^- concentrations (Webb and Walling 1985, van Herpe and Troch 2000, Tiemeyer *et al* 2006, Bauwe *et al* 2015), which may hold true for daily, monthly, seasonal or annual time steps (Schilling and Lutz 2004). These relationships have implications on long-term nitrate trends as these will be affected by the long-term discharge behavior (e.g. Schilling and Lutz 2004). Other examples are agricultural water management practices such as tile drainage, which are thought to increase the water throughput, leading to an enhanced NO_3^- export (Raymond *et al* 2012). Tile drainage causes a shortcut between the topsoil and the surface water bodies and is thus regarded as a hotspot of aqueous N losses from agriculture (Tiemeyer *et al* 2006). However, not only hydrological processes and agricultural practices but also climate change with increased temperatures may alter the N turnover in the soil leading to increased N mineralization rates and consequently, increased NO_3^- losses (Wang *et al* 2016). To unmask the effects of changes of land management practices on NO_3^- concentrations in surface waters, Abbott *et al* (2018) suggested that long-term data with high temporal resolution are necessary (see for example Stålnacke *et al* 2014). However, monitoring programs in tile-drained catchments usually cover less than 10 years (Hirt *et al* 2011).

Starting in 1996, the German fertilizer regulation (DüV 1996, 2006, 2017) has tightened rules for maximum N surpluses from 90 kg N ha^{-1} in 2006 down to 50 kg N ha^{-1} from 2018 onwards to improve the quality of ground and surface waters. In this study, we used a unique long-term data set of 14 years comprising climate, land management, stream flow as well as NO_3^- concentration and export data using high-frequency sampling (i) to test whether any trends in N management are reflected in trends in annual NO_3^-

losses. In addition, we hypothesize that (ii) NO_3^-N losses are controlled by water balance rather than by N management. Finally, we aimed (iii) at identifying any ongoing hydrological change, which than might substantially influence NO_3^-N losses. We conducted our study in a small agricultural tile-drained catchment in northeastern Germany.

2. Materials and methods

2.1. Study area
The study area is located in northeastern Germany near the city of Rostock (figure 1). It is under conventional arable use (94%) and contains only a small proportion of forests (6%). The catchment is located in a glacially formed landscape with a flat topography and only gentle slopes. Mineral soils (mainly sandy loams) are characteristic for the area, with a predominance of Luvisols (48%), Gleysols (43%), and Stagnic Gleysols (9%). The four agricultural fields (1: 60.9 ha, 2: 59.2 ha, 3: 17.8 ha, 4: 21.0 ha) in the study area (figure 1) are tile-drained (drainage depth 1.1 m, drainage distance 8 to 22 m) to improve soil aeration and agricultural management. Water originating from the tile drains flows into a main ditch (figure 1). The drainage plot (field 4) and the ditch catchment (including fields 1, 2, 3, 4) are 4.2 ha and 179 ha in size, respectively. Mean annual precipitation was 659 mm and the mean annual temperature was 8.8 °C (2004–2017). Precipitation is mainly rainfall, but there were some winters with considerable snow cover (2009, 2010). Due to the high proportion of tile drainage (94%), discharge is mainly controlled by tile flow and groundwater flow (Bauwe *et al* 2016). In contrast, surface runoff is only rarely being observed during heavy storm events.

2.2. Land management
The field records of the farmer that included all agricultural measures were used to calculate N balances. Figure 2 provides an overview of the crop rotation during the investigation period. Typical crops were winter wheat (*Triticum aestivum* L.), winter barley (*Hordeum vulgare* L.), rapeseed (*Brassica napus* L.), and maize (*Zea mays* L.). Occasionally, sugar beet (*Beta vulga*ris L.) and legumes such as peas (*Pisum sativum* L.) or alfalfa (*Medicago sativa* L.) were also grown. With a strong focus on cereals, maize and rapeseed, the crop rotation is typical for conventional arable farming in northern Germany.

Annual N applications depended on the choice of crops (table 1). The N fertilizers were applied three to four times a year, mostly in mineral form. Winter wheat and winter barley were seeded usually in the second half of September, while rapeseed was seeded about one month earlier, and they were harvested in summer of the following year. Maize, sugar beet, and peas were seeded in late April and harvested in late summer or early fall during the same year. Alfalfa was
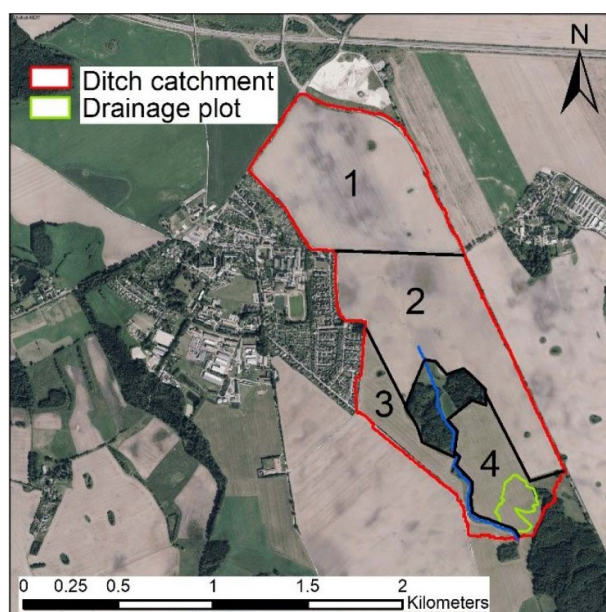
Figure 1. Study area including catchment boundaries, ditch (blue), and arable fields (numbers). Base map: © GeoBasis-DE/M-V.
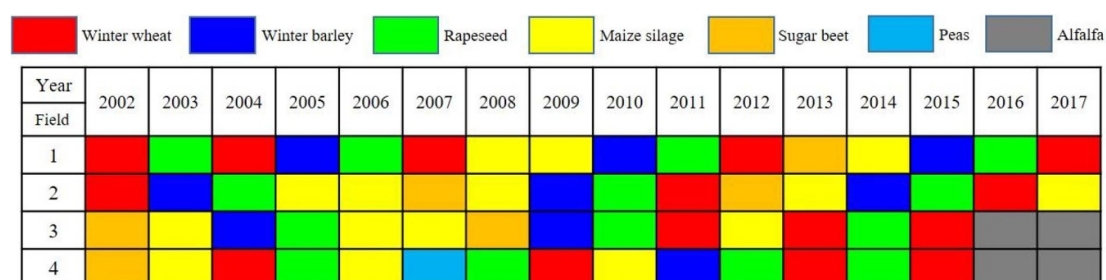

Figure 2. Crop rotation during the investigation period of the ditch catchment (comprising all four fields) and the drainage plot (field 4).

Table 1. Cultivated arable crops and crop management including seeding/harvesting months, N fertilization and yields. The yield is given as dry matter. ±indicate standard deviations.

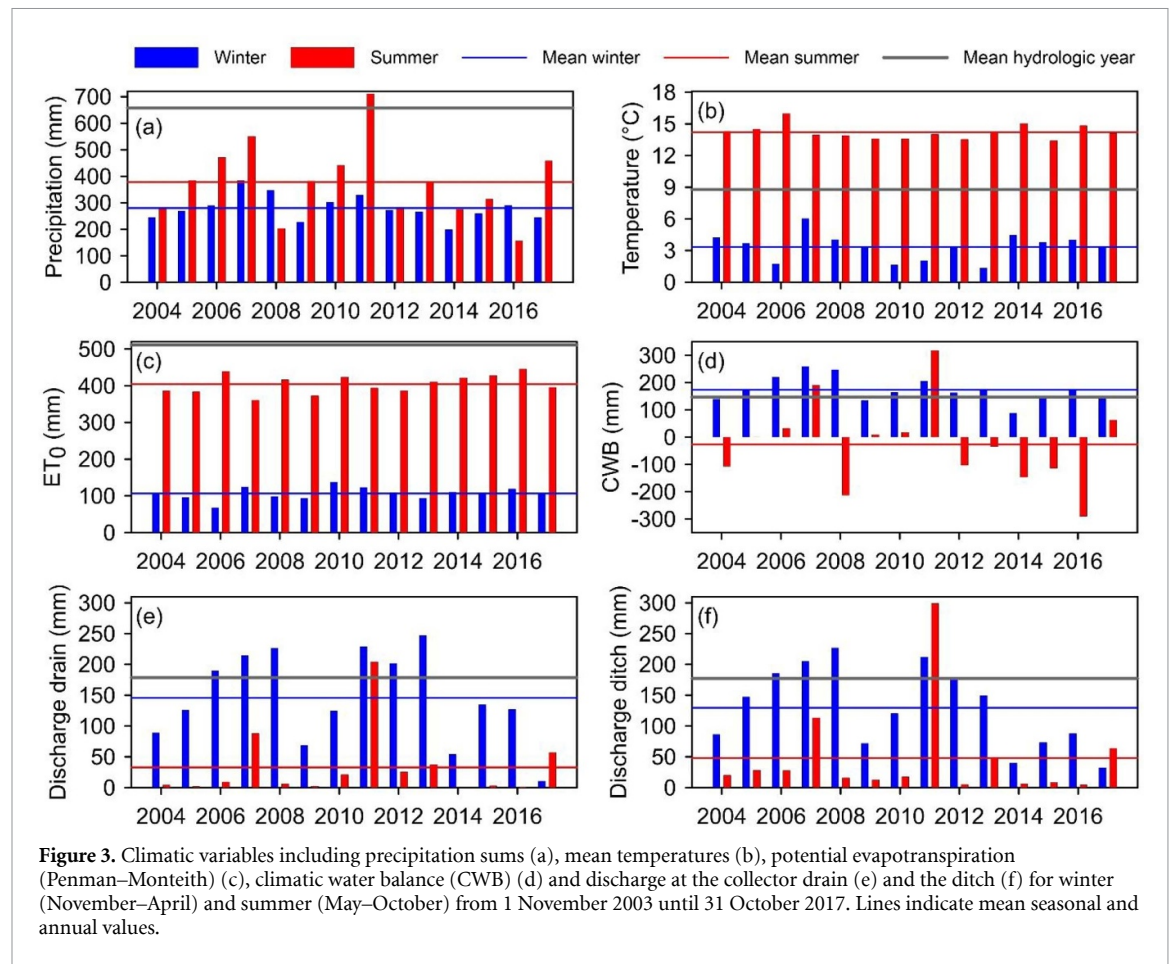| Crop	Seeding month	Harvest month	Fertilization (kg N ha^{-1})	Yield (t ha^{-1})
Winter wheat	September	August	221 ± 13	7.4 ± 0.9
Winter barley	September	July	176 ± 33	8.0 ± 0.5
Rapeseed	August	July	204 ± 32	4.4 ± 0.4
Maize silage	April	September	117 ± 59	14.3 ± 1.4
Sugar beet	April	October	170 ± 46	24.6 ± 4.9
Peas	April	August	0 ± 0	3.2 ± 0
Alfalfa	August	May[a] June August	33 ± 0	14.3 ± 0 14.3 ± 0 14.3 ± 0

[a]harvested three times a year

cultivated on two fields in 2016 (figure 3) and harvested three times per year. Table 1 provides also information about the yields for the individual crops.

2.3. Calculation of nitrogen soil surface surplus

N soil surface surpluses were calculated for all four fields in the catchment accounting for mineral and organic fertilization (accounting for ammonia volatilization that amounts to ca. 15% depending on fertilizer type), N fixation, and harvest as follows:

$$N_{sss} = N_{min} + N_{org} + N_{fix} - N_{yld} \qquad (1)$$

where N_{sss} is nitrogen soil surface surplus, N_{min} is amount of nitrogen in mineral fertilizer, N_{org} is amount of nitrogen in organic fertilizer accounting for ammonia volatilization, N_{fix} is amount of nitrogen fixed by legumes, and N_{yld} is amount of nitrogen in harvested yield. All units are (kg ha^{-1}). The N contents of the organic fertilizers, of the harvested crops as well as N fixation rates and losses by ammonia volatilization were taken from literature (MLUV-MV

Figure 3. Climatic variables including precipitation sums (a), mean temperatures (b), potential evapotranspiration (Penman–Monteith) (c), climatic water balance (CWB) (d) and discharge at the collector drain (e) and the ditch (f) for winter (November–April) and summer (May–October) from 1 November 2003 until 31 October 2017. Lines indicate mean seasonal and annual values.

2008, 2019). The N contents of mineral fertilizers are known from the manufacturer's specifications. The amounts and dates of fertilizer application and the crop yields of each field were provided by the farmer. N contents of organic fertilizers (cattle slurry and fermentation residues) were occasionally analyzed in the laboratory and fitted well with literature estimates (MLUV-MV 2008) used here. N fixation rates were relatively uncertain but leguminous uptake was only relevant in a few years. N soil surface surpluses of the drainage plot equal the N soil surface surpluses of field 4 (figure 1). N soil surface surpluses of the ditch catchment were calculated as area-weighted means of the four fields located within the catchment.

2.4. Data collection and analysis

A weather station at the catchment outlet recorded precipitation and air temperature. Precipitation was measured at a 0.1 mm resolution using a tipping bucket rain gauge. Since precipitation measurement errors vary from 10% to 25% depending on season and position of the rain gauges in northeastern Germany, data were corrected by applying corresponding correction factors (Richter 1995). In order to guarantee a high data quality, automatic precipitation recording was verified by weekly readings of a manual rain gauge.

The sampling station at the outlet of the drainage plot connected with the collector drain consists of a Venturi flume (RBC-flume, UGT GmbH, Germany) for continuous (15 min interval) and automatic water level measurement and an ISCO sampler (Teledyne Isco, Inc. Lincoln, NE) to collect water samples. Water level data were transferred into discharge data at the collector drain by applying regression equations provided by the manufacturer. The sampling station at the ditch located at the catchment outlet is equipped with an automatic, ultrasonic, water level measurement device (Teledyne Isco, Inc. Lincoln, NE) and an ISCO water sampler. We conducted frequent (usually once a week) discharge gauging with an inductive flowmeter (Flo-Mate TM, Marsh-McBirney, Inc. Frederick, MD) to develop rating curves for this station. Rating curves were used to calculate hourly discharge data using the automatic water level measurements and eventually transferred to daily values. Water samples for both stations were taken usually at least twice a week. After collecting the water samples, they were brought immediately into the laboratory and frozen at $-20\,^{\circ}$C until the analysis for NO_3^-. NO_3^- was analyzed by ion chromatography (Metrohm AG, Herisau, Switzerland). Ammonium (NH_4^+) concentrations were also measured between 2007 and 2008, but found to be non-detectable on almost all dates.

In order to analyze NO_3^-N concentrations, we calculated annual values as time-weighted mean concentrations (TWMC):

$$TWMC = \frac{\sum_1^n (c_i * t_i)}{\sum_1^n (t_i)} \qquad (2)$$

We used TWMC to identify potential relationships with discharge and the CWB. TWMC was used instead of arithmetic mean, because the frequency of water sampling differed throughout the year. To exclude the influence of flow, we additionally determined annual values as flow-weighted mean concentrations (FWMC) using the following equation:

$$FWMC = \frac{\sum_1^n (c_i * t_i * q_i)}{\sum_1^n (t_i * q_i)} \qquad (3)$$

In these two equations is c_i the NO_3^-N concentration on day i in mg l^{-1}, t_i is the time window for the ith sample in days, and q_i is discharge on day i in mm.

Daily NO_3^-N loads for the drainage plot and the ditch catchment were calculated by multiplying streamflow and NO_3^-N concentrations. Then, due to the strong relationship between load and streamflow, reliable regression equations using (measured) NO_3^-N loads as functions of streamflow could be established. These regression equations were used to calculate NO_3^-N loads for days, at which no measured NO_3^-N concentrations were available. Finally, daily NO_3^-N loads were summed up to annual values. Annual values for hydro-meteorological data and NO_3^-N data refer to a hydrologic year (November until October).

We tested the statistical significance of temporal trends by using the Mann-Kendall trend test. All statistical analyses were performed using R software (R Development Core Team 2011) including package trend version 1.1.1 (Pohlert 2018).

3. Results and discussion

3.1. Hydro-meteorological conditions
Seasonal precipitation showed strong variations over the course of the study period. 2011 (712 mm) and 2007 (551 mm) experienced the wettest summers (May–October), while 2016 (156 mm) and 2008 (204 mm) were very dry (figure 3(a)). Seasonal temperatures (figure 3(b)) and potential evapotranspiration (figure 3(c)) varied over the study period, but the CWB (figure 3(d)) is largely driven by precipitation. Thus, the summers of 2011 and 2007 showed an extraordinarily positive CWB. Winter (November–April) precipitation and CWB did not fluctuate that much. Probably due to the limited length of our hydro-meteorological time series, we could not yet find any trends such as projected wetter winters and presumably drier summers for Central Europe (Frei *et al* 2006, Beniston *et al* 2007, Kjellström *et al* 2011).

The CWB controlled to a large degree discharge rates at the collector drain and the ditch. For example, a rising CWB in winter between 2004 and 2008 led to increased discharge rates in the same seasons. Extremes are, again, the summers 2007 and 2011, in which an extraordinary high water balance led to exceptional high discharge rates at both measurement stations.

3.2. Discharge and nitrate concentrations
The discharge at the drainage plot outlet and in the ditch showed a similar and strong seasonal pattern (figure 4). The main discharge period was the winter (figure 3). In summer, the collector drain was often dry. Major runoff events usually fell into the winter period. Nevertheless, the highest ever recorded daily discharge rate occurred in July 2011 and amounted to 27 mm in the ditch (28 mm at the drainage outlet), which was induced by an extraordinary summer storm event of 98 mm rain per day. The onset of tile drainage was strongly linked to the CWB of the proceeding summer, causing e.g. a very late onset in 2016.

The NO_3^-N concentrations at the drainage outlet and in the ditch followed mostly a similar pattern (figure 4). As the drainage pipes fall dry in summer, very low concentrations such in the ditch are not reached. This concurrent behavior is an indication that NO_3^-N dynamics at the smaller scale (drainage plot) control to a large degree the NO_3^-N dynamics at the larger scale (ditch catchment).

Overall, the NO_3^-N concentrations followed the discharge dynamics (figure 4). NO_3^-N concentrations usually increased when stream flow rates increased and decreased with the falling limb of the hydrograph. This pattern was apparent over the entire observation period and can probably be attributed to (I) long-term N accumulation in the soil, (II) mineralization and nitrification processes even during winter, and (III) the mobilization of nitrate-rich soil water of the upper soil layers by the tile drains (Tiemeyer *et al* 2006).

Mean annual (±standard deviation) TWMC of NO_3^-N at the drainage outlet and the ditch were 9.7 ± 2.9 mg l^{-1} and 6.8 ± 2.4 mg l^{-1}, respectively, while mean FWMC were 11.7 ± 3.6 mg l^{-1} and 10.1 ± 2.7 mg l^{-1}, respectively. Following the general pattern of NO_3^-N and discharge, NO_3^-N concentrations were high in wet and low in dry years (figure 4). In particular, in winter (figure 5) and during wet periods in summer, NO_3^-N concentrations frequently exceeded the drinking water limit of 11.3 mg l^{-1} according to the EU-Nitrate directive (EC 1991). Under wet conditions, when the groundwater table is high, nitrate enriched in upper soil layers in concert with tile drainage induce high NO_3^-N concentrations in the receiving brook. This mechanism of nitrate mobilization is not unusual and corresponds exemplarily to findings of a Dutch study, in which a
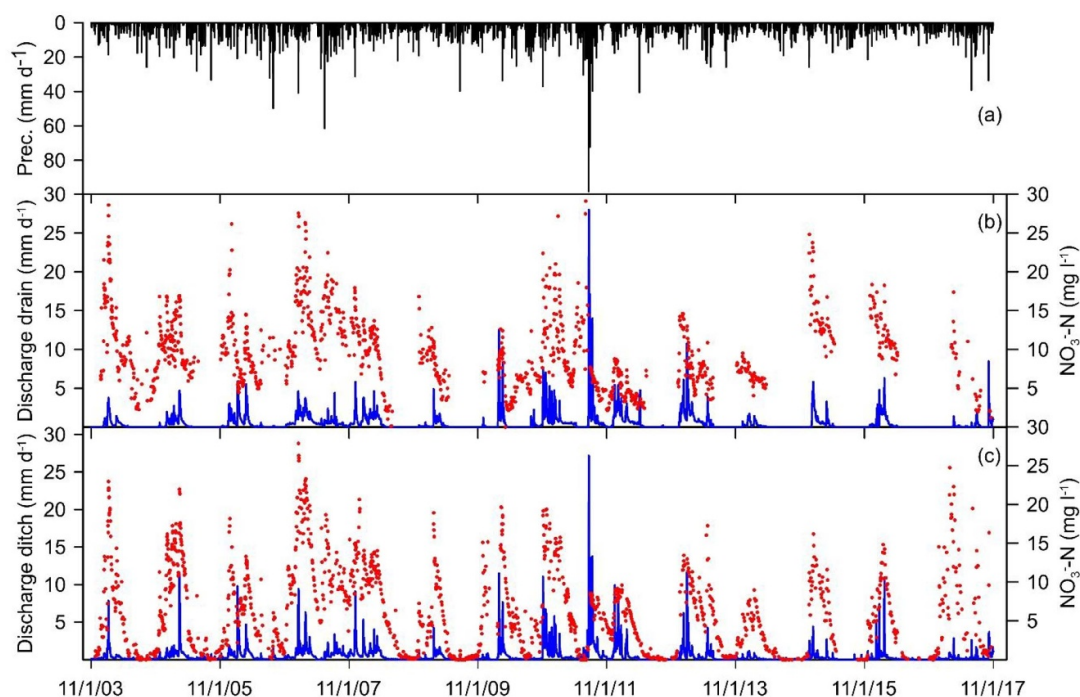
Figure 4. Daily precipitation sums (a), discharge (blue), and NO₃⁻N concentrations (red) at the collector drain (b) and the ditch (c) from 1 November 2003 until 31 October 2017.
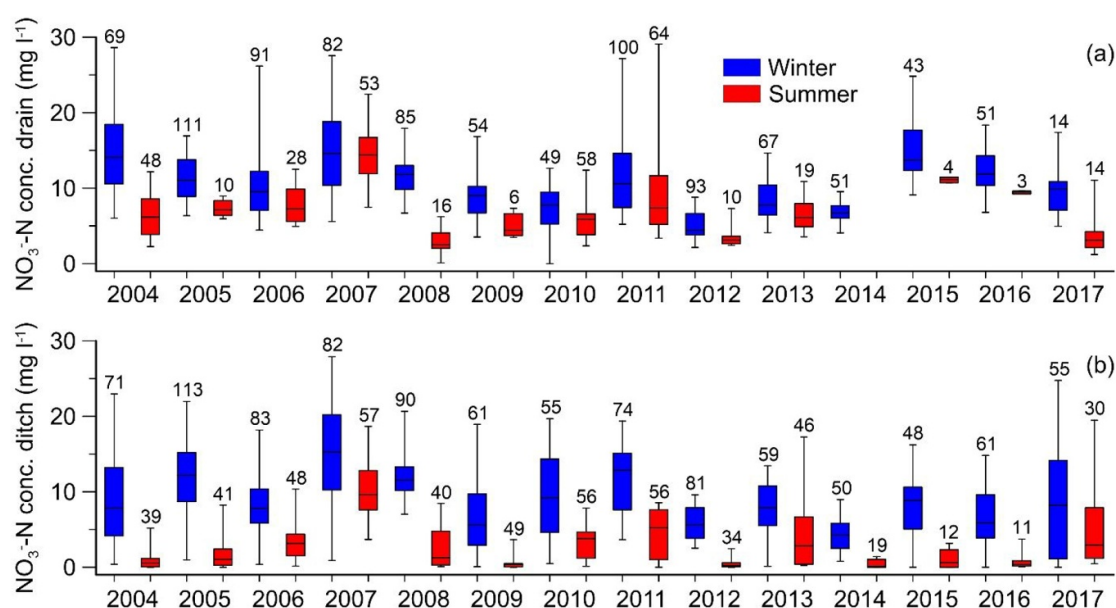

Figure 5. Summarized NO₃⁻N concentrations at the collector drain (a) and the ditch (b) for winter (November–April) and summer (May–October) from 1 November 2003 until 31 October 2017. The boxes indicate number of samples, maximum, 75% quartile, median, 25% quartile, and minimum.

general increase of NO₃⁻N concentrations in agricultural streams at times of high precipitation have been demonstrated (Rozemeijer and Broers 2007).

Highest NO₃⁻N concentrations were measured in winter, while NO₃⁻N concentrations were lower in summer (figure 5). During the growing period (March–October), the applied N fertilizers were

quickly taken up by the growing crops. N concentrations were therefore low during this period except for extraordinary wet summers (2007, 2011). In fall, soil moisture increases, but the temperature is still high enough (figure 3) for soil microbial activity, while plant growth comes to a halt. Thus, ongoing mineralization of organic N and decreased N plant uptake
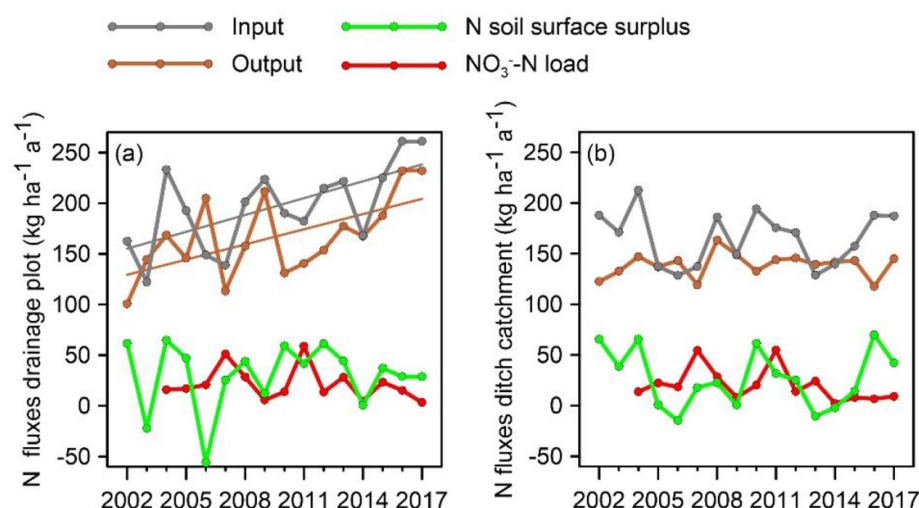
Figure 6. Annual nitrogen input, output, and soil surface surplus and NO_3^--N loads for the drainage plot (a) and the ditch catchment (b). The straight lines indicate significant trends ($p < 0.05$).

are thought to result in N-enriched soil water that is susceptible to leaching. Consequently, arable soils are most prone to N leaching in temperate regions after harvest in fall and winter (Di and Cameron 2002).

3.3. Nitrogen balances and nitrate losses

Due to higher input than export by harvest, N soil surface surpluses were mostly positive from 2002 until 2017 (figure 6). N input in the ditch catchment ranged from 129 kg ha^{-1} a^{-1} to 213 kg ha^{-1} a^{-1}. N input for field 4 (figure 1) and thus for the drainage plot ranged from 122 kg ha^{-1} a^{-1} to 261 kg ha^{-1} a^{-1}. N output by harvest ranged from 118 kg ha^{-1} a^{-1} (101 kg ha^{-1} a^{-1} for field 4) to 163 kg ha^{-1} a^{-1} (232 kg ha^{-1} a^{-1}) for the ditch catchment. These numbers are typical for conventional farming practices in northeastern Germany. For Mecklenburg-Western Pomerania, average values of 207 kg ha^{-1} a^{-1} for N input were calculated for arable land and the time span 2001–2008 (MLUV-MV 2009) and N soil surface surpluses were 76 kg ha^{-1} a^{-1}. For our study area, mean N soil surface surpluses were lower and amounted to 32 kg ha^{-1} a^{-1} for the drainage plot and 28 kg ha^{-1} a^{-1} for the ditch catchment, respectively. Data of N input, output and N soil surface surpluses during the investigated time period were less variable for the ditch catchment, since four fields of different cultivation (figure 2) instead of one (drainage plot) were taken into account.

Over the course of the study period, both the N input as well as the N output increased significantly for field 4 (drainage plot), while no trend was observed for the entire ditch catchment. This results in more or less stable N soil surface surpluses for the considered time period. The missing trend in N soil surface surpluses coincides with missing trends in N losses. Our findings are in line with nationwide

estimates revealing both increased N inputs and outputs from 1995 to 2017, while N soil surface surplus remained nearly the same (UBA 2019). The temporal stable N soil surface surplus probably resulted in almost constant NO_3^--N loads (figure 6) and concentrations (figures 4 and 5) over time.

Adding the atmospheric N deposition (https://gis.uba.de/website/depo1/) to the cumulative N balances over 14 years and subtracting cumulative aqueous NO_3^--N losses leaves a surplus of 333 kg N (drainage plot) and 237 kg (ditch catchment) in the cumulative total N balance (table 2). Therefore, N might still be accumulated in the soils. On the other hand, N might have been lost by other processes, i.e. mainly denitrification. Furthermore, the calculation of the total N balance is inherently uncertain due to the magnitude of ammonia volatilization and assumptions on leguminous N uptake. As organic fertilizers were applied via trailing hose and mainly at temperatures below 10 °C, we estimated ammonia losses of 10.5% for fermentation residues and 17.6% for cattle slurry, respectively (KTBL 2018, MLUV-MV 2019). However, the effect of these uncertainties is limited as only 16% of all N fertilizer applied over the whole study period was applied in organic form, and 5% was assigned to leguminous uptake.

3.4. Relationship between N soil surface balance, hydrology and NO_3^--N losses

No relationships were found between NO_3^--N loads or concentrations and N soil surface surpluses (figure 7). Furthermore, a possible positive relationship of the last years' land management on the actual NO_3^--N loads or concentrations could not be established. This holds true both for the drainage plot and the ditch catchment.

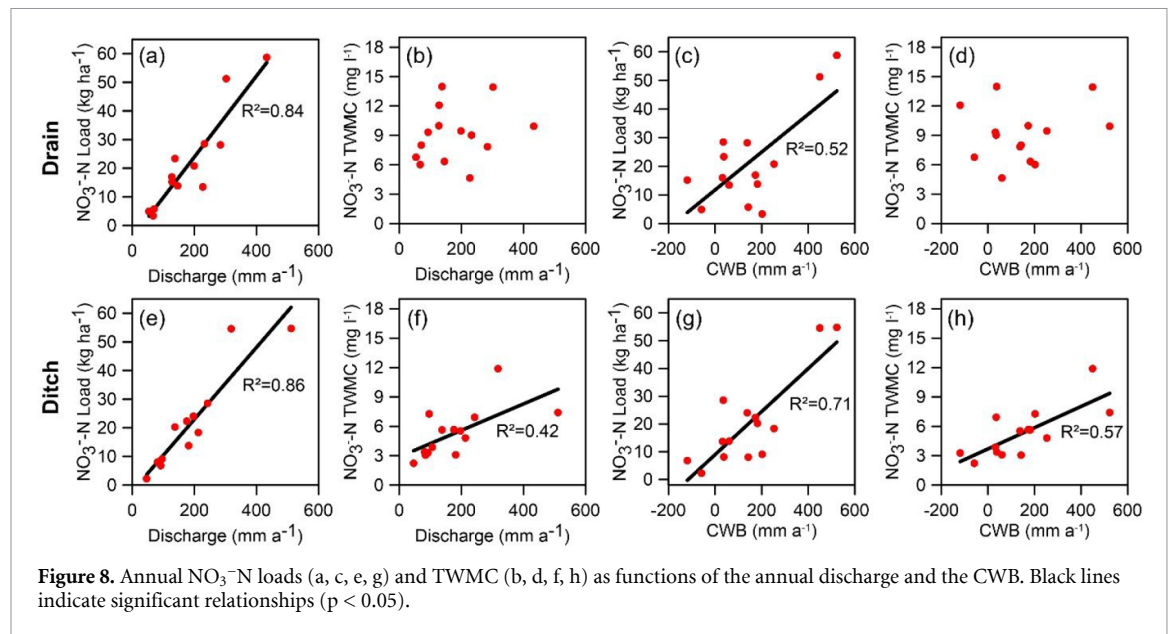A missing direct relationship between the N soil surface surplus and annual NO_3^--N losses does not

Table 2. Mean annual (±indicate standard deviation) and cumulative components of the total nitrogen (N) balance for the period 2004–2017.

	Drainage plot Mean (kg N ha^{-1} a^{-1})	Sum (kg N ha^{-1})	Ditch catchment Mean (kg N ha^{-1} a^{-1})	Sum (kg N ha^{-1})
Mineral and organic fertilizer (incl. N fixation)	+204 (±36)	+2863	+164 (±26)	+2293
Deposition	+14 (±0)	+196	+14 (±0)	+196
Harvest	−173 (±35)	−2424	−141 (±11)	−1968
Aqueous losses	−22 (±16)	−302	−20 (±16)	−284
Mean or sum over 14 years	**+23 (±32)**	**+333**	**+17 (±11)**	**+237**
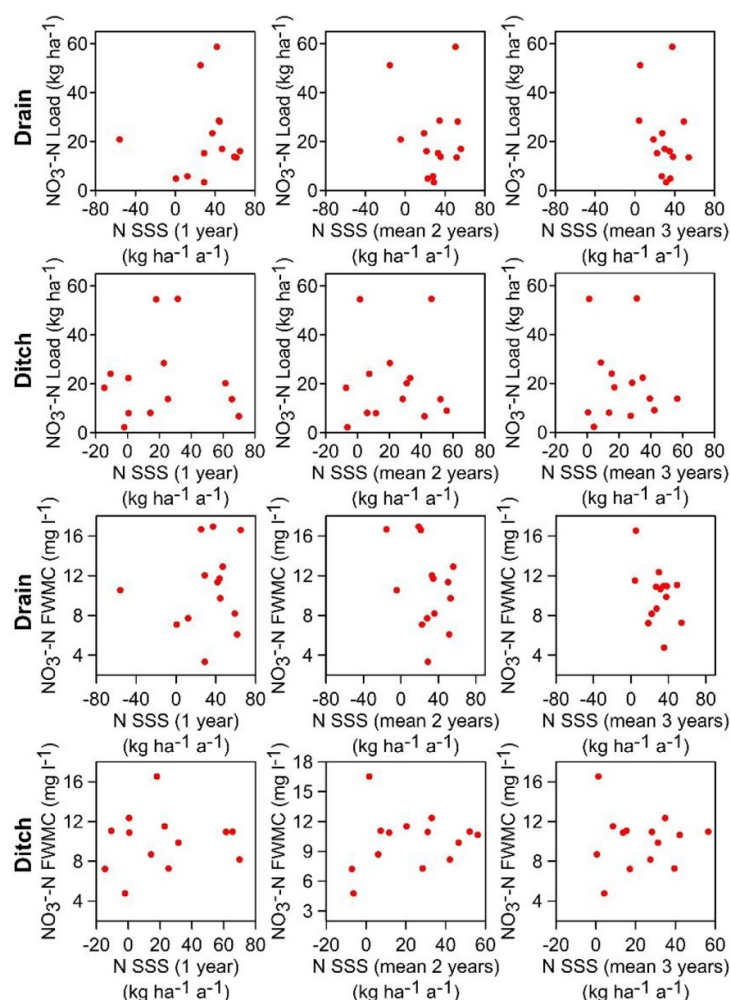
Figure 7. Annual NO_3^-–N loads and FWMC as functions of the N soil surface surplus (SSS) of the current year (1 year), of the mean of the current and the previous year (mean 2 years), and of the current and the previous two years (mean 3 years) for the drainage plot and the ditch catchment.

necessarily mean that land management has no effect on N export in the long run. Long-term soil-N enrichment impedes an amelioration of surface water quality and a change in land management with reduced N surpluses probably only affects the chemical constitution of stream water after long lag times or after a drastic change of management practices, which was not the case in our study area. Assuming consistent discharge rates over time, NO_3^-N loads will only be reduced, if NO_3^-N concentrations decrease. However, there is evidence that stream NO_3^-N concentrations may not be affected for decades after fertilizer

inputs have been reduced at least if large-scale river basins are considered (van Meter *et al* 2016). The so-called legacy N (van Meter *et al* 2016), which is slowly released to waters makes it so difficult to find causal relationships between actual land management practices and NO_3^-N concentrations and/or NO_3^- loads. Lag times for smaller catchments may be shorter (Kyllmar *et al* 2014) and it has been proven that nitrogen reduction measures have had a positive effect on NO_3^-N concentrations in small Danish and Swedish catchments (Stålnacke *et al* 2014), while increasing NO_3^-N concentrations in Estonia and Latvia were

Figure 8. Annual NO_3^--N loads (a, c, e, g) and TWMC (b, d, f, h) as functions of the annual discharge and the CWB. Black lines indicate significant relationships ($p < 0.05$).

attributed to the increased use of N fertilizers since the early 2000s (Bechmann *et al* 2014). Probably, differences in fertilizer application without a significant temporal trend might be too small not to be superimposed by hydro-meteorological conditions, especially as the study was conducted not as a strictly controlled parcel experiment, but under 'real-world' farming conditions. Thus, it might be difficult to disentangle effects of hydro-meteorological conditions and management choices such as timing of fertilizer application in a (fairly) diverse crop rotation where some crops (peas, winter barley) were only grown once within 14 years. Thus, we could not discern any patterns in the relationship between N soil surface surplus and NO_3^--N loads related to e.g. the duration of bare soil or the growth of winter cereals compared to other crops (data not shown).

In contrast to N soil surface surplus, there was as hypothesized a strong significant positive relationship between annual discharge and annual NO_3^--N loads both at the drainage plot and in the ditch (figures 8(a) and (e)). A positive relationship between discharge and NO_3^--N load was expected, since the calculation of loads includes discharge. However, the strength of this relationship with R^2 values >0.8 was surprising, but could be explained by the positive relationships between discharge and NO_3^--N concentrations (figures 4, 8(b) and (f)). While there is only a tendency for higher mean annual NO_3^--N concentrations at higher discharge at the drainage plot, this relationship is statistically significant in the ditch.

Ongoing climatic change possibly gradually alters the hydrological processes within the catchment with consequences for the transport of solutes, although we could not yet establish trends within our relatively short time series. NO_3^--N loads correlated significantly positively with the CWB (figures 8(c) and (g)). Relationships between NO_3^--N concentrations

and CWB are weaker but, nevertheless, show a positive and at the ditch a significant correlation. Long-term past climate studies revealed increasing winter rainfall in Central Europe, while trends for summer are less clear (Hänsel *et al* 2009, Zolina 2014, Murawski *et al* 2016). The results of these (past) climate studies are consistent with climate projections predicting wetter winters and presumably drier summers for Central Europe (Frei *et al* 2006, Beniston *et al* 2007, Kjellström *et al* 2011). Assuming the climate forecasts prove to be true, it is likely that, on the long run, flow rates will increase in winter resulting in increased NO_3^--N loads and concentrations (figure 8) as long as there is still a surplus of N in the system and unless no further countermeasures are taken.

4. Conclusions

A 14 years data set consisting of weather data and high-frequency discharge and NO_3^--N loads and concentration records as well as comprehensive land management data were used to detect the main drivers for NO_3^--N losses in a small agricultural tile-drained catchment. Data suggest that hydrology largely controls the dynamics and magnitudes of NO_3^--N concentrations and NO_3^--N loads. Therefore, we conclude that the magnitude of NO_3^--N loads and concentrations can be primarily attributed to discharge rates at a seasonal or annual time scale.

The N soil surface balance could not be directly linked to the N losses, but neither of them showed temporal trends. On the basis of individual years, the impact of N soil surface surplus was overridden by the hydrologic conditions. The results indicate that a study period of 14 years with nearly constant N management is presumably still too short to disentangle effects of—in our case quite diverse—land management and hydrology on NO_3^--N loads

and concentrations even in small catchments. Still, NO_3^-N concentrations are on a high level (15% of the data exceeded the drinking water limit) despite average N soil surface surpluses have been well below regulatory demands as well as below regional and national averages for years. The analysis of the data has further shown that, due to possible enrichment of soil N, effects of a land management aiming at reducing N losses into surface waters might only become visible with a delay of years or even decades and the evaluation of the effects of any policy measures will demand patience.

In the light of regional climate projections predicting wetter winters in the coming decades, agricultural catchments with similar characteristics as ours may face increased N losses unless countermeasures are established. However, our time-series is still too short to discern any trends in hydro-meteorological conditions. Thus, the study also highlights the dire need for long-term data sets, which will be needed to disentangle the—possibly counteracting—effects of land management and climate change on both biogeochemical and hydrological processes.

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Data availability statement

All data that support the findings of this study are included within the article (and any supplementary information files (available online at stacks.iop.org/ERL/15/094050/mmedia)).

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