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Insights gained from two decades of intensive monitoring: hydrology and nitrate export in a tile-drained agricultural catchment

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Nitrate (NO_3^-) export from agricultural land poses an ongoing threat to both inland and coastal waters. Experimental studies investigating the hydrology-NO_3^--export mechanisms require long-term data to identify reliable causal relationships. In this study, utilizing a 23-year continuous dataset with a high temporal resolution (daily to twice a week), we aim to identify potential drivers for NO_3-losses and assess the impact of nitrogen (N) soil surface budgets on NO_3-export. A drainage plot (4.2 ha) and a ditch catchment (179 ha) were fully equipped to register hydrological parameters, including water sample collection. Mean annual NO_3^--N concentrations (loads) for the drainage plot and the ditch catchment were 9.4 mg l^{-1} (20.6 kg ha^{-1}) and 6.0 mg L^{-1} (20.9 kg ha^{-1}), respectively. Annual discharge was closely positively correlated with annual NO_3-losses, highlighting the significant influence of prevailing weather and, consequently, hydrologic conditions on NO_3-export rates. The majority of the annual NO_3^--load was exported during winter (56% at the drainage plot, 51% at the ditch catchment), while the rest was exported during spring (28, 29%), summer (9, 9%) and fall (7, 11%). We could not find any direct relationships between N soil surface budgets and NO_3-losses. Putting all results together, it can be concluded that agricultural activities for many decades resulted in high soil N stocks, which determined the general high NO_3^--N concentration levels. Nevertheless, temporal NO_3-export dynamics during the last two decades were clearly driven by hydro-meteorological conditions, nearly independently of land management and N soil surface budgets on the fields.

KEYWORDS

agriculture, nitrate, nitrogen, long-term monitoring, soil surface budget, tile drainage agriculture, tile drainage

1 Introduction

Nutrient export from agricultural land is continuing to threaten our inland and coastal waters. To achieve a good ecological status for surface water bodies, numerous directives and action plans have been established at both the European Union level (EC, 1991, 2000) and regionally (OSPAR, 1992; HELCOM, 2007). Nitrogen (N) pollution remains elevated and ecologically unacceptable in various regions worldwide (UBA, 2018; Oelsner and Stets, 2019; Yu et al., 2019; Malone and Newton, 2020).

Compelling evidence indicates that long-term N fertilizer application has resulted in soil N enrichment in numerous agricultural watersheds (Van Meter et al., 2016; Ascott et al., 2017; Chen et al., 2018). Owing to positive N soil surface budgets, N enrichment in agricultural soils is likely to be continued in the near future. However, a modeling study examining the evolution of N surplus in Europe for more than a century identified a gradual decline in N surplus in European soils in the recent past (Batool et al., 2022). The reasons for this decline are undoubtedly multifaceted, with administrative regulations playing a pivotal role. For instance, in its efforts to counteract excessive N export into surface water bodies, Germany has progressively tightened regulations concerning annual N surplus, reducing it from 90 kg N ha^{-1} in 2006 to 50 kg N ha^{-1} from 2018 onwards (DüV, 1996, 2006, 2017). These administrative measures appear to be gradually having a positive effect on surface water quality. For example, there is a weak tendency of decreasing nitrate-nitrogen (NO_3^--N) concentrations in German rivers (BMUB, 2017). At European level, a decrease in NO_3^--N concentrations can be observed in some rivers (Rhine, Odense, Thames), while NO_3^--N concentrations continue to rise in other large rivers (e.g., Loire) (Bouraoui and Grizzetti, 2011). The different trends were partially explained by different NO_3-lag times (Bouraoui and Grizzetti, 2011).

NO_3^--N concentrations and loads in streams are controlled by a variety of factors. One important factor is land use. For example, NO_3^--N concentrations are usually lower in forested catchments compared to agricultural catchments (Mayer et al., 2002; Poor and McDonnell, 2007; Barnes and Raymond, 2010), in which N is added in form of mineral and/or organic fertilizers. Hydro-meteorological conditions also play a crucial role in driving NO_3^--N concentrations and export patterns in streams. Many studies have demonstrated a close positive correlation between discharge and NO_3^--N loads (e.g., Tomer et al., 2003; Bauwe et al., 2020). Tomer et al. (2003) reported for a tile-drained watershed in Iowa, USA, that NO_3^--N concentrations were not typically diluted by large flows resulting in high NO_3^--N loads during discharge events. They concluded that practices to optimize N management within fields should be encouraged, because edge-of-field measures such as constructed wetlands cannot effectively remove NO_3^--N during large flows (Tomer et al., 2003). Furthermore, NO_3^--N concentrations often exhibit seasonal variations depending on discharge composition. An Austrian study has shown that alternating aquifer contributions within a year controlled to a large degree the seasonality of the NO_3^--N concentrations in the streams due to significantly higher NO_3^--N concentrations in the shallow aquifer compared to the deep aquifer (Exner-Kittridge et al., 2016). At the event scale, NO_3^--N concentration patterns present a nuanced picture, with scientific literature reporting both dilution effects (Inamdar et al., 2004; Feinson et al., 2016) and accretion effects (Fučík et al., 2012; Bauwe et al., 2015).

A limited investigation period is often an obstacle in experimental studies to identify long-term trends or to uncover relations between hydrology and nitrate export patterns. In 2001, we fully equipped a 4.2 ha drainage plot and a 179 ha ditch catchment for the purpose of monitoring hydrological parameters, including water sample collection. The continuous monitoring effort has yielded a large continuous data set with a high temporal resolution (daily to twice a week) spanning over 22 years (2001–2022). Here, we use this extensive dataset to identify potential drivers for NO_3-losses and investigate the impact of N soil surface budgets on NO_3-export.

2 Materials and methods

2.1 Study site

The study area is situated in northeastern Germany, approximately 10 km southeast of Rostock (Figure 1). Characterized by conventional arable use covering 94% of the region, the landscape features minimal forested areas, accounting for only 6%. The catchment is located in a glacially formed landscape with a flat topography and only gentle slopes (slope gradients <3%). 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 in the study area (Figure 1) are tile-drained with a drainage depth 1.1 m and a drainage distance from 8 to 22 m.

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. Tile drain flow is most pronounced during the winter months, extending from November to April. This temporal pattern corresponds to a precipitation surplus attributable to lower temperatures and reduced evapotranspiration rates during this period. Mean annual precipitation was 614 mm and the mean annual temperature was 9.1°C (2001–2022). While precipitation is predominantly in the form of rainfall, certain winters experienced substantial snow cover, notably in 2009 and 2010. Due to the high areal proportion of tile drainage (94%), discharge in the ditch is mainly controlled by tile flow and groundwater flow (Bauwe et al., 2016). In contrast, surface runoff is only sporadically being observed during intense storm events.

2.2 Hydrological data

The sampling station positioned at the outlet of the drainage plot, linked to the collector drain, is equipped with a Venturi flume (RBC-flume, UGT GmbH, Germany). This apparatus enables continuous and automatic water level measurement at 15 min

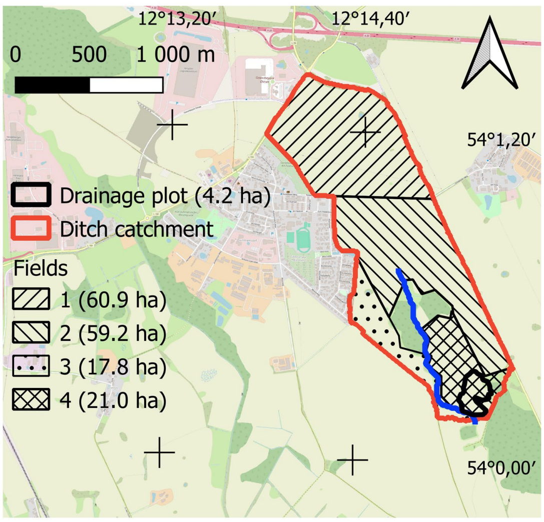

FIGURE 1
Study area.

TABLE 1 Main crops, N fertilization and N in yields.

Crop	Number of cropping years				Fertilization (kg N ha^{-1})	N yield (kg N ha^{-1})
	Field 1	Field 2	Field 3	Field 4		
Winter wheat	6	4	5	6	214 ± 22	171 ± 23
Winter barley	4	3	2	1	180 ± 10	153 ± 14
Rapeseed	5	4	4	5	211 ± 32	150 ± 21
Maize silage	6	8	5	4	135 ± 50	165 ± 25

±Indicate standard deviations.

intervals. The recorded water level data are subsequently converted into flow data at the collector drain through the application of regression equations supplied 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). We conducted frequent flow measurements, typically on a weekly basis, utilizing an inductive flowmeter (Flo-Mate TM, Marsh-McBirney, Inc., Frederick, MD) to develop rating curves for this station. Rating curves were used to calculate daily flow data using the automatic water level measurements.

2.3 Nitrate data

Water samples from both the drainage plot and the ditch were systematically collected using ISCO samplers (Teledyne Isco, Inc., Lincoln, NE), typically a minimum of twice a week. After collecting the water samples, they were promptly transported to the laboratory and frozen at −20°C to preserve their integrity until the subsequent analysis for NO_3^--N. The analysis of NO_3^--N was conducted through ion chromatography, utilizing equipment from Metrohm AG, Herisau, Switzerland. To derive daily, monthly and annual NO_3^--N loads for both the drainage plot and the ditch catchment, we used the R software package *RiverLoad* (Nava et al., 2019). This package provides different methods for load estimation, such as averaging methods, ratio estimators and regression methods. Based on our dataset with continuous flow data and instantaneous concentration data, load estimation using linear interpolation of concentration data was appropriate for this study:

$$L = K'' \sum_{j=1}^{n} C_j^{\text{int}} Q_j \tag{1}$$

where L is the NO_3^--N load, C_j^{int} (g m^{-3}) is the daily NO_3^--N concentration linearly interpolated between two measured samples, Q_j (m^3 s^{-1}) is the mean daily streamflow and K'' is a conversion factor for the period of load estimation (Moatar and Meybeck, 2005).

It is pertinent to note that the annual values for both hydrological data and NO_3^--N data pertain to a hydrologic year spanning from November to October.

2.4 Nitrogen soil surface budget

Field records from the local farmer, encompassing detailed agricultural practices, served as the foundation for calculating annual N soil surface budgets from 2001 to 2022. The crop rotation during the investigation contained common crops such as winter wheat (*Triticum aestivum* L.), winter barley (*Hordeum vulgare* L.), rapeseed (*Brassica napus* L.), and maize (*Zea mays* L.) (Table 1). Periodically, sugar beet (*Beta vulgaris* L.) and legumes such as peas (*Pisum sativum* L.) or alfalfa (*Medicago sativa* L.) were also cultivated. This crop rotation is typical for conventional arable farming in northern Germany. Annual N applications varied based on crop selection, as detailed in Table 1. N fertilizers, primarily in mineral form, were applied three to four times a year. Winter wheat and winter barley were typically sown in the latter part of September and harvested in the subsequent summer. Rapeseed was sown about a month earlier, and its harvest coincided with that of winter wheat and winter barley. Maize, sugar beet, and peas were sown in late April and harvested in late summer or early fall of the same year. Table 1 additionally provides information on N in yields for the main crops.

N soil surface budget was calculated for all four fields in the catchment accounting for mineral and organic fertilization, N fixation, and harvest as follows:

$$N_{sur} = N_{min} + N_{org} + N_{fix} + N_{atm} - N_{yld} \tag{2}$$

where N_{sur} is N soil surface budget, N_{min} is amount of N in mineral fertilizer, N_{org} is amount of N in organic fertilizer accounting for ammonia volatilization, N_{fix} is amount of N fixed by legumes, N_{atm} is amount of atmospheric N deposition, and N_{yld} is amount of N in harvested yield. All units are (kg ha^{-1}). Data on the N content of organic fertilizers, harvested crops, N fixation rates, and losses due to ammonia volatilization were sourced from literature (MLUV-MV, 2008, 2019). The atmospheric N deposition was 14 kg ha^{-1} year^{-1} for the study catchment.[1] The N content of mineral fertilizers was obtained from manufacturer specifications. Fertilizer application details and crop yields for each field were provided by the farmer. Notably, the N soil surface budgets of the drainage plot align with those of field 4 (Figure 1), while those of the ditch catchment were computed as area-weighted means of the four fields within the catchment.

3 Results and discussion

3.1 Flow regime and nitrate concentrations

Streamflow and NO_3^--N concentrations at both the collector drain and the ditch exhibited pronounced dynamics throughout the study

1 https://gis.uba.de/website/depo1/

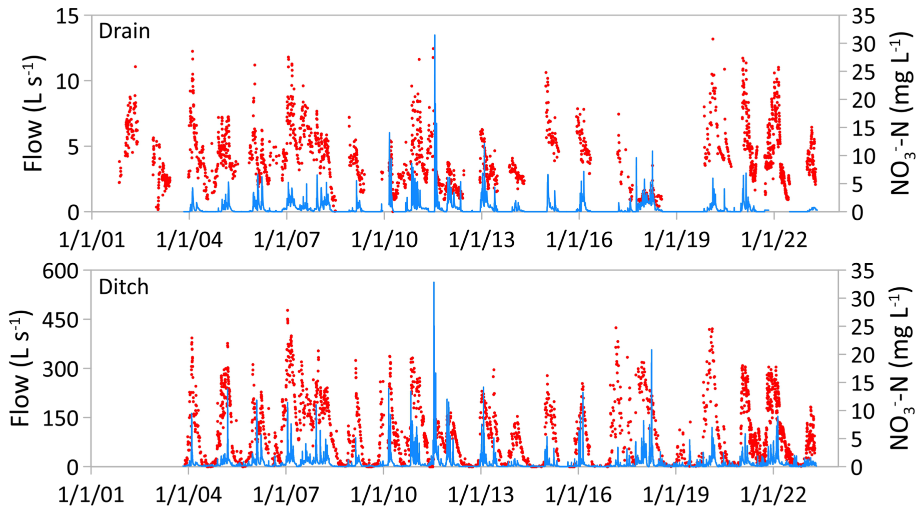

FIGURE 2
Daily flow rates (blue lines) and measured NO_3^--N concentrations (red dots) at the collector drain (above) and in the ditch (below) throughout the study period. NO_3^--N measurements at the collector drain began on November 4, 2001, and measurements at the ditch, along with flow measurements at both stations, commenced on November 1, 2003. Data for this study were analyzed until April 30, 2023.

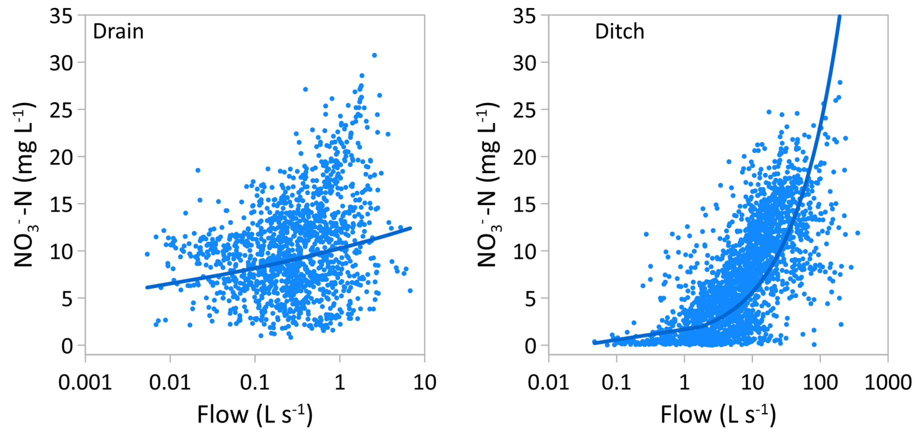

FIGURE 3
Relationships between NO_3^--N concentrations and flow rates at the collector drain (left) and the ditch (right).

period (Figure 2). Peak flows and maximum annual NO_3^--N concentrations predominantly occurred during the winter, while both streamflow and NO_3^--N concentrations approached near-zero levels during the summer months. NO_3^--N concentration levels were comparable at the collector drain and in the ditch during winter. Mean flow rates during the winter months (November until April) at the collector drain and in the ditch were 0.4 and 15.5 L s^{-1}, respectively. Conversely, the collector drain often experienced dry conditions in summer (May until October), with an exception in the summer of 2011 when an extraordinary precipitation event (98 mm day^{-1}) led to substantial flow. Mean flow rates in the ditch during summer were relatively low at 5.1 L s^{-1}. Although NO_3^--N concentrations were typically below the drinking water limit of 11.3 mg L^{-1} in summer (mean values of 8.0 and 3.1 mg L^{-1} at the collector drain and in the ditch, respectively), they frequently exceeded this limit in winter (Figure 2) (mean values of 10.8 and 9.0 mg L^{-1} at the collector drain and in the ditch, respectively).

The general trend of low NO_3^--N concentrations in summer and high NO_3^--N concentrations in winter (Figure 2) can be attributed to the activation of different N sources. During the summer, characterized by low groundwater tables, it is reasonable to assume that mainly groundwater, with low NO_3^--N concentrations, contributes to the flow. In contrast, during winter when the water table rises, NO_3^--rich water from the upper soil layers significantly influences the chemical composition of the discharge. This seasonal NO_3^--N pattern in surface waters of agricultural catchments aligns with findings described by various researchers, including Pionke et al. (1999), Dupas et al. (2016), Exner-Kittridge et al. (2016), Ford et al. (2018) and Abbott et al. (2018).

Over our observation period, no discernible trends were identified regarding the magnitude of NO_3^--N concentrations. The visual interpretation of Figure 2 and the plotting of NO_3^--N concentrations as a function of flow (Figure 3) revealed a positive relationship between flow and NO_3^--N concentrations. This positive relationship between streamflow and NO_3^--N concentrations was

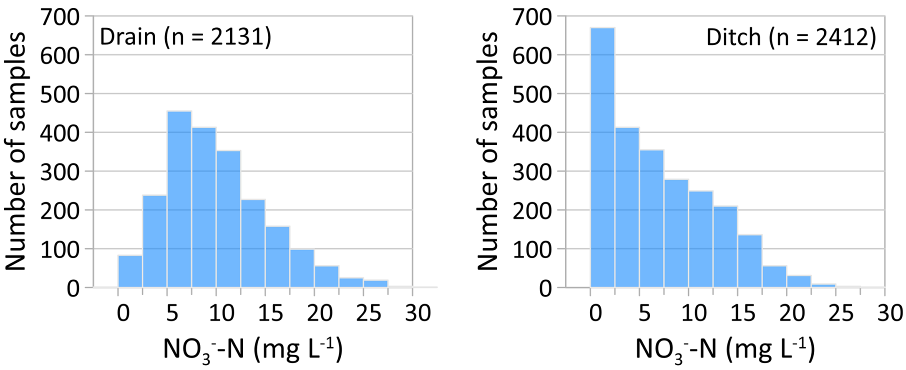

FIGURE 4
Histograms of NO_3^--N concentrations at the collector drain (left) and the ditch (right).

more pronounced in the ditch (Figure 3). This can be attributed primarily to water flow from groundwater with very low NO_3^--N concentrations in the ditch during summer, whereas the discharge season at the collector drain is typically confined to the winter, during which NO_3^--N concentrations from drainage water are generally higher. The different NO_3^--N concentration pattern of the collector drain and the ditch will be visible when plotting the NO_3^--N concentrations as histograms (Figure 4). The main difference between both figures is the large number of samples exhibit NO_3^--N concentrations from 0 to 5 mg L^{-1} in the ditch. These samples were taken in the vast majority of cases in summer, when there was no water in the collector drain. Excluding these low concentration samples, both histograms are similar. Only, a small number of samples (104 at the drain collector and 44 in the ditch) contained NO_3^--N concentrations beyond 20 mg L^{-1}.

In summary, both flow patterns and the dynamic behavior of NO_3^--N concentrations at the collector drain and in the ditch exhibit a high temporal parallelism (Figure 2), emphasizing the significance of the artificial drainage system in discharge generation. Furthermore, it identifies the drainage system as a notable source of NO_3-losses at larger spatial scales such as ditches, which confirms an earlier study conducted in the same study area analyzing a 3-years period of time (Tiemeyer et al., 2006).

3.2 Discharge and nitrate loads

NO_3^--N loads (Equation 1) were clearly driven by discharge. This holds true both for a daily and for an annual evaluation (Figure 5). The relationship between flow and NO_3^--N losses was even stronger compared to NO_3^--N concentrations. Annual NO_3^--N loads varied greatly. Highest annual NO_3^--N loads were observed at the collector drain (50 kg ha^{-1}) and at the ditch (52 kg ha^{-1}) in 2007 followed by very high loads in 2011, while NO_3^--N loads were lowest in 2019. In that year, no NO_3^--N was transported via the collector drain, since the collector drain did not carry any water and annual NO_3^--N load was 1 kg ha^{-1} at the ditch. These 2 years reflect extreme weather conditions. The high NO_3^--N loads in 2007 and 2011 were mainly caused by very wet conditions in summer, while the low NO_3^--N loads in 2019 were a result of a prolonged drought period in 2018. Mean annual loads at the collector drain (19.7 kg ha^{-1}) and the ditch (19.9 kg ha^{-1}) were similar.

Due to the strong relationship between flow and NO_3^--N loads it is not surprising that NO_3^--N loads follow a pronounced seasonal pattern (Figure 6). High discharge rates in winter, in particular in January, February and March, caused high NO_3^--N loads during that time. Consequently, highest monthly NO_3^--N loads were recorded in January, both at the collector drain (4.7 kg ha^{-1}) and at the ditch (4.4 kg ha^{-1}), while NO_3^--N loads were lowest in October at the collector drain (0.2 kg ha^{-1}) and in September at the ditch (0.3 kg ha^{-1}).

A significant amount of NO_3^--N was leached during a short period of time and the majority of NO_3^--N losses were restricted to winter. For example, the extraordinary high rainfall event (98 mm) on July 23, 2011 led to 10% of the annual NO_3^--N load being discharged within the next 7 days. To underpin the importance of seasonality of NO_3^--N leaching, it should be noted that 65 and 61% of the annual NO_3^--N load is being between January and March at the collector drain and the ditch, respectively (Figure 6). These NO_3^--N export mechanisms can be illustrated when the daily cumulative NO_3^--N load starting with the highest load is plotted against time (Figure 7).

When analyzing Figure 7, it is striking that the majority of the NO_3^--N load is being discharged during a very short period of time. Twenty-seven days (out of 4,259 discharge days) with the highest NO_3^--N loads were enough to contribute 10% of the total load at the collector drain. Even more extreme, 22 days out of 6,679 discharge days contributed to 10% of NO_3^--N load at the ditch. 50% of the total NO_3^--N load was discharged, when 10 and 5% of the considered time period has passed at the collector drain and the ditch, respectively. At the other end of the spectrum, it can be stated that only 10% of the total NO_3^--N load is being discharged in 50 and 75% of the total time at the collector drain and the ditch, respectively. The graph of the collector drain in Figure 7 is flattened compared to the graph of the ditch. This is mainly because there is no flow and therefore no NO_3^--N leaching at the collector drain in summer with very low NO_3^--N loads. Generally, the range of the NO_3^--N loads at the collector drain is smaller compared to the ditch due to missing low summer values.

3.3 Nitrogen soil surface budget and nitrate export

From 2001 to 2022, The N input at the drainage plot ranged from 40 kg ha^{-1} $year^{-1}$ to 275 kg ha^{-1} $year^{-1}$, while the N input varied from

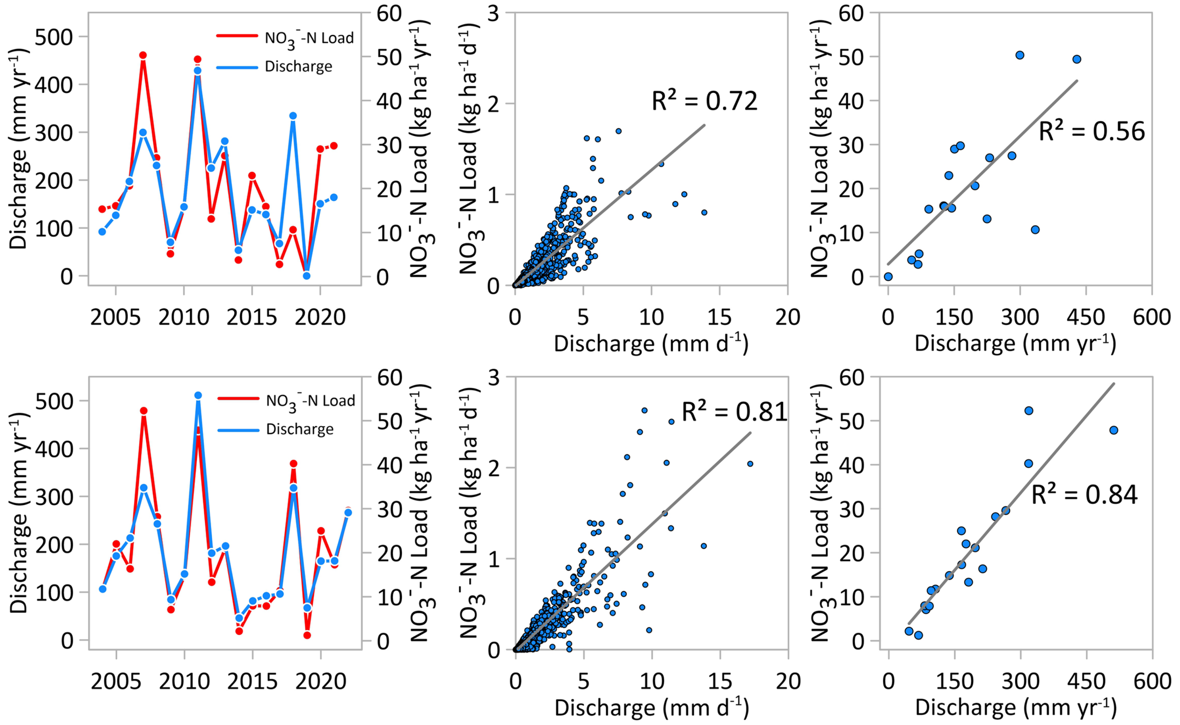

FIGURE 5
Annual discharge and NO_3^--N loads at the collector drain and the ditch and daily and annual relationships between these two parameters.

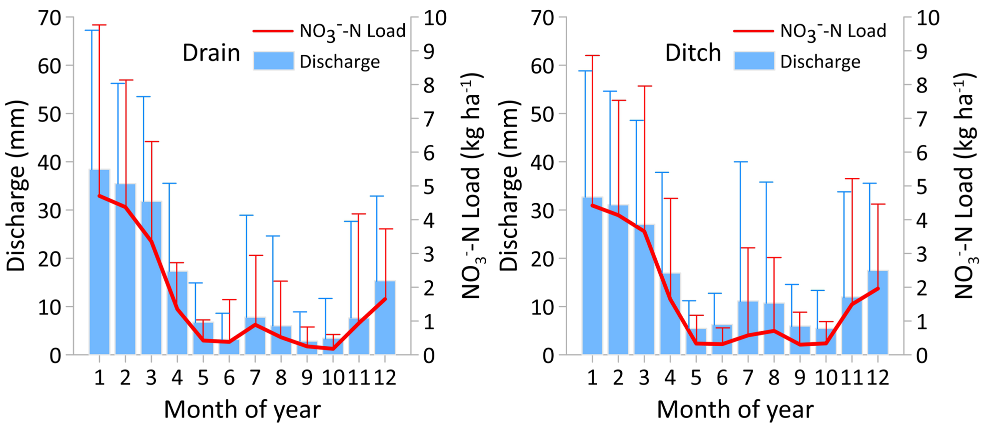

FIGURE 6
Mean monthly discharge and NO_3^--N loads at the collector drain and the ditch. The vertical lines indicate standard deviations.

109 kg ha^{-1} year^{-1} to 228 kg ha^{-1} year^{-1} in the ditch catchment. N output ranged from 26 to 232 kg ha^{-1} year^{-1} and from 99 to 163 kg ha^{-1} year^{-1} for the drainage plot and the ditch catchment, respectively. These values align with typical conventional farming practices in northeastern Germany. N soil surface budgets (Equation 2) in our study area mostly exhibited positive values due to higher input than export through harvest (Figure 8 and Table 1). Notably, the ditch catchment exhibited less variability in N input, output, and soil surface budgets, benefiting from the inclusion of four fields with different cultivation practices instead of one single field (drainage plot). When the cumulative N soil surface budgets and NO_3^--N losses via the collector drain and the ditch over the course of the study period are subtracted, a surplus of 295 kg N (drainage plot) and 268 kg N (ditch catchment) is evident. This suggests a long-term accumulation of N in the soils despite the fact that, based on the prevailing soil types and impact of groundwater, ca. 10–30 kg N ha^{-1} year^{-1} may have been lost through denitrification (Kreins et al., 2010). NO_3^--rich soil water is potentially available for leaching and becomes active when the groundwater table rises. This probably explains the observed seasonal patterns of NO_3^--N concentrations and the positive relationships between flow and NO_3^--N concentrations in our study catchment. The significance of groundwater head in nitrate export has been highlighted in previous studies (Musolff et al., 2016).

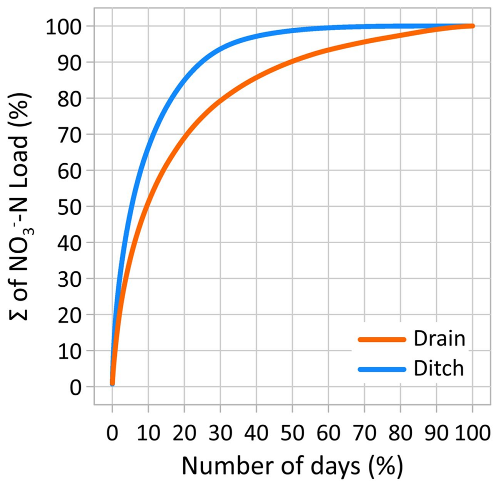

FIGURE 7
Daily cumulative NO_3^--N loads sorted starting with the highest load as a function of time.

NO_3^--N loads were completely independent from the N management on the arable fields and highest loads were observed during the wettest years 2007 and 2011 (Figure 8). In contrast to findings in other studies (Larsson and Jarvis, 1999; Ford et al., 2018; Liu et al., 2022), our results confirm earlier studies of our working group, in which no direct relationships between the N soil surface budget and N export (Bauwe et al., 2020) was identified. Contrarily, NO_3-export in our study catchment and also in other catchments of the Baltic Sea Region (Bechmann et al., 2014) appear to be entirely independent of N fertilizer application rates and is solely driven by hydro-meteorological factors.

3.4 Implications for catchment management

The data analysis in this study had two significant findings. Firstly, the previously described seasonal fluctuations in NO_3^--N concentrations and discharge had profound implications for NO_3-export. Considering the entire time frame, the majority of NO_3^--N has been lost via the collector drain or the ditch during a very short period (Figure 7). Secondly, no direct relationships between N soil surface budget and NO_3-export were discerned at either the drainage plot or the ditch (Figure 8). Although the N soil surface budget generally exhibited a decreasing trend, this reduction did not correlate with changes in NO_3^--N concentrations (Figure 2).

These findings have implications for a catchment management aiming at mitigating NO_3^--N losses. Efforts should be intensified to further reduce the N surplus. While there is an overarching trend in Germany and other European countries toward gradually decreasing agricultural N surplus (Batool et al., 2022), it is important to recognize that the positive impact on riverine NO_3-export may take many years to materialize. The predominant reason for this prolonged delay, known as the "N legacy effect," refers to the enduring accumulation of N in the root zone (Van Meter et al., 2016). The long-term N accumulation in the upper soil layers might also explain, why there are no functional seasonal relationships between agricultural activities such as N fertilization and NO_3^--N concentrations in our study area. In contrast, there are instances, in which NO_3^--transport in the river was affected by seasonal changes of the N soil system budgets (Pinardi et al., 2022). Under climate change conditions, both NO_3^--N export mechanisms and biogeochemical cycles may change, which makes predictions regarding NO_3^--N losses difficult. On the one hand, expected wetter winters in Central Europe (Frei et al., 2006) will probably lead to increased NO_3^--N loads because of elevated discharge in winter. On the other hand, rising temperatures will presumably result in higher denitrification rates (de Klein et al., 2017), which will might have a positive effect on NO_3^--N concentrations and loads in surface waters. For example, a downward trend in N export during the last three decades was observed for the Po River in Italy, and it was associated with an upward trend in water temperature along with enhanced denitrification rates (Soana et al., 2024).

To expedite the transition to a sustainable N management in agricultural catchments, technical measures should be implemented. Given that artificial drainage systems exert significant control over the hydrology and hydrochemistry of larger scales (Tiemeyer et al., 2006), reduction measures should primarily target drainage plot outlets. Effective strategies may include the establishment of constructed wetlands (Kovacic et al., 2000; Crumpton et al., 2020; Mitchell et al., 2023) or the implementation of reactive barriers (Christianson et al., 2012, 2021; Mitchell et al., 2023). Both approaches also have the potential to mitigate high flows, thereby positively influencing NO_3^--N concentrations during discharge events. However, in the scientific literature it is also argued that conservation practices (less fertilization, crop rotation etc.) possibly lead to faster success than controlled drainage, bioreactors, or saturated buffers (Liu et al., 2022).

Because of remaining uncertainties, probably the best approach for a sustainable N management in agricultural catchments is the implementation and strengthening of conservation practices together with technical solutions in hot spot areas.

4 Conclusion

In conclusion, our two-decade-long intensive monitoring of hydrology and NO_3-export in a nested tile-drained agricultural catchment has provided insights into the complex dynamics of N losses from agricultural land. The main findings of this investigation can be summarized as follows:

(1) Seasonal NO_3-export: the study revealed distinct seasonal patterns in NO_3^- concentrations and export, with elevated levels during winter months. NO_3-export dynamics were closely linked to hydro-meteorological conditions, which underlines the significant influence of weather and hydrologic factors on NO_3-losses. The majority of the annual NO_3^- load was transported within a few days, highlighting the episodic nature of NO_3-export in the catchment.
(2) Limited influence of N soil Surface budgets: surprisingly, no direct relationships were found between N soil surface budgets and NO_3-losses. The temporal dynamics of NO_3-export were primarily determined by hydro-meteorological conditions.
(3) Implications for catchment management: the findings underscore the challenges in mitigating NO_3-losses from agricultural land, suggesting that reductions in N surplus may take years to have a positive effect on NO_3^- concentrations due to the "N legacy effect." Effective catchment management

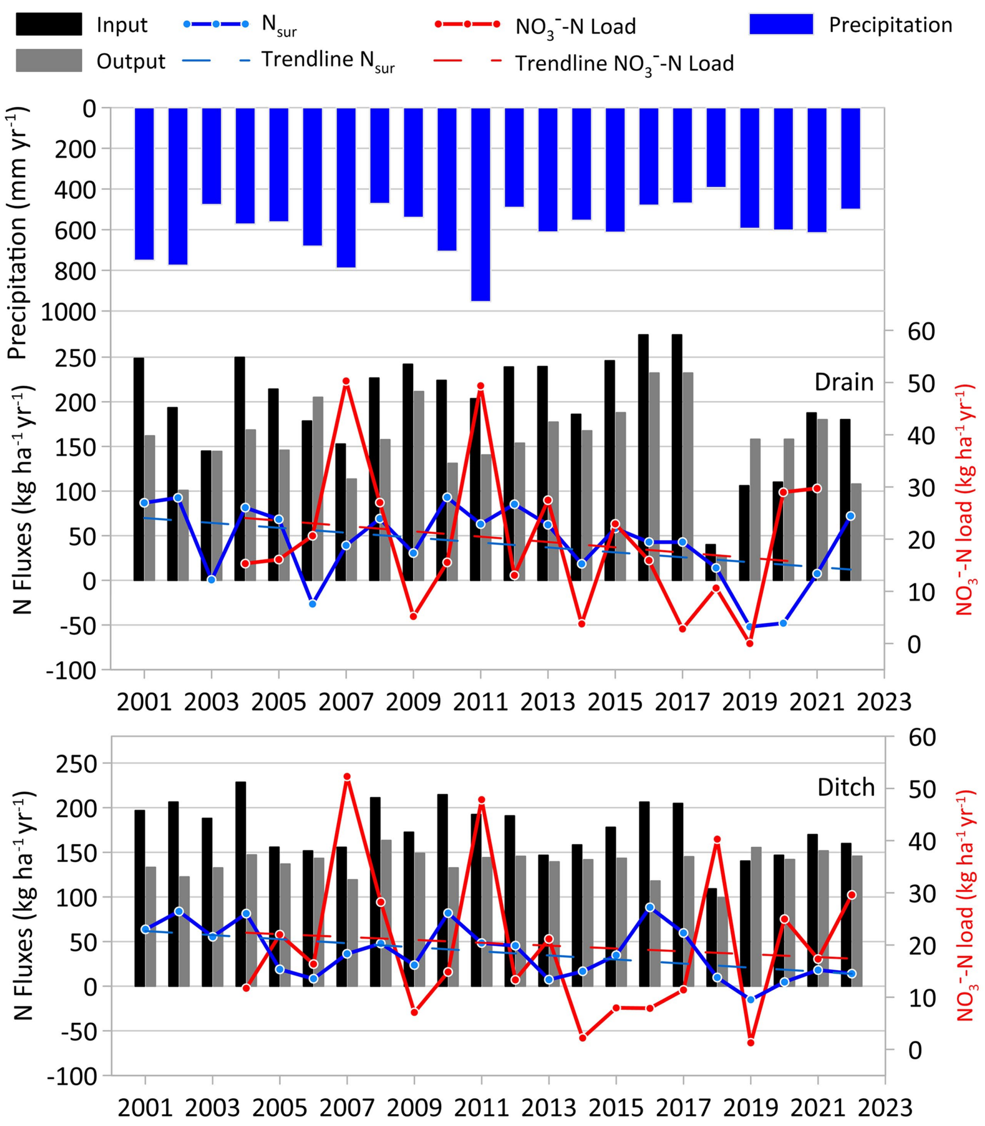

FIGURE 8
N input, N output, N soil surface budget (N_{sur}) and NO_3^--N loads for the drainage plot (above) and the ditch catchment (below) and annual precipitation.

strategies should consider the impact of artificial drainage systems on NO_3-export and focus on hot spot areas, possibly through the implementation of technical measures such as constructed wetlands or reactive barriers, in conjunction with conservation practices.

Data availability statement

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

Author contributions

AB: Conceptualization, Formal analysis, Methodology, Writing – original draft. BL: Conceptualization, Supervision, Writing – review & editing.

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Conflict of interest

The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

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