Acta Agriculturae Scandinavica, Section B — Soil & Plant Science

ISSN: 0906-4710 (Print) 1651-1913 (Online) Journal homepage: www.tandfonline.com/journals/sagb20

Nutrient leaching and water quality from agricultural observation fields during 50 years of environmental monitoring in Sweden

Lisbet Norberg, Maria Blomberg, Helena Linefur & Katarina Kyllmar

To cite this article: Lisbet Norberg, Maria Blomberg, Helena Linefur & Katarina Kyllmar (2025) Nutrient leaching and water quality from agricultural observation fields during 50 years of environmental monitoring in Sweden, Acta Agriculturae Scandinavica, Section B — Soil & Plant Science, 75:1, 2597105, DOI: 10.1080/09064710.2025.2597105

To link to this article: https://doi.org/10.1080/09064710.2025.2597105

ACTA AGRICULTURAE SCANDINAVICA, SECTION B — SOIL & PLANT SCIENCE
2025, VOL. 75, NO. 1, 2597105
https://doi.org/10.1080/09064710.2025.2597105

Taylor & Francis
Taylor & Francis Group

OPEN ACCESS Check for updates

Nutrient leaching and water quality from agricultural observation fields during 50 years of environmental monitoring in Sweden

Lisbet Norberg, Maria Blomberg, Helena Linefur and Katarina Kyllmar

Department of Soil and Environment, Swedish University of Agricultural Sciences, Uppsala, Sweden

ABSTRACT

Throughout the last 50 years, 16 agricultural fields in Sweden have been monitored for nutrient leaching losses and water chemistry via subsurface drainage water and groundwater. The fields represent various climatic regions, soil types, and farming practices. Twelve fields were equipped with groundwater pipes. Typically, the shallow (1–2 m) groundwater varied due to weather and agricultural management, whilst the deeper (4–5 m) groundwater stayed at a steadier level. Long-term mean annual losses of total nitrogen (N) ranged between 2.8 and 58.0 kg N ha^{-1} yr^{-1} across the 12 fields operated with flow-proportional water sampling. This variation was primarily due to the location in Sweden, as higher losses of N could be monitored in the southern part compared to further north in the country. For the same 12 fields, the mean annual losses of total phosphorus (P) ranged between 0.01 and 1.48 kg P ha^{-1} yr^{-1}. Here, the variation was mainly due to soil type, with higher losses from clay soils and lower losses from sandy soils, but with apparent exceptions. The well-known sites with long-term monitoring that the observation fields offer is a profound opportunity for assessing and understanding current nutrient losses and making predictions of water quality in the face of climate change.

ARTICLE HISTORY
Received 27 August 2025
Accepted 26 November 2025

KEYWORDS
Cropping systems; groundwater; nitrogen; Ph; phosphorus; subsurface drainage water; water chemistry

Introduction

Non-point sources of nutrients from agriculture contribute to eutrophication of the Baltic Sea area and inland water bodies and affect the quality of groundwater. To reduce eutrophication, both nitrogen (N) and phosphorus (P) mitigation measures must be mutually considered (Conley et al. 2009) and the water quality requires monitoring on a long-term basis to examine the impact of the implemented measures. Boesch (2002) concluded that negative impacts on the Baltic Sea area, e.g. excess N supply, anoxic zones, reduced water transparency, increased throughout 1950-1980. During this period, the agricultural sector intensified its use of fertilisers and in Sweden, animal production was centred around certain areas. In the late 1960s, the question was raised about how this would affect the losses of nutrients to surface-, drainage-, and ground water. A report concluded that knowledge about leaching losses from agricultural fields in Sweden was absent (Brink and Gustafson 1970). With financial support from Swedish authorities, the first agricultural observation fields were constructed in the early 1970s. The agricultural fields were prepared for subsurface water flow measurements and water sampling by delimiting the tile drainage system to a known area, thereby interconnecting and leading the tiles to an underground bunker.

During the 1970s and 1980s, a total of 19 fields were included in the monitoring programme. Today, 13 of the fields are still operating. The fields were distributed across Sweden to represent the wide range of climatic zones. Various soil types and agricultural management practices were factors involved in the selection of suitable locations, along with interested and agreeable landowners and farmers. In 1988, a field with both subsurface drainage and surface runoff measurements was included (Norberg et al. 2022). Measurements in subsurface drainage water as well as in groundwater at shallow (<2 m) and deeper (>2 m) depths, enable studies to investigate how nutrients move in the soil profile and to what depth the soil water is affected

CONTACT Lisbet Norberg lisbet.norberg@slu.se Department of Soil and Environment, Swedish University of Agricultural Sciences, Box 7014, Uppsala 750 07, Sweden
Supplemental data for this article can be accessed online at https://doi.org/10.1080/09064710.2025.2597105.

by management of the soil surface. Moreover, by monitoring all-year-round it is possible to follow how agricultural management practices and weather influence leaching losses in the short-term and to assess seasonal patterns. One of the core aspects of observation fields is that they represent normal Swedish agricultural management.

Although the aim of the observation fields was to monitor common agricultural practices and their impact on the environment, the information and data have been utilised for numerous research projects and studies. For example, one of the fields had a damaged drainage system during the first 23 years of monitoring. By studying the collected data from the drainage water and groundwater, Wesström et al. (2015) concluded that after drainage repair, the nutrient loads transported by the drainage system were higher than before repair. The same field also experienced a high application of poultry manure which was assessed and discussed by Ulén et al. (2014). The N and P leaching increased rapidly following the application of manure, and the concentrations in drainage water were elevated for several months after.

These observation fields are well-known and have been followed for a long time, therefore, data about, for example, soil properties, is available to be hand-picked for studies requiring properties that the observation fields can provide. This was the case in Villa et al. (2012), wherein several observation fields were selected for soil sampling intended for laboratory studies of simple methods assessing soil erodibility. Soils from all observation fields were, together with many other soils, used in a risk assessment for losses of dissolved reactive P in Ulén (2006). Sandström et al. (2023) used two of the fields in a modelling study of the transport of suspended sediments. Soil from the observation fields has been used for testing a P loss risk assessment tool; Swedish phosphorus index (Djodjic and Bergström 2005a). Furthermore, Djodjic and Bergström (2005b) used seven of the observation fields to explore how site-specific factors affect the leaching losses of dissolved and particulate bound P. They observed that soil P content and P sorption capacity influenced leaching of dissolved P whilst factors such as soil texture and slope of the field were more important for the leaching of particulate bound Ulén et al. (2019) used a field in south-west Sweden to explore and demonstrate how the North Atlantic Oscillation influences winter conditions and nutrient leaching in that region. Total N and total P leaching were correlated to the North Atlantic Oscillation index but short-time events on the field e.g. management practices such as fertilisation, had a greater impact on changes in nutrient leaching on a short-term basis. Additionally, one of the most important tasks for the observation fields is their inclusion in simulation models and calculations of national leaching losses of N och P in Sweden (Johnsson et al. 2023).

Long-term environmental monitoring in the agricultural landscape is crucial to assess and better understand how agricultural practices, together with soil type and climate, affect water quality and nutrient leaching. The long-term series of water quality data is important when evaluating single values and placing them within a larger context enhances understanding if the value is in the range of normality. Today, long-term environmental data contributes to knowledge and predictions of future scenarios in a changing climate through the input of data in models.

The observations fields are a national monitoring programme, financed by the Swedish Environmental Protection Agency. This paper marks the 50-year celebration of the agricultural observation fields in Sweden. The main aims were to: (1) describe the monitoring methods used for the network of observation fields, (2) present and evaluate water quality data in subsurface drainage water and groundwater in relation to annual and seasonal variations, climate, soil type, and cropping system.

Materials and methods

The observation fields were distributed across Sweden to cover the main variations in climate, soil types, and agricultural management practices (Figure 1). The exact locations of the fields were not displayed so that the farmers and landowners were kept anonymous and thus protected from outside pressure regarding how the field was managed. Each year, the farmers managing the fields were asked to report what had been performed on the field throughout the year e.g. soil management, fertilisation, application of manure, and crops grown. All fields followed the usual cultivation methods in Sweden as well as more specific ones for the particular region. The size of the fields ranged from four to 34 hectares. There was a total of 19 observation fields between 1973 and 1989. Three of the early fields were closed after just a few years of monitoring and thus, their data is not presented here. Three additional fields (8C, 17S, and 18T) were closed after 22, 23, and 15 years respectively. Hence, 13 fields are still running in 2025, two of them since 1973, eight from the

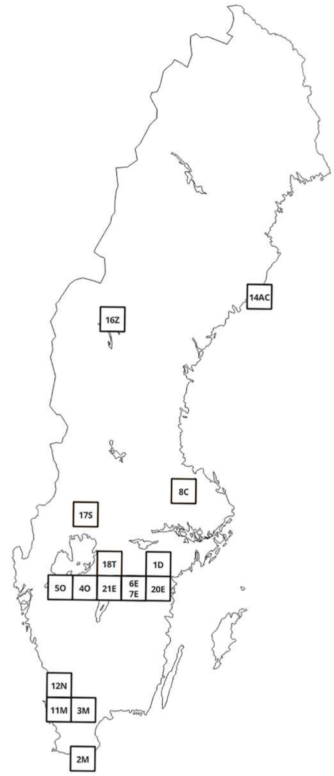

Figure 1. Map with the approximate location of the agricultural observation fields marked.

1970s, and three from the 1980s. Selected information of the fields is presented in Table 1 and soil data in Table 2.

Measuring stations and water sampling

When the observation fields were established, a defined drainage area was identified from existing maps of the drainage system. The field drains were usually placed at a depth of around 1 m in a fishbone pattern connected to a main pipe. To collect water from the field an underground measuring station was built which the main pipe was connected (Figure 2). The underground measuring station was built with concrete and contained a water basin that slowed down and balanced the drainage water. The outflow of the basin was a triangular v-notch weir, with an opening angle of 90 degrees, which was used to determine the water flow (Figure 3). The size of the basin was adapted to the size of the respective field to ensure good measuring conditions.

The water level in the basin was measured by a displacement body connected to a load cell that converted the tension force into a signal. From the beginning, the water level was logged by a mechanical writing stage recorder. The paper in the stage recorder was replaced monthly and digitised on an hourly basis. Between 1998 and 2004, the stage recorders were replaced or complemented by a battery and float operated shaft encoder, OTT Thalimedes, with integrated data logger that digitally recorded and stored water level data hourly. The data was annually transferred to a personal computer. During periods of water flow, manual water sampling at the v-notch was performed every other week, except during the 1970s when the water was sampled every fourth week.

The exception from this procedure was field 18 T, which was located on an area of drained peatland. There, the water samples were taken directly in the main drainage pipe entering a measuring well. The drainage water was then transported by an electrical pump to an outflow well and released to the stream. The water flow was calculated from the capacity and operation time of the pump. The pump normally operated during wet periods i.e. spring.

Table 1. Field code, years of monitoring, area of the field, soil type, standard normal precipitation and air temperature (SMHI), type of farm or cropping system, starting year of using Thalimedes logger and flow-proportional water sampling for the 16 observation fields, and number of groundwater pipes.

				Standard normal Precipitation		Standard normal air temperature			Start of Thalimedes logger/flow-prop.	Groundwater pipes
Field code	Year	Areal ha	Soil type	1961–1990	1991–2020	1961–1990	1991–2020	Type of farm/cropping system	Year	No.
14AC	1988-	8	Silt loam	588	635	2.7	4.1	Ley, cereals, dairy cows	2010/2010	-
16Z	1977-	7	Loam	544	548	2.6	3.7	Ley, cereals, dairy cows, biogas production	2010/2010	1
8C	1975–1997	14	Clay loam	550	586	5.4	6.5	Cereals	–	2
17S	1977–2000	11	Sandy loam	691	718	5.7	6.6	Ley, cereals	–	-
18T	1982–1997	11	Peat soil	624	677	5.8	7.1	Ley, cereals, dairy cows	–	2
1D	1974-	7	Clay loam	519	612	5.6	6.8	Ley, cereals, dairy cows, organic farming since 1989	2013/2009	5
7E	1976-	27	Silty clay	508	610	6.2	7.6	Cereals, cattle, organic farming since 2013	2004/2009	2
20E	1989-	5	Clay	592	567	6.2	7.6	Cereals, pigs	2001/2008	-
6E	1974-	11	Loam	516	565	6.1	7.1	Cereals, potatoes	2001/2011	4
21E	1989-	4	Sandy loam	481	540	6.2	7.6	Cereals	2001/2012	-
5O	1977-	11	Loam	558	611	6.1	7.2	Cereals	2000/2013	2
4O	1975-	19	Silty clay loam	556	622	6.2	7.3	Ley, cereals, dairy cows	2001/2009	4
12N	1976-	15	Sandy loam	773	709	7.2	8.4	Cereals, potatoes, pigs (1976-2002), dairy cows (2002-), biogas production	1999/2013	3
11M	1976-	22	Silty clay loam	740	785	7.5	8.5	Ley, cereals, dairy cows	1998/2009	2
3M	1973-	9	Sand	558	611	7.5	8.4	Maize, sugar beets, potatoes, cereals, dairy cows	1999/–	2
2M	1973-	34	Loam	663	698	7.3	8.3	Cereals, sugar beets	1998/2009	2

Table 2. Selected soil properties (pH (H_2O), total C (% of dry weight), total N (% of dry weight), P-Al (mg 100 g^{-1}), K-Al (mg 100 g^{-1})) and soil texture (clay % (particle size <0.002 mm), silt % (particle size 0.002-0.06 mm) and sand % (particle size 0.06–2 mm)) of the fields, sampled and analysed in 2005, for the soil profile depth's of 0–20, 20–60 and 60–90 cm.

	pH			Total C (% of DW)			Total N (% of DW)			P-Al (mg 100g^{-1})			K-Al (mg 100g^{-1})			Clay %			Silt %			Sand %		
Field	0–20	20–60	60–90	0–20	20–60	60–90	0–20	20–60	60–90	0–20	20–60	60–90	0–20	20–60	60–90	0–20	20–60	60–90	0–20	20–60	60–90	0–20	20–60	60–90
14AC	6.1	5.8	5.1	2.2	1.2	0.6	0.2	0.1	0.1	10.3	4.3	4.0	8.7	6.6	7.4	5	7	12	61	69	77	34	24	11
16Z	6.7	6.9	7.3	7.6	3.8	1.9	0.7	0.4	0.2	5.5	3.5	2.9	8.3	5.9	5.5	25	23	21	48	47	48	27	30	31
1D	5.7	6.3	6.7	1.9	0.7	0.4	0.2	0.1	0.1	5.2	5.2	7.4	14.2	15.4	19.7	35	47	58	43	40	36	23	12	6
7E	6.5	6.7	6.9	2.4	1.1	0.4	0.3	0.2	0.1	5.7	3.9	8.5	16.8	16.1	18.1	43	51	55	41	37	37	16	12	9
20E	6.9	7.2	7.4	2.8	1.2	0.9	0.3	0.1	0.1	6.9	5.7	9.9	20.8	19.4	22.0	59	66	62	36	31	35	5	3	3
6E	6.4	6.6	7.0	2.5	0.5	0.2	0.2	0.1	0.1	9.4	6.9	11.6	12.2	8.1	8.8	13	18	27	47	55	63	41	27	10
21E	7.0	7.1	7.3	1.8	1.5	1.0	0.2	0.2	0.1	9.2	7.0	5.5	8.9	7.4	5.5	17	17	15	27	29	31	56	55	54
5O	6.3	6.4	6.7	2.9	1.3	0.3	0.3	0.2	0.1	4.8	3.2	9.2	6.8	7.1	10.2	15	22	36	39	46	48	46	31	15
4O	6.6	6.7	6.8	1.9	0.6	0.3	0.2	0.1	0.1	3.2	3.7	7.7	9.2	10.0	11.0	28	36	41	60	56	54	12	8	5
12N	6.3	6.3	6.3	3.8	1.2	0.3	0.3	0.1	0.1	7.2	1.4	0.5	4.6	2.4	1.9	11	5	2	19	12	7	71	83	91
3M	6.8	7.0	7.6	1.7	0.6	0.4	0.3	0.1	0.1	34.7	22.4	14.1	8.6	4.2	3.2	5	3	2	7	3	2	88	94	96
2M	6.9	7.1	7.4	1.5	1.1	1.6	0.2	0.2	0.2	6.9	4.6	5.9	8.4	7.0	7.2	19	20	21	30	30	29	51	50	50

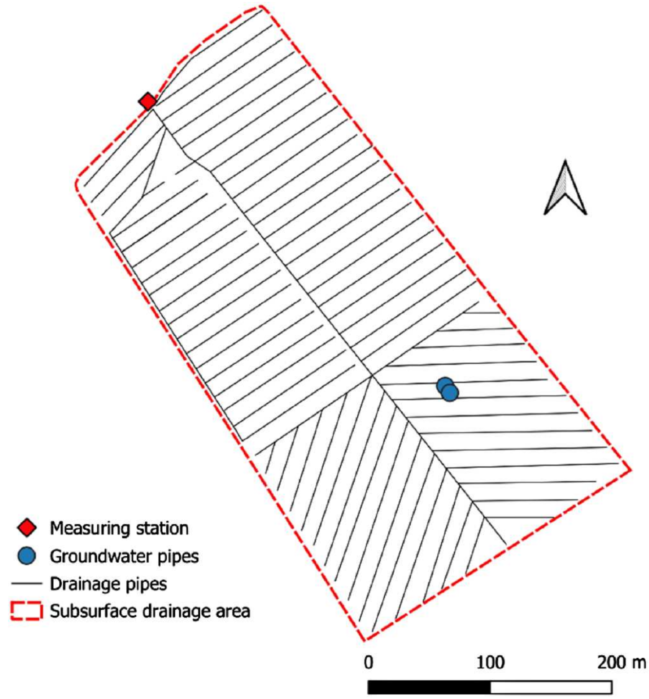

Figure 2. Map of the drainage system of field 5O.

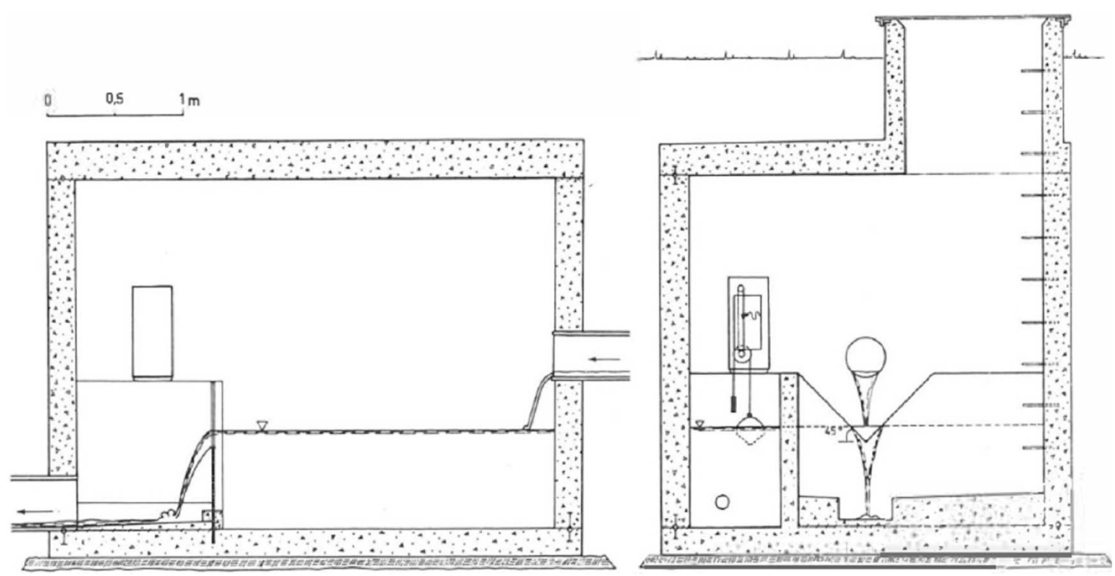

Figure 3. Schematic picture of the concrete underground measuring station.

Between 2008 and 2013, flow-proportional water sampling equipment was installed in all stations that were running at that time. This system replaced the manual water sampling performed fortnightly in the v-notch. Field 3M was excluded from the programme for a couple of years during that time and therefore did not receive the new equipment. When the field was reintroduced to the programme, the water sampling continued as manual sampling every other week. To enhance comparisons between the results from the two water sampling methods, they were run in parallel for 1–3 years.

A Campbell logger (Campbell Scientific Ltd.) recorded the water level and calculated current flow (L sek^{-1}) twice a minute. When a pre-set water volume (approximately 0.1 mm of discharge) had passed, a peristaltic pump took a subsample of approximate 20 ml through a plastic tube that extracted water from the basin. A capacitive sensor detected water before the pump, which enabled the sampling time and thereby the volume of each sample to be more precise. This system also enabled visual notification of the water sampling functionality. Before the sample was taken, the pump reversed to empty water from the tube. The subsamples were collected in a 10 L glass bottle which was emptied bi-weekly. When the composite sample was emptied, the exact time was recorded to enable an accurate calculation of nutrient load. The water samples were instantly sent by mail to a laboratory for analysis.

This type of sampling method resulted in different amounts of water in the composite sample depending on the amount of discharge. When the water flow was very low, time-controlled sampling (2 samples per day) was applied to obtain enough water for analysis. In order to protect the equipment from breakdown during periods with frozen water, the water sampling was programmed to stop until water temperature increased above freezing.

To provide the new equipment with electricity and achieve a dry environment, a small house was built close to the underground measuring station. The house was equipped with a solar panel and a backup battery. Also, precipitation (mm) and temperature (°C) were recorded hourly at the stations with rain gauges and type T Thermocouples (copper-constantan). Temperature was measured in air outside the small house and near the water sampler. A modem enabled wireless connection and transfer of logger data from the field to a computer at the university.

Ground water

Twelve of the fields had groundwater pipes installed in the 1970s, ranging from one to five at each field. The probed depth ranged from 1.0 m to 5.8 m. At observation depth, the piezometers had vents surrounded by filters of sand and above the sand, the pipes were secured by bentonite up to surface level.

The soil water pressure head was measured monthly using a dipping tape measure with a metal weight. When the metal weight touched the water surface in the pipe, an audible sound was heard, and the depth could be noted on the tape measure. Water from the pipes was sampled with a peristaltic pump attached to a silicone tube. The pipes were drained, and fresh water was allowed to enter the pipes for a couple of days prior to sampling. Ground water was sampled monthly between 1973 and 1982, four times per year in 1983–1986, and six times per year since 1987 i.e. in January, March, May, July, September, and November.

Water analyses

Subsurface drainage water was analysed for total nitrogen (tot-N), nitrate and nitrite nitrogen ($NO_3 + NO_2$-N), total phosphorus (tot-P), phosphate phosphorus (PO_4-P), suspended material (susp.mtrl), and total organic carbon (TOC). Total nitrogen was determined on unfiltered samples where all N was converted to nitrous oxide through a combustion catalytic oxidation method before analysis. Nitrate and NO_2-N was determined spectrophotometrically. Total phosphorus was determined on unfiltered samples and PO_4-P on filtered samples (0.2 µm) with a method where all P is oxidised with $K_2S_2O_8$ to PO_4-P and then analysed photometrically. Total P was analysed both before and after filtration (0.2 µm) and particulate phosphorus (part-P) was calculated as the difference between tot-P in filtered and unfiltered samples. Suspended material was determined by the amount of particles trapped in a fibreglass membrane after filtration of the water. Total organic carbon was determined by first removing inorganic carbon through acidification with HCl and the produced carbonic acid was removed with CO_2-free gas. Then the carbon was combusted, and the CO_2 was quantified.

Parameters that can be altered by storage e.g. pH, alkalinity, and conductivity, were analysed in water sampled taken manually at the v-notch at the same time as the composite water sample was emptied. pH was determined in water at 25°C with a combination electrode, alkalinity was measured with a combination electrode after titration with HCl to a set pH value (5.6) and conductivity was determined with a conductivity metre at 25°C. For field 3M, which lacks flow-proportional water sampling, all samples were taken in the v-notch.

Throughout 1980–2010, selected ions, [potassium (K^+), sodium (Na^+), magnesium (Mg^{2+}), calcium (Ca^{2+}), chlorine (Cl^-), and sulphate-sulphur (SO_4-S)] were also analysed in the subsurface drainage water. Potassium, Na^+, Mg^{2+} and Ca^{2+} was determined by inductively coupled plasma mass spectroscopy. Chloride and SO_4-S was determined by ion chromatography. Ammonium nitrogen (NH_4-N) was analysed during the period of manual sampling and determined spectrophotometrically. Ground water was analysed for nitrate and nitrite nitrogen ($NO_3 + NO_2$-N), pH, alkalinity, and conductivity. Up until 2010, selected ions (K^+, Na^+, Mg^{2+}, Ca^{2+}, Cl^-, and SO_4-S) were also analysed in the ground water.

Water samples, subsurface- and ground water, were analysed according to the Swedish standard methods (Supplemental Table 1). These methods have changed slightly over the years, but the changes are of minor importance and fall within general uncertainty.

Calculations

Daily subsurface leaching loads of nutrients were calculated by multiplying daily water discharge and estimated daily concentrations. For manual water sampling, the daily concentrations were calculated by linear interpolation between the analysed values. For flow-proportional water sampling, the analysed values represented all days between the samples. During the summer months, runoff typically ceased and consequently no water was sampled. The last analysed sample in spring was interpolated and used in the calculation until discharge had completely ended. Similarly, in autumn, the first analysed sample after the dry period represented the time up until sampling.

Daily loads were then summed up to monthly loads and monthly loads were summed up to annual load, here presented as the agrohydrological year (1 July – 30 June). Mean monthly and mean annual concentrations were calculated by dividing total monthly and total annual load by total monthly and total annual water discharge, respectively. Alkalinity, conductivity, and pH, which were always sampled by manual samples, were reported as arithmetic mean of the analysed values for agrohydrological years. Concentrations in groundwater were also reported as arithmetic mean of the analysed values but for calendar years (1 January – 31 December).

Agrohydrological year

For the observation fields, 'year' was presented as the agrohydrological year i.e. 1st of July to 30th of June. The two main arguments for using agrohydrological years concern the two included parts, agronomy and hydrology. The hydrological part considers that the break between years should occur during a period of low or absent subsurface discharge. If the break takes place between December and January, a continuous and dynamic discharge period is divided into two different years. The Swedish Meteorological and Hydrological Institute (SMHI) defines a hydrological year as being from October to September i.e. the breaking point is when the hydrological processes are at their lowest period. The agronomy part considers that the agricultural year commonly ends with harvest or some other interval of crop growth in July-August. Subsequently, the new cropping year commences with autumn soil management or sowing of a winter crop, just in time for the beginning of a new discharge period. Other divisions of agrohydrological years have been used in other studies e.g. 1st September to 30th August (Uusitalo et al. 2018; Norberg and Aronsson 2022), 1st May to 30th April (Bechmann 2014), and 1st March to 28 February (Liu et al. 2023), depending on what part of the agricultural year is most important for the assessment.

Results and discussion

The observation fields consist of a unique series of measurements with more than 20 analysed variables, at least 14,000 single water sampling occasions, throughout approximately 700-site years, calculated up to June 2023. The fields constitute an infrastructure, with a substantial collection of water quality data along with information about field management and soil properties. This data collection can be used for various tasks such as research projects, modelling, and monitoring of the development of Swedish agriculture. Water quality data from the observation fields are open access and available for anyone to use at https://miljodata.slu.se.

The observation fields represent, together with the monitoring of small streams in agricultural catchments (Kyllmar et al. 2014; Kyllmar et al. 2023) and the long-term field experiments on water quality (Norberg and Aronsson 2024; Norberg and Aronsson 2025), a comprehensive monitoring of nutrient losses from Sweden's agricultural landscape. The three scales, streams, fields, and experimental plots, enhance knowledge

Table 3. Data from manual subsurface drainage water sampling, starting from the years indicated in the table and ending when manual sampling finished for the respective field (see Table 1) or as indicated in the table. Annual mean discharge and precipitation (mm), and annual mean concentrations (mg L^{-1}) and loads (kg ha^{-1}) of total nitrogen (tot-N), nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N), total phosphorus (tot-P), phosphate-phosphorus (PO_4-P), suspended material (susp.mtrl) and total organic carbon (TOC). Mean and standard error in brackets for agrohydrological years (1 July – 30 June).

	Discharge	Precipitation	tot-N		NO_3-N		NH_4-N		tot-P		PO_4-P		Susp.mtrl		TOC	
	1973-	1973–	1973–		1973–		1973–2010		1973-		1973–2000		1985–		1994–	
Field	mm	mm	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}
14AC	113 (14.3)	646 (27.9)	4.8 (0.53)	4.6 (0.00)	4.1 (0.45)	4.0 (0.00)	0.08 (0.011)	0.07 (0.000)	0.04 (0.004)	0.04 (0.000)	0.01 (0.002)	0.01 (0.000)	23.2 (3.77)	19.5 (0.01)	5.7 (0.78)	2.7 (0.00)
16Z	288 (26.2)	520 (16.0)	17.0 (2.17)	6.1 (0.01)	15.4 (2.06)	5.5 (0.01)	0.04 (0.006)	0.02 (0.000)	0.11 (0.028)	0.04 (0.000)	0.06 (0.033)	0.02 (0.000)	11.8 (3.24)	2.8 (0.01)	15.1 (2.11)	2.6 (0.00)
8C	49 (8.0)	585 (22.8)	4.2 (0.74)	9.2 (0.01)	3.9 (0.69)	8.4 (0.01)	0.02 (0.006)	0.03 (0.000)	0.07 (0.015)	0.16 (0.000)	0.04 (0.009)	0.09 (0.000)	12.8 (4.35)	21.7 (0.07)	1.4 (0.55)	1.2 (0.00)
17S	110 (9.4)	706 (23.8)	5.3 (0.77)	4.8 (0.01)	4.3 (0.70)	3.8 (0.01)	0.10 (0.032)	0.13 (0.001)	0.16 (0.021)	0.17 (0.000)	0.09 (0.011)	0.10 (0.000)	5.6 (1.37)	3.4 (0.01)	6.6 (1.21)	2.0 (0.00)
18T	334 (48.8)	650 (23.0)	21.1 (3.93)	6.0 (0.00)	6.8 (1.94)	1.8 (0.00)	5.37 (1.131)	1.74 (0.003)	0.51 (0.081)	0.15 (0.000)	0.33 (0.063)	0.09 (0.000)	38.9 (8.58)	11.1 (0.03)	323.1 (73.46)	12.2 (0.00)
1D	220 (16.0)	574 (14.7)	13.7 (1.93)	6.7 (0.01)	10.6 (1.78)	5.3 (0.01)	0.13 (0.027)	0.06 (0.000)	0.95 (0.105)	0.42 (0.000)	0.46 (0.058)	0.14 (0.000)	510.6 (88.23)	158.4 (0.26)	30.3 (4.72)	6.4 (0.00)
7E	301 (23.3)	538 (14.4)	12.4 (1.40)	3.9 (0.00)	10.4 (1.18)	3.3 (0.00)	0.05 (0.009)	0.02 (0.000)	0.35 (0.050)	0.11 (0.000)	0.17 (0.023)	0.04 (0.000)	230.4 (58.72)	47.7 (0.10)	19.5 (2.79)	2.9 (0.00)
20E	119 (14.1)	575 (21.3)	7.7 (1.05)	6.8 (0.01)	6.4 (0.90)	5.8 (0.01)	0.03 (0.008)	0.03 (0.000)	0.26 (0.045)	0.21 (0.000)	0.16 (0.045)	0.08 (0.000)	161.8 (33.97)	135.1 (0.23)	12.9 (1.73)	7.8 (0.00)
6E	98 (12.6)	558 (17.0)	11.4 (1.63)	11.6 (0.01)	10.2 (1.50)	9.5 (0.01)	0.06 (0.036)	0.10 (0.001)	0.07 (0.021)	0.11 (0.000)	0.05 (0.024)	0.07 (0.000)	17.2 (6.80)	11.5 (0.03)	6.5 (1.21)	2.7 (0.00)
21E	131 (16.4)	527 (21.7)	17.4 (2.13)	15.3 (0.02)	15.6 (1.86)	14.0 (0.02)	0.04 (0.025)	0.03 (0.000)	0.03 (0.006)	0.02 (0.000)	0.01 (0.002)	0.00 (0.000)	5.8 (1.07)	5.6 (0.01)	6.3 (1.01)	3.4 (0.00)
5O	174 (16.1)	595 (12.3)	16.8 (2.83)	8.2 (0.01)	15.2 (2.62)	7.3 (0.01)	0.04 (0.009)	0.03 (0.000)	0.12 (0.027)	0.06 (0.000)	0.07 (0.037)	0.02 (0.000)	41.1 (8.61)	17.0 (0.04)	14.0 (1.36)	3.2 (0.00)
4O	195 (11.4)	603 (18.0)	17.8 (1.40)	10.1 (0.01)	15.5 (1.28)	8.8 (0.01)	0.05 (0.007)	0.03 (0.000)	0.23 (0.019)	0.12 (0.000)	0.12 (0.012)	0.05 (0.000)	86.3 (12.79)	28.5 (0.04)	22.6 (2.01)	4.8 (0.00)
12N	420 (24.4)	803 (18.2)	44.2 (4.06)	10.7 (0.01)	39.8 (3.61)	9.7 (0.01)	0.08 (0.014)	0.02 (0.000)	0.09 (0.008)	0.02 (0.000)	0.03 (0.003)	0.00 (0.000)	27.1 (5.78)	4.8 (0.01)	44.2 (3.92)	5.4 (0.00)
11M	232 (16.3)	788 (23.0)	16.7 (1.70)	7.7 (0.01)	12.7 (1.55)	5.9 (0.01)	0.38 (0.126)	0.18 (0.001)	0.67 (0.071)	0.28 (0.000)	0.19 (0.022)	0.06 (0.000)	652.8 (108.59)	202.9 (0.32)	35.6 (5.62)	7.7 (0.00)
3M	304 (13.1)	694 (19.4)	79.2 (4.626)	26.3 (0.01)	73.1 (4.42)	24.3 (0.01)	0.07 (0.013)	0.02 (0.000)	1.69 (0.136)	0.55 (0.000)	1.21 (0.116)	0.23 (0.001)	11.0 (1.23)	2.6 (0.00)	48.0 (3.07)	8.5 (0.00)
2M	241 (18.2)	708 (21.5)	30.1 (2.64)	13.2 (0.01)	26.8 (2.35)	11.8 (0.01)	0.07 (0.016)	0.03 (0.000)	0.15 (0.017)	0.06 (0.000)	0.08 (0.011)	0.02 (0.000)	38.6 (7.27)	11.4 (0.03)	23.9 (2.40)	5.3 (0.00)

Table 4. Data from flow-proportional subsurface drainage water sampling, starting when flow-proportional sampling started for respective field (see Table 1) until agrohydrological year 2022/2023. Annual mean discharge and precipitation (mm), and annual mean concentrations (mg L^{-1}) and loads (kg ha^{-1}) of total nitrogen (tot-N), nitrate-nitrogen (NO_3-N), total phosphorus (tot-P), phosphate-phosphorus (PO_4-P), particulate phosphorus (part-P), suspended material (susp.mtrl) and total organic carbon (TOC). Mean and standard error in brackets for agrohydrological years (1 July – 30 June).

Field	Discharge mm	Precipitation mm	tot-N		NO_3-N		tot-*P*		PO_4-P		part-P		Susp.mtrl		TOC	
			kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}	kg ha^{-1}	mg L^{-1}
14AC	90 (12.1)	636 (35.8)	2.8 (0.42)	3.1 (0.28)	2.4 (0.36)	2.7 (0.26)	0.04 (0.007)	0.04 (0.004)	0.00 (0.002)	0.00 (0.001)	0.04 (0.007)	0.04 (0.004)	20 (3.1)	21 (1.2)	4.7 (0.65)	5.4 (0.35)
16Z	252 (25.9)	519 (16.8)	23.8 (4.85)	8.9 (1.61)	22.0 (4.49)	8.2 (1.54)	0.07 (0.018)	0.03 (0.005)	0.04 (0.012)	0.02 (0.004)	0.02 (0.006)	0.01 (0.002)	15 (3.4)	6 (1.4)	11.2 (1.73)	4.4 (0.31)
1D	180 (15.5)	592 (25.3)	15.6 (4.24)	8.5 (2.07)	12.6 (3.93)	6.8 (1.96)	1.08 (0.136)	0.59 (0.048)	0.32 (0.059)	0.18 (0.017)	0.68 (0.109)	0.38 (0.052)	446 (78.1)	249 (36.7)	26.5 (2.83)	15.2 (1.28)
7E	265 (33.6)	577 (21.4)	15.6 (2.57)	6.8 (0.87)	13.2 (2.35)	5.8 (0.83)	0.82 (0.112)	0.31 (0.019)	0.32 (0.048)	0.12 (0.014)	0.46 (0.066)	0.17 (0.011)	475 (79.1)	175 (14.5)	21.2 (2.38)	8.6 (0.63)
20E	99 (14.3)	545 (19.7)	12.0 (2.43)	13.2 (1.91)	10.7 (2.15)	12.0 (1.75)	0.19 (0.037)	0.18 (0.015)	0.09 (0.021)	0.08 (0.011)	0.10 (0.019)	0.09 (0.009)	145 (31.2)	130 (17.9)	8.9 (1.35)	9.1 (0.51)
6E	101 (16.6)	585 (28.4)	10.9 (2.03)	10.6 (0.81)	10.2 (1.89)	10.0 (0.79)	0.03 (0.006)	0.03 (0.004)	0.02 (0.004)	0.01 (0.001)	0.01 (0.003)	0.01 (0.002)	9 (2.1)	9 (1.6)	4.9 (0.89)	4.7 (0.24)
21E	110 (16.3)	551 (20.8)	20.8 (4.08)	18.2 (2.51)	19.6 (3.85)	17.2 (2.39)	0.01 (0.003)	0.01 (0.003)	0.01 (0.002)	0.01 (0.002)	0.00 (0.003)	0.00 (0.002)	5 (1.1)	5 (0.7)	3.1 (0.45)	2.9 (0.16)
5O	195 (26.6)	576 (23.1)	24.4 (5.67)	13.5 (2.46)	22.7 (5.61)	12.6 (2.47)	0.22 (0.054)	0.10 (0.015)	0.05 (0.009)	0.03 (0.007)	0.16 (0.015)	0.07 (0.014)	108 (31.6)	48 (10.0)	17.5 (3.85)	8.3 (0.89)
4O	193 (14.4)	603 (19.9)	18.9 (2.63)	11.4 (2.22)	17.2 (2.55)	10.5 (2.18)	0.37 (0.047)	0.18 (0.016)	0.07 (0.014)	0.03 (0.005)	0.27 (0.016)	0.13 (0.014)	204 (25.9)	102 (9.6)	19.6 (2.17)	10.2 (0.90)
12N	363 (35.2)	690 (38.4)	58.0 (8.11)	17.0 (2.40)	54.6 (7.82)	16.1 (2.33)	0.05 (0.010)	0.02 (0.002)	0.02 (0.003)	0.00 (0.001)	0.03 (0.006)	0.01 (0.001)	18 (4.0)	5 (0.7)	38.8 (4.73)	10.5 (0.54)
11M	196 (15.7)	749 (27.0)	19.6 (3.45)	11.2 (2.18)	14.6 (3.06)	8.8 (2.06)	1.48 (0.313)	0.69 (0.094)	0.17 (0.044)	0.08 (0.013)	1.25 (0.268)	0.58 (0.084)	1137 (216.0)	527 (66.8)	44.9 (8.59)	21.7 (2.71)
2M	224 (21.4)	648 (34.2)	24.8 (4.04)	11.3 (1.61)	22.8 (3.86)	10.3 (1.53)	0.27 (0.048)	0.12 (0.025)	0.11 (0.026)	0.05 (0.015)	0.15 (0.030)	0.07 (0.012)	94 (18.8)	41 (7.6)	23.6 (2.17)	10.8 (0.49)

regarding how agricultural management affects water quality and how these three scales are linked. An advantage about measuring water quality from fields is that it is not affected by other land use types such as forests or urban areas (i.e. catchment monitoring) but is large enough to catch events of management or weather. Ulén et al. (2015) assessed observation field 2M and catchment M42, which field 2M is a part of, and observed a lagged response of nutrient leaching for the catchment compared to the field. This indicates that mitigation measures performed on agricultural fields will be detected in streams at a later stage.

Annual water discharge and leaching of N and P

The water passing the v-notch in the measuring stations, originates from precipitation and in the fields 6E, 12N, and 3M, several years also from irrigation. The proportion of precipitation that ended up in subsurface drainage was on average 33% for all fields, ranging from 8% (field 8C) to 56% (field 7E) (Table 3). The low discharge through the measuring station was the primary reason for closing field 8C after 22 years of monitoring. Field 6E also had a low annual average discharge (approximately 100 mm, Tables 3 and 4) and the drainage flow was usually zero during the summer months and in, very dry years, all-year-round. A large proportion of the precipitated water likely bypassed the drainage system and entered the nearby river directly. The discharge could also be higher than expected due to the addition of upcoming groundwater. This was difficult to estimate but was likely the case for field 3M, which also had discharge during the usually dry summer (Figure 4).

Once the flow-proportional water sampling system was installed, the manual water sampling was kept for one to three years at the fields and the two sampling systems ran in parallel. When all available pairs of data were used ($n = 20$), tot-N had on average slightly higher concentrations with flow-proportional sampling than with manual sampling (8.8 and 8.1 mg L^{-1} respectively, Student's *t*-test, $p < 0.05$), but not all compared pairs followed this trend. For tot-P, all compared pairs had higher concentrations with flow-proportional sampling than with manual sampling (average 0.20 and 0.14 mg L^{-1} respectively, Student's *t*-test, $p < 0.001$), which is in line with Ulén and Persson (1999). Apparently, P has a higher temporal variation than N. With manual water sampling, the linear interpolation of fortnightly or monthly concentration values may under- or overestimate nutrient loads during the period between samples. The concentrations in the

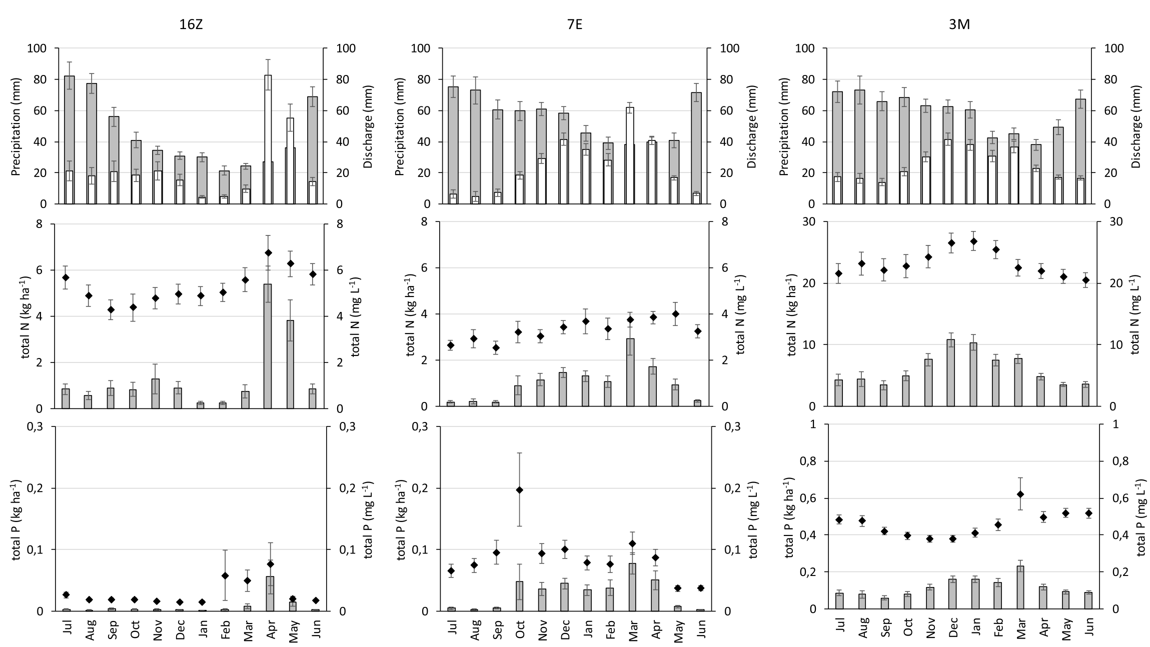

Figure 4. Average monthly precipitation (grey bars, mm), discharge (white bars, mm), concentrations (points, mg L^{-1}) and loads (bars, kg ha^{-1}) of total nitrogen (N) and total phosphorus (P) from field 16Z, 7E and 3M during 1977–2010 (16Z), 1976–2010 (7E) and 1973–2010 (3M). Mean and standard error for the respective month. Note the different scales for field 3M.

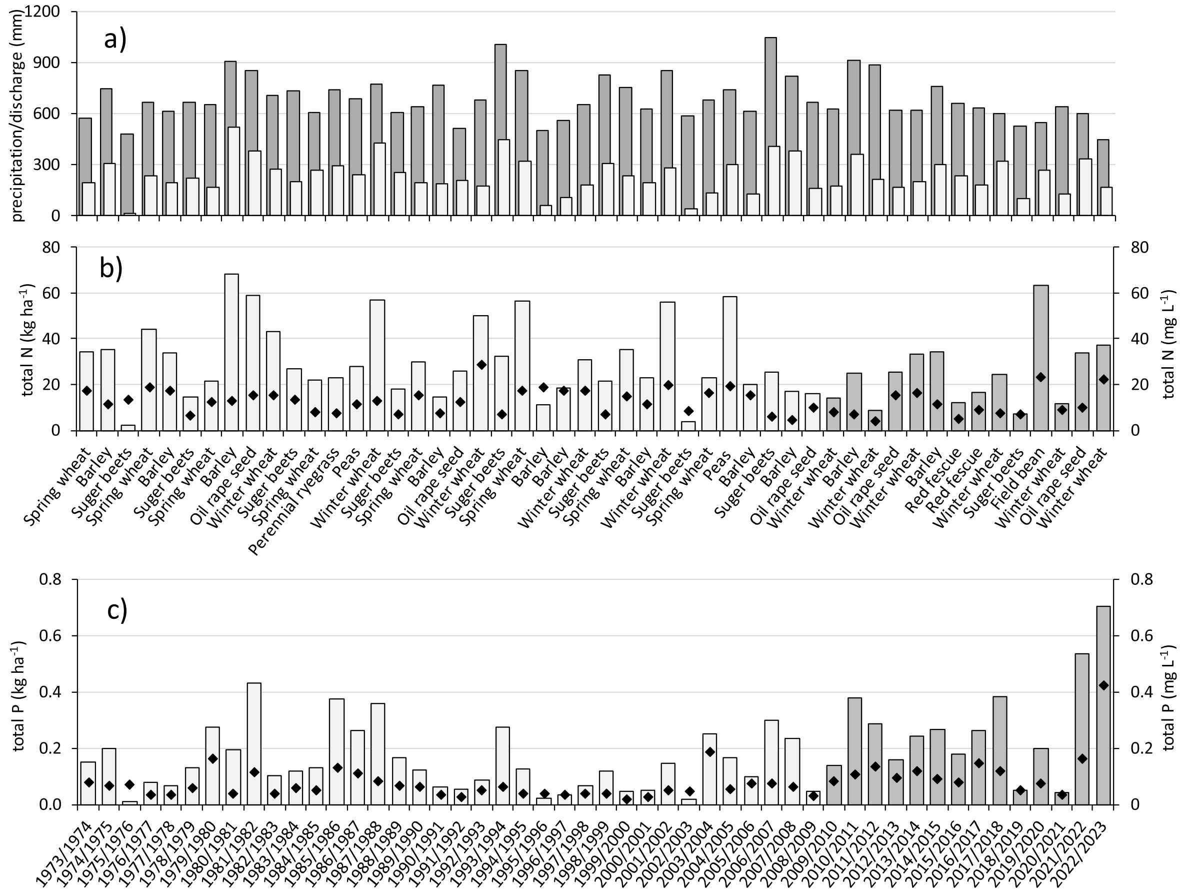

Figure 5. Mean annual (a) precipitation (dark colour bars, mm) and discharge (light colour bars, mm), loads (bars, kg ha^{-1}) and concentrations (squares, mg L^{-1}) of (b) total nitrogen (N) and (c) total phosphorus (P) for field 2M during 1973-2023. The crops grown on the field during the year starting the respective agrohydrological year (1 July – 30 June) is indicated in (b). Lighter coloured series in (b) and (c) indicate manual water sampling whereas darker coloured series represent flow-proportional water sampling.

drainage water may variate due to heavy rainfall events etc. and whether the sample was taken during, before, or after an event can also affect the results. With flow-proportional water sampling, the variation in nutrient loads is captured.

An example of a 50-year time series of water quality data from an agricultural field in the very south of Sweden (Field 2M) is illustrated in Figure 5. The variation across years was large, with the discharge differing between 16 mm in the very dry agrohydrological year of 1975/1976 to 520 mm in 1980/1981 (Figure 5(a)). At this site, discharge had a positive linear relationship with precipitation ($p < 0.001$), which generally means that more rain gave more discharge through the drainage system on an annual basis. The annual mean concentration of tot-N in the drainage water varied from 4.2 mg L^{-1} in 2011/2012 to 28.8 mg L^{-1} in 1992/1993 (Figure 5(b)). This large variation between years shed light on the importance of a long time series wherein single values can be evaluated within the right context. Furthermore, the annual mean concentration of tot-P reached its highest value (0.42 mg L^{-1}) in 2022/2023 due to one unusually high value on one measuring occasion in January 2023 (Figure 5(c)), which highlights how occasional events can considerably impact the annual mean. Aside from this extreme value, the annual mean concentration of tot-P ranged between 0.02 and 0.19 mg L^{-1} (Figure 5(c)). Compared to the other observation fields, field 2M was in the low end of the annual mean concentration of tot-P and in the middle section for tot-N (Tables 3 and 4).

On average for all fields, approximately 88% of the concentration of tot-N was in NO_3-N form (range 77% to 92%, field 11M and field 21E, respectively) and approximate 0.5% in the form of NH_4-N (range 0.1% to 2.7%, field 3M and field 17S, respectively) in manually sampled drainage water from the fields with mineral soils and the proportions were similar across all fields (Table 3). However, the peat soil (field 18 T) had a low share of NO_3-N concentration (30%) of tot-N and a high share of NH_4-N (29%) compared to the other fields (Table 3), thus 41% was of other forms of N e.g. organic forms.

In regard to P, the concentration of particulate P was on average 68% of tot-P with field 14AC having the highest proportion (87%) and field 16Z having the lowest (21%) (Table 4). The proportion of PO_4-P (mg L^{-1}) of tot-P varied between 9% (field 14AC) and 65% (field 16Z) with an average of 26% for all fields (Table 4). Resultingly, this showed that field 14AC and 16Z have an opposite trend of P fractions, likely due to the different soil types.

Observation field 14AC is the only field with measurements of surface runoff, along with subsurface drainage (Norberg et al. 2022). Moreover, at field 5O, surface runoff was measured for a couple of years in the 1980s, but the outcome was not satisfactory. There have been discussions about whether it would be possible, practical, and economical to construct surface runoff collections and measurements at some of the other observation fields but this has not been implemented. Although surface runoff measurements are challenging to implement and perform, they are important for estimations of different pathways nutrient losses. P bound to clay particles are particularly exposed to soil erosion and transported from the field through surface runoff. Norberg et al. (2022) showed that N and P exhibit opposite long-term trends in surface runoff and subsurface drainage water from field 14AC, with higher N losses from subsurface drainage, whilst the losses of P were higher from surface runoff.

During the first 23 years of monitoring field 5O, there was a leakage in the drainage system. This leakage likely underestimated the concentrations and loads of nutrients from the field during the period of manual sampling (Table 3). Wesström et al. (2015) observed that the concentration of nutrients was elevated after the repair of the drainage system on field 5O and consequently the discharge and load of nutrients increased.

Seasonal water discharge and leaching of N and P

The observation fields in the northern parts of Sweden are majorly affected by the long, cold winters with snowmelt events occurring in spring. As can be seen for field 16Z, the subsurface drainage and nutrient losses were low during the frozen parts of winter (January-March) and in spring, when the soil thaws (April-May), a higher runoff could be seen (Figure 4). This could also be seen at field 14AC where the surface runoff notably peaked during snowmelt (Norberg et al. 2022). Further south in Sweden, field 7E had high discharge and nutrient losses during late autumn and winter (October-February) and peaked in loss during snowmelt in March-April (Figure 4). This is due to both less pronounced soil frost and snow throughout winter compared to further north. In field 3M, in the very south part of Sweden, the winters are mild, resulting in short periods of soil frost. In addition, the soils are seldom covered with snow for long periods. Therefore, the discharge and nutrient losses peaked in midwinter (December-January), when there is no vegetation using the soil water and nutrients. However, an indication to elevated leaching of P in field 3M could be seen in March due to some snow and frost melting (Figure 4). Field 3M has a small inflow of ground water to the drainage system and that is why discharge and nutrient losses do not decrease as much compared to the other two fields during the summer months.

The patterns in monthly mean concentrations of N and P among the three fields varied largely (Figure 4). Field 16Z, with its loamy soil, had its highest concentrations of both N and P in spring, with a distinct peak of P in February-April compared to the rest of the year (Figure 4). The silty clay soil on field 7E had lower N concentrations during summer and slightly higher ones during winter. Phosphorus displayed a similar pattern to N, except for a clear peak in October, which could be due to the common autumn tillage that releases particulate P from this silty clay. The sandy soil on field 3M, on the other hand, shows a third type of pattern for N and P concentrations, with the highest concentrations of N during winter and lower concentrations during summer. Phosphorus had a clear peak in March and the lowest concentrations from October to December (Figure 4). Figure 4 demonstrates the differences in nutrient losses between different locations and soil types, and the importance of them.

Winter weather dynamics have a significant impact on nutrient cycling and leaching. A warm and wet autumn and winter enhance mineralisation of N and increases the risk of leaching losses. Freeze–thaw cycles can release dissolved P from damaged plant cells (Liu et al. 2014). The low average annual concentrations of N and P from the northern fields, 14AC and 16Z, compared to the southern fields (Tables 3 and 4) is due to the north having long winters with low mineralisation and cropping systems with often long growing seasons i.e. leys for forage. However, Lackner et al. (2023) used field 14AC in northern Sweden for modelling of freeze–thaw cycles and how these dynamics influence nutrient leaching, but they did not find any clear connection.

During the period 1988–2009, Ulén et al. (2012) followed 13 of the observation fields but no trends in annual concentrations or transports of tot-N and tot-P could be observed in any of the fields. Throughout the 50 years of monitoring, Sweden has experienced a warmer and, in most places, a wetter climate (Table 1), which most likely affects the soil system and leaching losses of nutrients. For all fields, the standard normal air temperature has increased with 0.9–1.4°C and the standard normal precipitation has changed from a decrease of 64 mm (field 12N) to an increase of 102 mm (field 7E), from 1961–1990 to 1991–2020 (Table 1). Ezzati et al. (2023) reported that temperature and precipitation were drivers of diffuse nutrient loads from agriculture dominated catchments but the drivers were not significant in all studied catchments, which indicates that other local factors also have an influence. However, increased nutrient losses can be counteracted by several agricultural management strategies e.g. spring tillage, catch crops (Norberg and Aronsson 2024).

Impact of soil type, climate, and cropping system on N and P leaching

The soil types of the observation fields range from clay (e.g. fields 7E and 20E) to sandy soils (e.g. fields 12N and 3M) and silty soils (e.g. fields 14AC and 4O, Tables 1 and 2). For 15 years, a field with peat soil was also monitored (field 18 T). In general, the fields with a higher clay content had higher tot-P concentrations in the drainage water (Tables 2–4). An exception to this trend was field 3M, with its course sandy soil, which had one of the highest average concentrations of both N and P in the drainage water compared to the other fields (Tables 2 and 3). The long-term trend for N and P concentration from field 3M was increasing, even though the application rate of manure and fertiliser was not increasing. This somewhat unusual behaviour for a sandy soil has raised questions and the soil has been used in several research projects, particularly those examining P (Djodjic and Bergström 2005a; Djodjic and Bergström 2005b; Djodjic et al. 2023). The high P losses from field 3M were likely due to a high application rate of nutrients, both through cow manure and mineral fertiliser, high soil P (Table 2), and low P sorption capacity (Djodjic et al. 2023). On the other hand, field 12N, which is also a sandy soil (Table 2), exhibits low concentrations of P in the drainage water, due to e.g. high P sorption capacity (Djodjic and Bergström 2005b). The sorption capacity can be positively related to high concentrations of iron (Fe) and aluminium (Al) in the soil (Ulén et al. 2008a). Further, field 11M has high concentrations of Fe and Al (Ulén et al. 2008b), but here, P was probably partially transported to the drainage system by surface runoff and by preferential flow in this clayey soil, and thereby avoided absorption in the soil matrix. At the catchment level, easily releasable soil P (P-AL) is the predominant driver in leaching losses of dissolved reactive P to stream water and P-AL can elevate through a surplus soil P balance at field level (Liu et al. 2023). As Ulén et al. (2001) concluded, P losses are highly variable between the observation fields, due to field specific factors concerning both natural factors e.g. soil and hydrology, and artificial factors e.g. agricultural management practices.

Compared to P, N was affected by location in Sweden as opposed to soil type. The fields in south and south-western Sweden (2M, 3M, 11M and 12N) had higher leaching losses of N than fields further north (1D, 16Z, and 14AC) (Figure 1, Tables 3 and 4). This was due to a warmer climate and higher precipitation in southern Sweden, along with less years which are cropped with ley. For example, field 1D demonstrated that growing a ley crop over several years decreased leaching losses of N compared to cultivation of cereals (data not shown). Data from 35 years of manual water sampling at field 1D showed that during years when the ley was terminated, usually by ploughing in autumn, the concentration of tot-N increased approximately five times compared to years with ley covering the soil all-year-round (12.0 and 2.3 mg L^{-1}, respectively). Moreover, a study of field 12N revealed that growing a catch crop in autumn decreased the NO_3-N concentration in drainage water whilst potatoes increased the concentration, whereas P, was unaffected by different crops (Ulén et al. 2008a). In addition, time series of field 2M (Figure 5), revealed no visible impact on tot-P of the crops grown on the field whilst the concentration of tot-N, for example, was notably lower for the years cropped with sugar beet, due to the long growth period in autumn. Furthermore, Norberg et al. (2022) showed with field 14AC that years with ley lowered the concentration of tot-N in subsurface drainage water compared to years with barley whilst the concentration of tot-P in surface runoff was lower during years with barley compared to years with ley. For field 6E, the impact of spring tillage instead of autumn tillage was estimated with a model simulation to reduce N leaching losses by 10% (Ulén and Johansson 2009).

Two of the observation fields were converted to organic farming during the monitoring period: field 1D in 1989 and field 7E in 2013. Upon visual inspection of N and P monitoring data, this change in management could not be detected in nutrient leaching losses. This is perhaps due to small changes in management practices and the fact that leaching of nitrogen has shown to be similar from organic and mineral nitrogen fertilisation sources (Wallman and Delin 2022).

Other water chemical variables

In addition to N and P, pH, alkalinity, conductivity, suspended material, and TOC have been analysed, and over a 30-year period (1980–2010), a set of different ions e.g. K^+, Mg^{2+}, Na^{2+}, Ca^{2+}, SO_4-S, Cl^- was analysed (Table 3–5).

pH in drainage water was in all cases, excluding field 14AC, in the neutral range, with low variation between years (Table 5). Conductivity is a measure of the concentration of ions in the water and field 20E had the highest conductivity (100 mS m^{-1}), likely because of the high concentration of Na^+, but also Mg^{2+} and Cl^- (Table 5). Field 1D had the lowest conductivity (15 mS m^{-1}) and thus, a low concentration of most analysed ions (Table 5). Alkalinity is a measure of the water's capacity of neutralising acids and consequently, the low pH in field 14AC was connected to low alkalinity (0.3 mmol L^{-1}, Table 5). In general, fields with a pH in the lower range (5.3–7.0) had alkalinity in the lower range (0.3–1.7 mmol L^{-1}) whilst fields with pH in the higher range (7.1–7.7) had alkalinity in the higher range (2.3–6.8 mmol L^{-1}, Table 5).

Suspended material had a positive linear relationship with both tot-P and particulate P, when flow-proportional water data was used (Table 4). This is in line with Sandström et al. (2020), who concluded that the concentration of particulate P had a clear relationship with the concentration of suspended material in stream water, which in turn had a strong link to soil clay content in the agricultural dominated catchments used in that study. Similarly, this was indicated for the observation fields as the five fields with the highest clay content (1D, 7E, 20E, 4O and 11M) also had the highest concentrations of suspended material in the drainage water (Tables 2 and 3). Mean annual TOC load (Table 3), field 18 T excluded, ranged between 1.4 kg ha^{-1} (field 8C) and 48 kg ha^{-1} (field 3M), which for most fields was lower than findings shown in

Table 5. Annual mean pH, conductivity (mS m^{-1}), alkalinity (mmol L^{-1}) and concentrations (mg L^{-1}) of potassium (K^+), sodium (Na^{2+}), magnesium (Mg^{2+}), calcium (Ca^{2+}), chloride (Cl^-) and sulphate-sulphur (SO_4-S) from manual subsurface drainage water sampling for the years indicated in the table. Mean and standard error in brackets for agrohydrological years (1 July – 30 June).

Field	pH 1973–2023	Conductivity 1973–2023 mS m^{-1}	Alkalinity 1981–2023 mmol L^{-1}	K^+ 1980–2010 mg L^{-1}	Na^{2+} 1980–2010 mg L^{-1}	Mg^{2+} 1980–2010 mg L^{-1}	Ca^{2+} 1980–2010 mg L^{-1}	Cl^- 1980–2010 mg L^{-1}	SO_4-S 1980–2010 mg L^{-1}
14AC	5.3 (0.06)	50 (1.5)	0.3 (0.03)	10.9 (0.01)	27.0 (0.02)	15.7 (0.01)	52.4 (0.03)	28.9 (0.02)	72.6 (0.04)
16Z	7.4 (0.02)	67 (0.8)	5.7 (0.09)	5.2 (0.00)	6.5 (0.00)	6.9 (0.00)	101.5 (0.07)	10.1 (0.01)	13.9 (0.01)
8C	7.6 (0.05)	46 (2.8)	3.8 (0.03)	1.3 (0.00)	3.6 (0.01)	5.3 (0.01)	55.8 (0.09)	6.2 (0.01)	3.6 (0.01)
17S	6.4 (0.03)	21 (1.0)	0.6 (0.05)	5.0 (0.00)	5.8 (0.01)	2.7 (0.00)	17.8 (0.02)	19.4 (0.02)	4.1 (0.01)
18T	6.0 (0.05)	70 (1.6)	1.7 (0.09)	2.0 (0.00)	6.4 (0.00)	7.2 (0.00)	138.3 (0.05)	10.7 (0.01)	93.0 (0.04)
1D	6.8 (0.04)	15 (0.5)	0.7 (0.03)	3.3 (0.00)	3.8 (0.00)	4.1 (0.00)	11.3 (0.01)	6.1 (0.01)	2.6 (0.00)
7E	7.5 (0.03)	52 (0.5)	4.3 (0.06)	2.1 (0.00)	7.0 (0.01)	18.0 (0.01)	56.4 (0.04)	12.5 (0.01)	11.6 (0.01)
20E	7.7 (0.02)	100 (1.2)	6.8 (0.15)	2.6 (0.00)	111.8 (0.08)	22.8 (0.01)	42.3 (0.03)	57.2 (0.04)	18.7 (0.01)
6E	7.7 (0.02)	75 (2.0)	4.5 (0.14)	1.5 (0.00)	10.9 (0.01)	9.6 (0.01)	83.7 (0.09)	30.1 (0.04)	20.0 (0.02)
21E	7.5 (0.02)	71 (1.1)	5.3 (0.15)	0.9 (0.00)	4.5 (0.00)	2.9 (0.00)	126.5 (0.11)	25.8 (0.04)	12.4 (0.01)
5O	7.1 (0.03)	37 (0.7)	2.3 (0.06)	1.9 (0.00)	16.6 (0.02)	14.5 (0.01)	18.5 (0.02)	9.9 (0.01)	7.1 (0.01)
4O	7.0 (0.04)	26 (0.5)	1.3 (0.05)	1.6 (0.00)	6.2 (0.01)	5.3 (0.00)	25.8 (0.02)	5.8 (0.01)	4.3 (0.00)
12N	6.7 (0.04)	31 (0.9)	1.0 (0.03)	4.1 (0.00)	8.4 (0.01)	2.8 (0.00)	29.2 (0.03)	13.9 (0.01)	8.5 (0.01)
11M	7.4 (0.03)	44 (1.2)	3.2 (0.12)	4.5 (0.00)	10.0 (0.01)	8.9 (0.01)	49.3 (0.04)	13.0 (0.01)	7.7 (0.01)
3M	7.3 (0.03)	75 (1.0)	3.6 (0.08)	18.9 (0.02)	13.2 (0.02)	4.5 (0.01)	76.6 (0.09)	21.0 (0.02)	15.6 (0.02)
2M	7.7 (0.02)	69 (0.8)	5.2 (0.09)	0.8 (0.00)	9.5 (0.01)	4.0 (0.00)	103.3 (0.09)	18.8 (0.02)	12.7 (0.01)

Table 6. Data for groundwater during the periods indicated in the table and at different depth (m). Annual mean concentration of pH, conductivity (mS m^{-1}) and alkalinity (mmol L^{-1}), nitrate-nitrogen (NO_3-N, mg L^{-1}), ammonium-nitrogen (NH_4-N, mg L^{-1}), potassium (K^+, mg L^{-1}), sodium (Na^{2+}, mg L^{-1}), magnesium (Mg^{2+}, mg L^{-1}), calcium (Ca^{2+}, mg L^{-1}), chloride (Cl^-, mg L^{-1}) and sulphate-sulphur (SO_4-S, mg L^{-1}). Mean and standard error in brackets for years (1 January – 31 December).

Field	Years	Depth m	pH 1973–2023	Conductivity 1973–2023 mS m^{-1}	Alkalinity 1981–2023 mmol L^{-1}	NO_3-N 1973–2023 mg L^{-1}	NH_4-N 1973–1982 mg L^{-1}	K^+ 1975–2010 mg L^{-1}	Na^{2+} 1980–2010 mg L^{-1}	Mg^{2+} 1980–2010 mg L^{-1}	Ca^{2+} 1980–2010 mg L^{-1}	Cl^- 1980–2010 mg L^{-1}	SO_4-S 1980–2010 mg L^{-1}
16Z	1977–2023	1.8	7.4 (0.03)	92 (1.8)	6.8 (0.06)	0.35 (0.117)	0.08 (0.016)	1.9 (0.07)	11 (0.7)	17 (0.5)	197 (4.2)	4 (0.3)	76 (3.6)
8C	1974–1997	2.0	7.5 (0.05)	57 (0.8)	5.6 (0.08)	7.84 (0.945)	0.03 (0.005)	2.5 (0.08)	9 (0.2)	15 (0.2)	98 (1.8)	9 (0.5)	8 (0.5)
		4.0	7.5 (0.05)	59 (1.0)	6.7 (0.06)	1.21 (0.517)	0.29 (0.236)	4.6 (0.11)	18 (0.3)	21 (0.2)	94 (1.7)	7 (0.3)	7 (0.2)
18T	1987–1997	2.0	6.7 (0.04)	63 (1.1)	6.3 (0.05)	0.55 (0.384)	n.d.	2.4 (0.07)	14 (0.2)	19 (0.3)	84 (2.4)	23 (0.2)	6 (0.9)
		4.0	7.0 (0.03)	54 (1.0)	5.0 (0.20)	0.31 (0.131)	n.d.	3.1 (0.15)	19 (0.5)	14 (1.1)	70 (2.6)	26 (0.6)	5 (1.3)
1D*	1973–2023	2.2	7.5 (0.02)	42 (0.9)	4.3 (0.07)	1.68 (0.599)	0.03 (0.005)	2.1 (0.08)	28 (0.5)	24 (0.3)	34 (0.5)	4 (0.2)	9 (0.3)
		4.1	7.5 (0.03)	39 (0.4)	3.4 (0.04)	1.24 (0.104)	0.03 (0.004)	7.5 (0.10)	16 (0.3)	18 (0.2)	42 (0.5)	7 (0.1)	11 (0.1)
7E	1977–2023	2.5	7.9 (0.02)	63 (0.6)	6.2 (0.02)	0.01 (0.001)	0.02 (0.003)	5.7 (0.10)	11 (0.2)	29 (0.3)	89 (1.1)	8 (0.2)	14 (0.3)
		4.0	7.8 (0.02)	64 (0.6)	6.2 (0.02)	0.02 (0.007)	0.03 (0.004)	5.8 (0.12)	11 (0.2)	29 (0.3)	87 (1.3)	9 (0.3)	16 (0.2)
6E*	1974–2023	2.0	7.3 (0.06)	45 (2.5)	1.7 (0.12)	8.63 (1.123)	0.02 (0.003)	2.2 (0.28)	25 (2.9)	6 (0.4)	55 (4.0)	26 (3.4)	21 (2.3)
		4.0	7.7 (0.02)	57 (0.5)	5.4 (0.07)	0.92 (0.248)	0.03 (0.008)	2.3 (0.07)	26 (0.4)	15 (0.3)	82 (1.1)	16 (0.4)	7 (0.2)
5O	1975–2023	2.0	7.1 (0.03)	38 (0.9)	2.7 (0.12)	5.52 (1.309)	0.03 (0.004)	3.6 (0.08)	39 (1.6)	19 (0.4)	18 (0.4)	14 (0.6)	7 (0.5)
		4.0	7.3 (0.03)	60 (0.3)	6.3 (0.03)	0.11 (0.037)	0.04 (0.010)	10.3 (0.25)	62 (0.6)	32 (0.3)	30 (0.3)	20 (0.3)	2 (0.2)
4O*	1975–2023	2.0	6.9 (0.04)	32 (0.5)	1.7 (0.07)	5.91 (0.861)	0.03 (0.004)	1.3 (0.07)	39 (0.9)	12 (0.2)	12 (0.4)	7 (0.5)	16 (1.1)
		4.0	7.0 (0.03)	33 (0.6)	2.0 (0.05)	4.42 (0.313)	0.03 (0.004)	2.3 (0.06)	41 (1.0)	13 (0.2)	16 (0.6)	8 (0.2)	13 (0.4)
12N*	1976–2023	1.7	6.6 (0.05)	37 (1.4)	1.1 (0.10)	6.17 (0.805)	0.02 (0.004)	5.0 (0.16)	22 (1.2)	5 (0.2)	33 (1.4)	23 (1.0)	21 (0.6)
		5.5	7.8 (0.02)	164 (7.8)	9.1 (0.33)	0.33 (0.028)	0.24 (0.052)	12.6 (0.53)	310 (20.9)	17 (0.8)	32 (1.1)	231 (15.3)	25 (0.8)
11M	1977–2023	3.6	7.8 (0.02)	81 (1.2)	8.6 (0.17)	0.20 (0.021)	n.d.	11.4 (0.45)	105 (4.5)	30 (0.5)	45 (1.3)	16 (0.3)	7 (0.5)
		5.8	7.7 (0.03)	79 (0.4)	8.2 (0.07)	0.17 (0.024)	n.d.	11.4 (0.32)	100 (3.5)	30 (0.5)	48 (1.9)	16 (0.4)	7 (0.5)
3M*	1974–1986	1.0	7.6 (0.07)	78 (2.7)	3.8 (0.14)	12.76 (1.642)	0.03 (0.004)	11.2 (1.48)	26 (1.4)	5 (0.3)	146 (12.5)	39 (4.0)	49 (3.9)
		4.0	7.9 (0.05)	33 (0.7)	3.0 (0.09)	0.13 (0.018)	0.04 (0.004)	1.9 (0.11)	19 (0.8)	7 (0.2)	47 (1.3)	12 (0.3)	4 (0.3)
2M	1973–2023	2.9	7.4 (0.02)	91 (1.2)	7.6 (0.14)	1.63 (0.332)	n.d.	1.7 (0.06)	40 (1.0)	11 (0.4)	159 (2.8)	81 (2.8)	15 (0.5)
		5.6	7.4 (0.02)	87 (1.1)	7.1 (0.11)	0.55 (0.134)	0.07 (0.027)	1.3 (0.08)	43 (1.8)	8 (0.3)	153 (3.2)	75 (3.3)	15 (0.7)

The field has more ground water pipes than presented here. n.d. – not determined.

Manninen et al. (2018) where the fraction of dissolved organic carbon (DOC) was between 25 and 59 kg ha^{-1} in drainage water from experimental fields in Finland.

Field 18 T, with its peat soil, deviated from the other fields dominated by mineral soils, for instance, by having the highest load of TOC (Table 3), low pH (Table 5), and the highest concentrations of Ca^{2+} and SO_4-S (Table 5).

Groundwater

The composition of the groundwater is affected by soil management practices, soil and mineral type, depth of the pipe, and if the pipe is located at a recharge or discharge area. A shallow pipe is mainly affected by downward movements of water, whilst a deeper pipe can be affected by through-flow water from e.g. a forest located up-wards the field. Consequently, shallow groundwater can be more affected by agricultural management practices and cohere with drainage water, compared to deeper groundwater. For example, after a dry period, N concentrations in shallow groundwater and drainage water can be elevated due to the accumulation of mineral N in the soil during the drought and downward transport following precipitation. On average across all fields, the concentration of NO_3-N was almost always higher in shallow groundwater compared to deeper groundwater, often much higher (Table 6). For instance, the 2 m deep groundwater at field 8C had a NO_3-N concentration that was approximately six times higher than the 4 m deep groundwater (7.84 and 1.21 mg L^{-1}, respectively, Table 6).

The differences in water chemistry between shallow and deep groundwater can also be due to a change in soil type over depth. This is the case in field 12N, where the shallow groundwater pipe (1.7 m) was located in approximately 2 m of deep course sandy soil, whilst the deeper groundwater pipe (5.5 m) was in clay soil. Therefore, the deeper groundwater had higher conductivity and high values of Na^{2+} and Cl^{-} due to the low laying area and marine origin of the clay (Table 6). Furthermore, this change in soil type means that most precipitation stayed in the sandy layer and thus, ended up in the drainage system (Table 4).

The repair of a damaged drainage system in field 5O, resulted in higher NO_3-N concentrations in both subsurface drainage water and in shallow (2 m depth) ground water, whilst the deeper ground water (4 m depth) was not affected by the change in water movement (Wesström et al. 2015). The study by Wesström et al. (2015) highlighted how soil nutrients are affected by changes in water balance and the depth to which this effect can be seen. Deeper groundwater has slower fluctuations, and long-term measurements are crucial to evaluate the effect of agricultural management practices and climate change on ground water quality.

According to the Swedish Food Agency, NO_3 concentration in drinking water should not exceed 50 mg N L^{-1} (corresponds to 11.5 mg NO_3-N L^{-1}) and only one of the locations exceeds this value (Field 3M, 1 m depth, Table 6). Some of the shallow groundwater pipes exceed, on a long-term average, the level of good drinking water (4.6 mg NO_3-N L^{-1}, Swedish Food Agency) but in general, none of the locations had poor groundwater quality (Table 6).

Conclusions

This type of long-term monitoring of the agricultural landscape and its impact on the environment is crucial in assessing and understanding the presence and predicting the future. Time series that commenced when the temperature in Sweden was 1–1.5°C colder than today are important in following changes of the water quality impacted by agriculture. As climate change progresses, one can expect that the pattern of leaching losses in the northern parts of Sweden will be similar to the patterns we can currently see in the southern parts. Field 14AC and 16Z have a major task in representing the northern two thirds of Sweden that will experience shorter winter periods with less snow, shallower ground frost, and potentially increased freeze–thaw cycles that all of which can lead to increased leaching losses. The observation fields present a magnitude of variation in water quality values related to climate, soil types, cropping systems, etc. The observation fields methodology can be used in any fields with tile-drainage, and the presented data can be related and applied to other locations where similar climatic, agronomic and soil conditions occur. As the network of observation fields are unique in the world, they can play an important role also internationally. Furthermore, the observation fields can help policy makers and authorities to evaluate how regulations and subsidies have been implemented by the farmers, and how new regulations can be designed. The fields can be used for follow-up on mitigation measures for nutrient leaching losses from agriculture, since the fields provide information about agricultural practices and their environmental impact in a long-term perspective. The observation fields also offer information that advisors and farmers can utilise to develop agricultural practices for improved environmental protection. Nevertheless, there are many factors and processes left to explore regarding how nutrient losses and water chemistry in drainage water and groundwater are controlled and regulated at a site-specific level. The observation fields offer a diverse set of locations with a well-known history, and thus profound opportunities for future applied research.

Acknowledgements

We wish to send our greatest gratitude to the farmers and landowners at the sites. A special thanks to the persons taking the water samples every other week, year after year, and the water laboratory at the Department of Aquatic sciences and Assessment, SLU, analyzing the water. A special regard to Arne Gustafsson, who was part of starting the observation fields in the 1970s and continued the work until his retirement several decades later. Thanks also to all other persons working with the observation fields in one way or the other. The observation fields are part of the Environmental Monitoring Program financed by the Swedish Environmental Protection Agency.

Authors contributions

Lisbet Norberg: project administration, conceptualisation, formal analysis, visualisation, writing – original draft. Maria Blomberg: data curation, resources, writing – review and editing. Helena Linefur: project administration, data curation, writing – review and editing. Katarina Kyllmar: data curation, writing – review and editing. All co-authors have critically reviewed and approved the final version.

Disclosure statement

No potential conflict of interest was reported by the author(s).

Funding

This work was supported by the Swedish Environmental Protection Agency.

Notes on contributors

Lisbet Norberg is a researcher and specialist in soil science, with focus on long-term monitoring of water quality in arable field runoff. Her research interest is the impact of agricultural management practice, soil type and climate on nutrient leaching in the agricultural landscape.

Maria Blomberg is a research engineer with focus on managing data, water sampling and water flow measurements from agricultural fields and agricultural catchment areas.

Helena Linefur is a researcher with the focus on environmental monitoring on small agricultural catchments. Her focus lies within nutrient losses from different soils and cropping systems.

Katarina Kyllmar is a Senior Environmental Specialist working in the area between research and society. Her research focuses on water and nutrient management in the agricultural landscape.

References

Bechmann M. 2014. Long-term monitoring of nitrogen in surface and subsurface runoff from small agricultural dominated catchments in Norway. Agric Ecosyst Environ. 198:13–24.

Boesch DF. 2002. Challenges and opportunities for science in reducing nutrient over-enrichment of coastal ecosystems. Estuaries. 25:886–900.

Brink N, Gustafson A. 1970. Kväve och fosfor från skog åker och bebyggelse (Nitrogen and phosphorus from forest, agriculture and urban areas). Vattenvård Nr. 1 (In Swedish). Institutionen för markvetenskap, Lantbrukshögskolan.

Conley DJ et al. 2009. Controlling eutrophication: nitrogen and phosphorus. Science. 323(5917):1014–1015.

Djodjic F, Bergström L. 2005a. Conditional phosphorus index as an educational tool for risk assessment and phosphorus management. Ambio. 34(4):296–300.

Djodjic F, Bergström L. 2005b. Phosphorus losses from arable fields in Sweden – effects of field-specific factors and long-term trends. Environ Monit Assess. 102:103–117.
Djodjic F et al. 2023. Soils potentially vulnerable to phosphorus losses: speciation of inorganic and organic phosphorus and estimation of leaching losses. Nutr Cycling Agroecosyst. 127:225–245.
Ezzati G, Kyllmar K, Barron J. 2023. Long-term water quality monitoring in agricultural catchments in Sweden: impact of climatic drivers on diffuse nutrient loads. Sci Total Environ. 864:160978.
Johnsson H et al. 2023. Läckage av näringsämnen från svensk åkermark - Beräkningar av normalläckage av kväve och fosfor för 2019. Ekohydrologi 178 (in Swedish). Swedish University of Agricultural Sciences.
Kyllmar K et al. 2023. Nitrogen and phosphorus losses in Nordic and Baltic agricultural monitoring catchments – spatial and temporal variations in relation to natural conditions and mitigation programmes. Catena. 230:107205.
Kyllmar K, Forsberg LS, Andersson S, Martensson K. 2014. Small agricultural monitoring catchments in Sweden representing environmental impact. Agric Ecosyst Environ. 198:25–35.
Lackner A, Klöffel T, Jansson P-E, Barron J. 2023. The impact of freeze-thaw dynamics on nutrient leaching in a Swedish agricultural field Poster presentation, EGU, 23-28 April, Vienna, Austria.
Liu J, Bechmann M, Eggestad HO, Øgaard AF. 2023. Twenty years of catchment monitoring highlights the predominant role of long-term phosphorus balances and soil phosphorus status in affecting phosphorus loss in livestock-intensive regions. Sci Total Environ. 898(1–3):165470.
Liu J, Ulén B, Bergkvist G, Aronsson H. 2014. Freezing-thawing effects on phosphorus leaching from catch crops. Nutr Cycling Agroecosyst. 99:17–30.
Manninen N, Soinne H, Lemola R, Hoikkala L, Turtola E. 2018. Effects of agricultural land use on dissolved organic carbon and nitrogen in surface runoff and subsurface drainage. Sci Total Environ. 618:1519–1528.
Norberg L, Aronsson H. 2022. Mitigating phosphorus leaching from a clay loam through structure liming. Acta Agric Scand Sect B Soil Plant Sci. 72(1):987–996.
Norberg L, Aronsson H. 2024. Effects of spring and autumn tillage, catch crops, and pig manure application on long-term nutrient leaching from a loamy sand. Eur J Agron. 156:127156.
Norberg L, Aronsson H. 2025. Conventional and reduced tillage, and set-aside land – effects on nitrogen, phosphorus and potassium subsurface leaching from a clay soil during 9 years. Soil Tillage Res. 250:106515.
Norberg L, Linefur H, Andersson S, Blomberg M, Kyllmar K. 2022. Nutrient losses over time via surface runoff and subsurface drainage from an agricultural field in northern Sweden. J Environ Qual. 51(6):1235–1245.
Sandström S et al. 2020. Particulate phosphorus and suspended solids losses from small agricultural catchments: links to stream and catchment characteristics. Sci Total Environ. 711:134616.
Sandström S et al. 2023. Distributed dynamic modelling of suspended sediment mobilization and transport from small agricultural catchments. Front Environ Sci. 11:1196048.
Ulén B. 2006. A simplified risk assessment for losses of dissolved reactive phosphorus through drainage pipes from agricultural soils. Acta Agric Scand Sect B Soil Plant Sci. 56(4):307–314.
Ulén B, Johansson G. 2009. Long-term nutrient leaching from a Swedish arable field with intensified crop production against a background of climate change. Acta Agric Scand Sect B Soil Plant Sci. 59(2):157–169.
Ulén B, Johansson G, Kyllmar K. 2001. Model predictions and long-term trends in phosphorus transport from arable lands in Sweden. Agric Water Manag. 49(3):197–210.
Ulén B, Johansson G, Kyllmar K, Stjernman Forsberg L, Torstensson G. 2015. Lagged response of nutrient leaching to reduced surpluses at the field and catchment scales. Hydrol Processes. 29(13):3020–3037.
Ulén B, Johansson G, Simonsson M. 2008a. Changes in nutrient leaching and groundwater quality during long-term studies of an arable field on the Swedish south-west coast. Hydrol Res. 39(1):63–77.
Ulén B, Johansson G, Simonsson M. 2008b. Leaching of nutrients and major ions from an arable field with an unfertilized fallow as infield buffer zone. Acta Agric Scand Sect B Soil Plant Sci. 58(1):51–59.
Ulén B, Lewan E, Kyllmar K, Blomberg M, Andersson S. 2019. Impact of the North Atlantic oscillation on Swedish winter climate and nutrient leaching. J Environ Qual. 48(4):941–949.
Ulén B, Persson K. 1999. Field-scale phosphorus losses from a drained clay soil in Sweden. Hydrol Processes. 13(17):2801–2812.
Ulén B, Von Brömssen C, Johansson G, Torstensson G, Forsberg LS. 2012. Trends in nutrient concentrations in drainage water from single fields under ordinary cultivation. Agric Ecosyst Environ. 151:61–69.
Ulén B, Wesström I, Johansson G, Forsberg LS. 2014. Recession of phosphorus and nitrogen concentrations in tile drainage water after high poultry manure applications in two consecutive years. Agric Water Manag. 146:208–217.
Uusitalo R, Lemola R, Turtola E. 2018. Surface and subsurface phosphorus discharge from a clay soil in a nine-year study comparing no-till and plowing. J Environ Qual. 47(6):1478–1486.
Villa A, Djodjic F, Bergström L, Wallin M. 2012. Assessing soil erodibility and mobilization of phosphorus from Swedish clay soils – comparison of two simple soil dispersion methods. Acta Agric Scand Sect B Soil Plant Sci. 62:260–269.
Wallman M, Delin S. 2022. Nitrogen leaching from tile-drained fields and lysimeters receiving contrasting rates and sources of nitrogen. Soil Use Manag. 38(1):596–610.
Wesström I, Ulén B, Joel A, Johansson G, Forsberg LS. 2015. Effects of tile drainage repair on nutrient leaching from a field under ordinary cultivation in Sweden. Acta Agric Scand Sect B Soil Plant Sci. 65:228–238.