Science of the Total Environment 898 (2023) 165470

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

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

Jian Liu *, Marianne Bechmann, Hans Olav Eggestad, Anne Falk Øgaard

The Norwegian Institute of Bioeconomy Research (NIBIO), P.O. Box 115, 1431, Ås, Norway

A R T I C L E I N F O

Editor: José Virgílio Cruz

Keywords:
Animal agriculture
Catchment monitoring
Eutrophication
Phosphorus balance
Soil test phosphorus
Water quality

A B S T R A C T

Livestock husbandry has raised enormous environmental concerns around the world, including water quality issues. Yet there is a need to document long-term water quality trends in livestock-intensive regions and reveal the drivers for the trends based on detailed catchment monitoring. Here, we assessed the concentration and load trends of dissolved reactive phosphorus (DRP) in streamwater of a livestock-intensive catchment in southwestern Norway, based on continuous flow measurements and flow-proportional composite water sampling. Precipitation and catchment-level soil P balance were monitored to examine the drivers. At the field level, moreover, the relationship between soil P balance and soil test P (measured using the ammonium lactate extraction method, P-AL) was assessed. Results showed that on average of 20 years 95 % of the P was applied to the catchment during March–August, when 40 % of annual precipitation and 25 % of annual discharge occurred. The low runoff helped reduce P loss following P applications. However, flow-weighted annual mean DRP concentration significantly increased with increasingly cumulative soil P surplus ($R^2 = 0.55$, $p = 0.0002$). With a mean annual P surplus of 8.8 kg ha^{-1}, the annual mean DRP concentration (range: 49–140 μg L^{-1}; mean: 80 μg L^{-1}) and annual DRP load (range: 0.35–1.46 kg ha^{-1}; mean: 0.65 kg ha^{-1}) significantly increased over the 20-year monitoring period ($p =$ 0.001 and 0.0003, respectively). At the field level, P-AL concentrations were positively correlated with soil P balances ($R^2 = 0.48$, $p < 0.0001$), confirming the long-term impact of P balances on the risks of P loss. The study highlights the predominant role of long-term P balances in affecting DRP loss in livestock-intensive regions through the effect on soil test P.

1. Introduction

Livestock is an important source of proteins for humans. However, in livestock-intensive regions the production has raised environmental concerns, among which undesirable nutrient losses from livestock farms and their subsequent consequences on eutrophication in downstream water bodies have gained worldwide attention (Li et al., 2015; Sharpley et al., 2015; Kleinman et al., 2020). In livestock-intensive regions, particularly, manure is often an important (if not predominant) source of nutrients applied to the agricultural land and also a key contributor of nutrient loads in the stream (Johnes et al., 2022; Kleinman et al., 2022). In Norway, it has been estimated that livestock manure accounts for roughly 50 % of the total phosphorus (P) and 25 % of the total nitrogen (N) supplies to agricultural lands (Kolle and Oguz-Alper, 2020). In livestock-intensive catchments, manure may constitute up to 90 % of the total P and 50 % of total N inputs to the catchments (Bechmann, 2014a). In such catchments, P surplus (more P is applied than removed) concerns over water quality have raised much discussion on P management because P is widely regarded as a limiting nutrient for eutrophication in freshwater ecosystems (Schindler et al., 2016).

Norway implements a series of measures for livestock and manure management. In some regions, livestock and manure are restricted by rules of the European Nitrates Directive (European Economic Community, 1991) and for the whole country according to the Water Framework Directive (European Commission, 2000). National legislations in Norway restricts the density of animals on a livestock farm to 2.5 animal units per hectare manure spreading area (2.5 AU ha^{-1}, equivalent to 35 kg P ha^{-1}) (Andersen et al., 2014). The regulation is implemented with the aim to reduce surplus manure P inputs on livestock farms, because large quantities of manure input increase the risk of P loss not only over

* Corresponding author.
E-mail address: jian.liu@nibio.no (J. Liu).

https://doi.org/10.1016/j.scitotenv.2023.165470
Received 12 May 2023; Received in revised form 20 June 2023; Accepted 9 July 2023
Available online 13 July 2023

a crop season or a year but also over a long term when the P input continuously builds up soil test P (a measurement of plant available form of P) (Bechmann, 2014b). Typically, concentration of soil test P increases with increasing P balances (Bechmann, 2014b), which consequently increases the concentrations of water-soluble P in the soil (Kristoffersen et al., 2020) and the concentrations of dissolved P in runoff (Liu et al., 2021). Although the regulation has been in place since 1989 (FOR-1989-03-01-151), soil P surplus remains a concern in livestock-intensive regions, especially when the historically large rates of manure applications are considered (Vagstad, 2014).

Although the impacts of soil P balances on the risk of P loss have been demonstrated in both Norwegian and other studies (e.g., Ekholm et al., 2005; Bechmann, 2014b; Bergström et al., 2015; Hanrahan et al., 2019; Bos et al., 2021), most of the studies were conducted at the scales ranging from soil cores to fields. Still, there is a large knowledge gap regarding the impacts at the catchment scale, which involves greater complexity of agronomic, hydrological and biogeochemical factors that influence P loss than at smaller spatial scales. Catchment monitoring plays an indispensable role in detecting water quality trends and understanding the drivers and management needs for altering the trends. Ideally, water quality monitoring in the streamwater should be combined with monitoring of weather conditions, catchment properties and field management to explicitly understand the drivers. For this purpose, a monitoring network of 12 agricultural catchments (the Norwegian Agricultural Environmental Monitoring Programme – JOVA) were established across Norway since 1990s to monitor water quality trends and study the drivers (Bechmann and Deelstra, 2013).

Based on 20 years of catchment monitoring, in this study, we attempted to understand the long-term water quality trend as affected by field management (specifically P management) and precipitation in a Norwegian livestock-intensive catchment. Given dissolved reactive P (DRP) is the most biologically available form of P for primary production in freshwater ecosystems (Reynolds and Davies, 2001), here we focused our water quality analyses on this P form. Specifically, our objectives were to: (1) assess the long-term trend of DRP in streamwater of the livestock-intensive catchment; and (2) examine the impacts of precipitation and soil P balances on the DRP trends in streamwater.

2. Materials and methods

2.1. Study catchment

The study was conducted in an agriculture-dominated catchment located in Rogaland County in southwestern Norway, where intensive livestock production is common (Bechmann et al., 2021). The catchment has a total area of approximately 100 ha, about 90 % of which is agricultural land. The catchment has an elevation ranging from 35 to 100 m above the sea level, and dominant soils as loamy sand and organic soils developed on nutrient poor glacial till (Hauken and Kværnø, 2013). The main soils are classified in the World Reference Base for Soil Resources as Endostagnic Umbrisol, Umbric Endostagnic Podzol, Umbric Gleysol and Sapric Histosol (FAO/ISRIC/ISSS, 1998). The region has a coastal climate, with high annual precipitation and relatively mild winters as compared to many other regions in Norway (Wenng et al., 2021). During 1992–2021, the catchment has an average of 240 growing days, mean annual temperature of 8.3 °C and mean annual precipitation of 1370 mm. Mean monthly temperature ranges from 1.7 °C in February to 16.0 °C in July. Due to the generally mild weather conditions, annual evapotranspiration is typically below 500 mm, which in combination with the high precipitation results in a large amount of annual water surplus. The hydrology of the catchment is dominated by subsurface flow through tile drainage networks, which are widespread in the catchment to drain excess rain water and in rare cases snow melt. The weather conditions of the southwestern coastal regions favor forage production rather than small grain crops, which has led to intensive grass-based livestock production in these regions, including our study catchment where grass (including clovers) and small grain crop area account on average for 93 % and 6 % of the total agricultural area, respectively. Main facts of the catchment are summarized in Table S1.

Livestock production in the study catchment is dominated by cattle (mostly milking cows), followed by small numbers of pigs, poultry, sheep/goats and horses. The animals are mainly raised with indoor feeding operations, where manure is collected and applied mechanically to cropland. During warmer months (usually April–October), cattle and sheep/goats stay partly outside for grazing. The crops are mainly perennial grasses (in combination with various amounts of clover) and, to a small extent, annual grasses and small grains. Typically, perennial grasses (and clovers) grow for 5–7 years before being replaced. Each year, the grasslands receive multiple times of nutrient applications and are usually harvested about three times. For preparing a new seedbed, the soil is tilled with methods ranging from harrowing to deep plowing.

2.2. Catchment monitoring

The catchment consists of 13 farms and 105 individual fields. Monitoring in the catchment dated back to 1985 but became more systematic in 1992 as part of the JOVA monitoring programme. Since 1992, water discharge and water quality (including suspended sediments, different forms of P and N, pesticides, pH, and conductivity) were monitored at the outlet of the catchment. Specifically, water levels were recorded automatically at a 1-min time step using a pressure transducer in combination with a Campbell datalogger, and were converted to discharge based on a known head-discharge relationship of the monitoring weir (Deelstra et al., 2012). For the water quality measurements, water samples were collected using a flow-proportional composite sampling approach, in which a 50 mL water sample was taken from the stream each time a certain volume of water has passed the monitoring station. The volume varied depending on weather conditions for practical reasons, e.g., avoiding overfilling the container for storing water samples in heavy rain events (Cassidy et al., 2018). The water samples collected at each individual sampling event were stored in a 20 L container that was situated in a refrigerator (4 °C) to prevent chemical and biological reactions. At an approximate two-week time interval, a composite sample was collected from the container after thoroughly mixing the water, and delivered to the laboratory for chemical analyses. After sampling, the container was emptied and used for collecting the next sample. Given DRP is the most biologically available form of P for primary production in freshwater ecosystems (Reynolds and Davies, 2001), we focused the study on this P form. Specifically, DRP was measured colorimetrically on filtered water samples (0.45 μm), using an Alpkem Flow Solution IV Automated wet chemistry system (O.I. Analytical, College Station, Texas) at a wavelength of 880 nm (Standard Norway, 2019).

To understand drivers of water quality trends, air temperature and precipitation were measured hourly at the catchment outlet. Information of field management practices was synthesized each year through a survey with individual farmers in the catchment. For each field and each year, management information on crop (i.e., crop type, seeding date, harvest date, yield, and straw management), tillage (i.e., date and method), nutrient (i.e., fertilizer and manure type, and application rate, timing and method), and animal (i.e., animal type, number of grazing animals, and days of grazing) was collected. In 2005, a soil survey was conducted in 28 fields to determine concentrations of soil P-AL, which is a standard measurement of plant available P for Norwegian agricultural soils using the ammonium lactate extraction method (Egner et al., 1960).

2.3. Data analyses

Soil P balance (*SPB*, kg ha^{-1}) was calculated for each field and each year using Eq. 1, and the field-level balances were area-weighted to estimate the catchment-level P balance.

$$SPB = I_nP_{fert} + I_nP_{manu} + I_nP_{graz} – E_xP_{harv} – E_xP_{graz} \quad (1)$$

Where, I_nP_{fert} (kg ha^{-1}) was the P input through mineral fertilizer applications, and was calculated as the product of the fertilizer rate reported by the farmer in the survey and the standard P concentration in that fertilizer type. Similarly, I_nP_{manu} (kg ha^{-1}) was the P input through mechanical manure applications, and was calculated as the product of the farmer's reported manure application rate and estimated P concentration in the corresponding manure type. For manure P concentrations, we used a book value for a specific manure type on average of many manure samples collected in our study region or across the country (0.55, 0.66 and 8.1 kg P Mg^{-1} manure for cattle, pig and poultry, respectively; Daugstad et al., 2012). The value was adjusted based on typical total solid content in the manure and the water added by the farmer to the manure before the application in the field. I_nP_{graz} (kg ha^{-1}) was the P input from livestock manure excreted at grazing, and was calculated as the product of the number of grazing livestock, the time length of grazing in the year and the manure P production rate per unit time. In one year, a milking cow was estimated to produce 14 kg P (FOR 2003-07-04-951). E_xP_{harv} (kg ha^{-1}) was the P exported from the soil through the harvest of both crop and crop residue. Most fields in our study catchment were used for producing animal feed, and E_xP_{harv} was calculated as the product of the farmer's reported crop yield and the estimated plant P concentration. Here, we used the yield data from the period of 1992–2001 (10–12 Mg ha^{-1}) and applied the average to later years, considering the estimated yield was most reliable for those earlier monitoring years. The plant P concentration was from book values on average of many samples analyzed in the region or in the country, i.e., 0.25–0.30 % for different types of crops (Heje, 2000). E_xP_{graz} (kg ha^{-1}) was the P exported from the soil through livestock grazing, and was estimated in a similar way to I_nP_{graz}. Several data sources could contribute to the uncertainty of the estimated P balances, including the farmers' reported grass yields which is difficult to quantify/estimate (Bakken and Steinshamn, 2022), amounts of applied manure, and book values of P concentrations in grass and manure.

Measured weather and discharge data were aggregated to daily values. For water quality measurements, each composite water sample represented the average nutrient concentration in streamwater over the sampling period. Daily discharge and the DRP concentration for the day were used to calculate daily P load, which was further cumulated to obtain monthly and annual loads. A flow-weighted mean DRP concentration was thereafter calculated based on the DRP load and the total discharge for a specific period of interest, i.e., monthly or annually. In this study, we used 20 years' DRP data that were of the best data quality, i.e., 1995/1996–1998/1999, 2004/2005–2005/2006, and 2007/2008–2020/2021. Each year was defined as from March 1 of the year to February 28 (or 29) of the next year, conforming the fact that manure application in the catchment of any year often started in March. Each year was divided into two seasons: manure-intensive season spanning from March to August, and non-intensive season from September to February.

Statistical analyses were conducted using SAS 9.4 (SAS Institute, 2012). Correlations between soil P-AL and soil P balance, between annual DRP load and annual discharge, and between flow-weighted annual mean DRP concentration and cumulative P balance (i.e., the sum of annual P balances for the period from 1995/1996 to a specific year; the annual P balance was consistently positive over the 20 years in this study, i.e., P surplus.) were assessed using the Pearson's correlation analysis in a Proc GLM (generalized linear model) procedure. Temporal trends of annual discharge, cumulative soil P surplus, annual DRP load, and flow-weighted annual mean DRP concentration were assessed using the Mann-Kendall test in a Proc CORR procedure. A Proc UNIVARIATE procedure was used to confirm data normality before the above analyses. A significance level of $\alpha = 0.05$ was used throughout the study unless specific p values were given.

3. Results

3.1. Long-term P inputs and balances

On average of the 20 monitoring years, the agricultural land in the catchment was estimated to receive 40.3 kg P ha^{-1} year^{-1} from all P sources (Table S1). Mineral fertilizers, livestock grazing and mechanical manure applications contributed 10 %, 16 % and 74 % of the total P input to the catchment, respectively. Of the total manure P input, cattle, pig and poultry accounted for 82 %, 8 % and 8 %, respectively (Table S1). The estimated total P input peaked at 52.8 kg P ha^{-1} in 2007/2008 and then declined to slightly below 35 kg P ha^{-1} in recent years. As described above, there was a high uncertainty in the crop yield data in later monitoring years. The data for 1992–2001, which were most reliable, showed that on average 31.5 kg P ha^{-1} year^{-1} was removed from the soil through crop harvests. Applying the average crop P removal to all monitoring years showed a P surplus ranging from 0.4 to 21.3 kg P ha^{-1} year^{-1} (average: 8.8 kg P ha^{-1} year^{-1}) (Fig. 1). This resulted in a cumulative P balance of 176 kg P ha^{-1} over the 20-year period.

3.2. Relationship between soil P balances and P-AL concentrations (field-level results)

Soil P-AL concentrations, which were measured in 28 fields during the autumn of 2005, ranged from 10 to 450 mg P kg^{-1} soil (Fig. 2). According to the Norwegian fertilizer recommendation guidelines (NIBIO, 2023), 75 % of the fields had higher P-AL concentrations than the threshold value for the "very high" soil P-AL category (i.e., 140 mg kg^{-1}). No yield response to P application would be expected at P-AL > 140 mg kg^{-1}. An analysis of the relationship between the P-AL concentrations and average soil P balance of the respective fields for the period of 1992–2005 showed that the P-AL values increased significantly with increasing P balances ($R^2 = 0.48$, $p < 0.0001$), despite a likely difference in the starting value of P-AL in 1992.

3.3. Concentrations and loads of dissolved reactive P in streamwater

3.3.1. Trends of daily values

A trend analysis of daily P application rates, for which each daily value was an average of 20 years of results for the same calendar date, showed that P input to the catchment was spread out during March–August with daily P input ranging from 0 to 1 kg ha^{-1} (Fig. 3A). During September–February, P input was zero on most days. There were three groups of P input peaks around March/April, June, and July/August, reflecting applications of nutrients to the forage crops for the first, second, and third main cut, respectively. The 20-year average daily precipitation ranged from 0.2 to 11.6 mm, and daily discharge ranged from 0.3 to 6.1 mm (Fig. 3A). While precipitation peaks spread out across the year, most of the discharge peaks occurred during September–February, when precipitation was relatively higher, but evapotranspiration was much lower than March–August.

Interestingly, the trend of daily DRP concentration in streamwater did not seem to follow the patten of either the P input to the catchment or the discharge (Fig. 3B). There was a tendency of DRP concentration increases following manure applications. The increases were not immediate possibly because of the delays by hydrological conditions, i.e., no big runoff events immediately after manure applications. The high peaks in January could be due to P release from grasses, as many studies showed release of P from plant materials during/after freeze-thaw cycles (Liu et al., 2019). The DRP concentration did not exhibit an apparent seasonality. The median daily DRP concentrations were 79 μg L^{-1} during March–August (range: 39–151 μg L^{-1}) and 74 μg L^{-1} during September–February (range: 56–195 μg L^{-1}), respectively. However, the trend of DRP load appeared to follow the pattern of discharge (Fig. 3C), suggesting the critical influence of discharge on DRP load from the

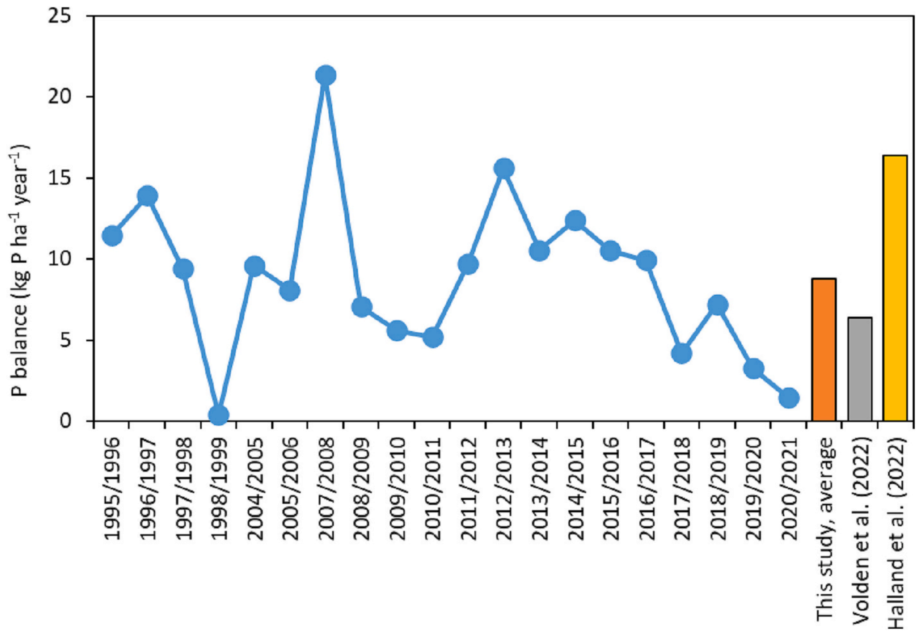

Fig. 1. Estimated annual P balance in the study catchment over the 20-year study period, and the mean annual average P balance in comparison to the average farm-gate level P balances in the region reported by Volden et al. (2022) for seven farms in 2021 and by Halland et al. (2022) for 20 farms during 2018–2020.

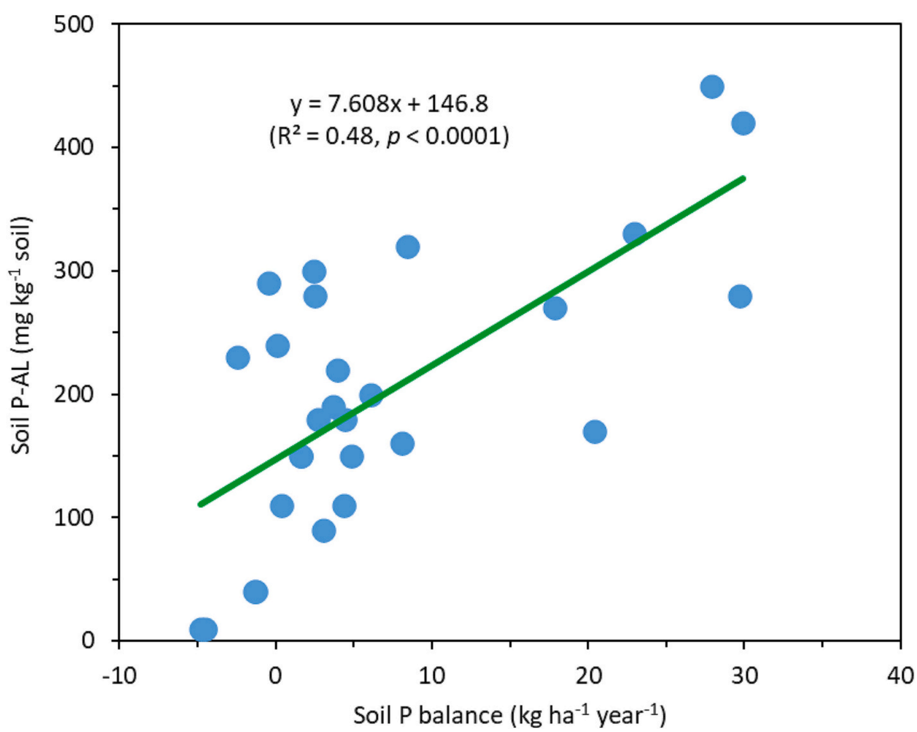

Fig. 2. Relationship between soil P-AL concentrations that were measured in the autumn of 2005 and average soil P balances of the respective yields for years 1992–2005 (n = 28; each datapoint represents a crop field).

catchment. Daily DRP load ranged from 0.1 to 4.5 g ha^{-1} (median: 0.9 g ha^{-1}) during March–August and from 1.1 to 6.6 g ha^{-1} (median: 2.5 g ha^{-1}) during September–February.

3.3.2. Trends of monthly and seasonal values

Precipitation, discharge, catchment-level P input and DRP load in streamwater all exhibited a strong seasonality, though with different patterns. Monthly precipitation, as an average of 20 years of results for the same calendar month, ranged from 65 mm in May and June to 169 mm in October (Fig. 4A). Correspondingly, monthly discharge ranged from 13 mm in June to 122 mm in October (Fig. 4A). While the manure non-intensive season (i.e., September–February) accounted for 60 % of the annual precipitation, it contributed 75 % of the annual discharge. Discharge was high throughout September–February, at 76–122 mm month^{-1}, due to low evapotranspiration in these months. As described above, most of the P was applied to the catchment during the manure-intensive season (March–August). During the manure-intensive and non-intensive seasons, total P inputs were on average 38.6 and 1.8 kg

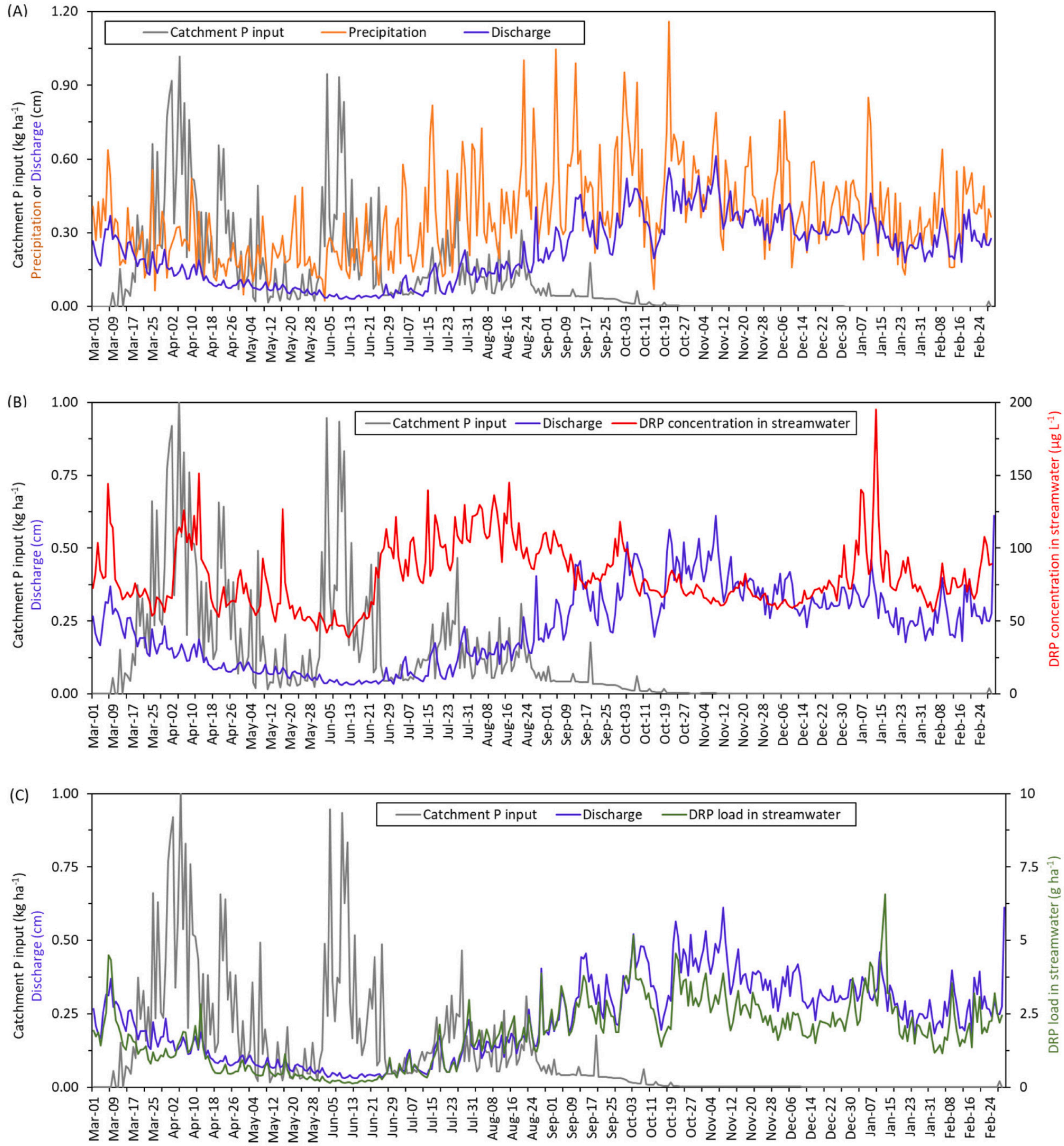

Fig. 3. Trends of daily catchment-level P input, precipitation, discharge, and concentration and load of DRP in streamwater. Each value is an average of 20 years of results for the same calendar date.

ha^{-1}, or 95 % vs. 5 % of the annual P input, respectively (Fig. 4B). Greatly differing from the pattern of P input, 70 % of the DRP load occurred during the manure non-intensive season (Fig. 4B). Monthly DRP load ranged from 8 g ha^{-1} in June to 97 g ha^{-1} in October.

Similarly to the trends of daily values as described above, monthly DRP concentrations in streamwater (on average of 20 monitoring years) did not exhibit a large variability among months (Fig. 4C). The concentration varied from 61 to 109 μg L^{-1}, with an average of 81 μg L^{-1}.

3.3.3. *Trends of annual values*

Over the 20-year study period, annual precipitation and annual discharge in our study catchment ranged from 935 to 1599 mm (mean: 1335 mm) and from 586 to 1160 mm (811 mm), respectively. Both annual precipitation (data not shown) and annual discharge (Fig. 5A) varied widely with year, but their temporal trends were insignificant. In contrast, annual load of DRP in streamwater (range: 0.35–1.46 kg ha^{-1}; mean: 0.65 kg ha^{-1}) significantly increased with time (p = 0.0003; Fig. 5A). Although annual discharge and annual DRP load had varying temporal trends, a correlation analysis showed that annual DRP load significantly increased with increasing discharge ($R^2 = 0.35$, $p = 0.006$; Fig. 5B). It should be noted that, however, under similar amounts of discharge the annual DRP loads in more recent 10 years were frequently greater than those in the first 10 years (Fig. 5B), indicating changes in streamwater DRP concentrations over time. Indeed, flow-weighted

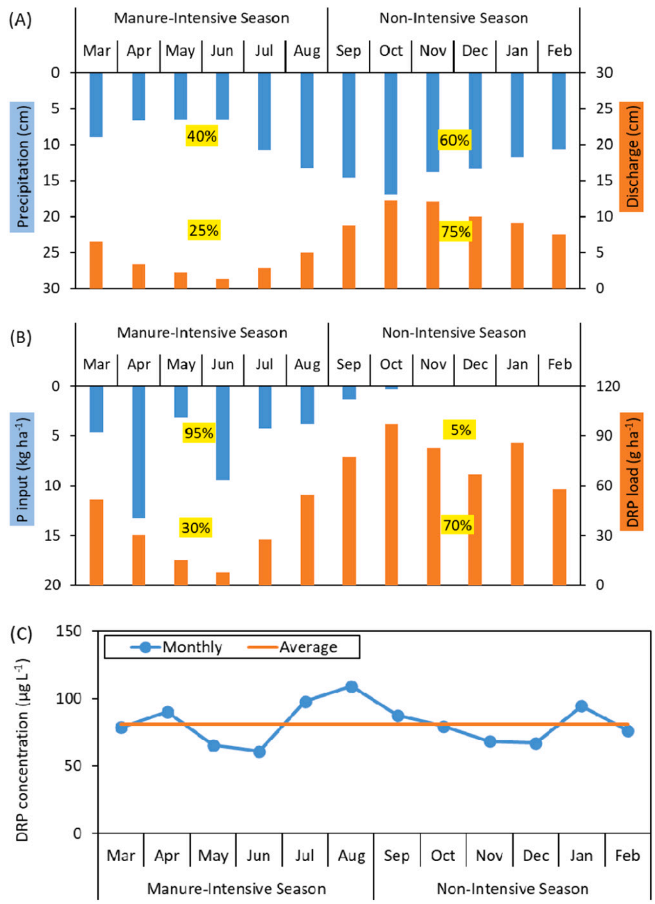

Fig. 4. Trends of monthly and seasonal precipitation, discharge, catchment-level P input, DRP load in streamwater, and flow-weighted mean DRP concentration. Each value is an average of 20 years of results for the same calendar month. The percentages presented in the yellow boxes in (A) and (B) are the approximate contributions of manure-intensive season and non-intensive season to the annual values, respectively, rounding to the nearest 5 %.

annual mean DRP concentration (range: 49–140 μg L^{-1}; mean: 80 μg L^{-1}) trended to increase significantly over the 20-year study period ($p = 0.001$; Fig. 5C). The mean DRP concentration significantly increased with increasingly cumulative soil P balance since start of the monitoring period ($R^2 = 0.55$; $p = 0.0002$; Fig. 5D).

4. Discussion

4.1. The origin of P surplus in agricultural soils

Livestock production is a major concern for water quality in agricultural catchments around the world (Li et al., 2015; Sharpley et al., 2015; Kleinman et al., 2020). In many countries, livestock production has been largely specialized, intensified, and separated from crop production areas to increase production efficiency (Gerber et al., 2005; Neumann et al., 2009; Peyraud et al., 2014; Kleinman et al., 2015; Nesme et al., 2015), in Norway also for prioritizing grain production in regions with suitable climate. In intensive livestock production systems, however, large rates of manure are often frequently applied to a limited land area, resulting in significant buildup of soil nutrients and elevated risks of nutrient losses from land to water (Kleinman et al., 2015). The buildup of soil P in livestock regions happens because crops usually require a higher N:P ratio than that in typical manures, and when manure is applied based on crop need of N, it leads to P surplus accumulating in the soil (Sharpley et al., 1994). In Norway, the production systems of livestock and grain crops were separated gradually starting from the year 1950, under the guidance of a national policy (Melås, 2019). As a result, large quantities of manure P inputs and very high soil P-AL are found in southwestern Norway (Bechmann et al., 2013).

4.2. The importance of P surplus and soil test P in affecting catchment DRP loss

In our study catchment, average annual soil P surplus was estimated to be 8.8 kg P ha^{-1} during the 20-year study period (Fig. 1), i.e., a total of 176 kg ha^{-1} of P surplus over the 20 years. Although the P surplus was relatively low as compared to many other livestock-intensive regions outside Norway (e.g., Li et al., 2015; Svanbäck et al., 2019), it appeared to be high enough to increase flow-weighted annual mean DRP concentrations in streamwater (Fig. 5C). Often, up to 50–60 % of the total P in cattle manure is water soluble (Liu et al., 2018b). After the manure is applied to the soil and at the presence of rainfall or snowmelt, this P fraction can contribute to an immediate source of DRP loss in surface runoff or leaching (Kleinman et al., 2020). When the rainfall or snowmelt is absent, the water-soluble P is adsorbed to the soil and becomes part of the soil labile P pool. The portion of manure P that is unused by crops (i.e., P surplus) accumulates in the soil and builds up the soil labile P pool over time, which constitutes a long-term source of DRP loss (Kleinman et al., 2020). Consequently, annual DRP load also increased over time, despite an insignificant change in discharge (Fig. 5A). Our study region has a long history of intensive livestock production even before the start of catchment monitoring in 1990s (Krogstad, 1987), which explains the very high soil P-AL concentrations in most of the fields measured in 2005 (Fig. 2). It is quantitatively unclear regarding how the P surplus during the monitoring period had interacted with the large P pool established before the start of the monitoring. However, it is probable that the P surplus has helped towards P saturation in the soil. After a dose of manure application, Liu et al. (2012) found greater increases in DRP concentrations, as compared to before the application, in drainage from topsoils with higher degrees of P saturation than from those with lower P saturation. Also, Kristoffersen et al. (2020) showed low P adsorption in high-P soils. Given many soils in our study catchment had high soil test P, any increase in P saturation might accelerate DRP concentrations in drainage.

The long-term average daily and monthly DRP concentrations were relatively stable within the year (Fig. 3B & Fig. 4C), despite 95 % of the P was applied during March–August and only 5 % during September–February (Fig. 4B). The pattern may be due to three reasons. First, there was a relatively small water surplus during March–August, when 40 % of the annual precipitation led to 25 % of the annual discharge (Fig. 4A). Second, plants were active to take up P during March–August. Finally, most of the manure had a high water content, which allowed for good infiltration of manure and sufficient contact between P and soil before episodes with discharge. In Norway, it is mandated by law to avoid manure applications during late autumn and winter when crop uptake of nutrients is minimal (Liu et al., 2018a). The regulation has likely helped prevent P loss during those months with high P loss risks. The stability of DRP concentration in streamwater during the year points to soil P as the main source for DRP runoff compared to incidental losses related to manure spreading. Both Norwegian and other studies have found increasing risks of dissolved P loss with increasing soil test P concentrations (e.g., Kristoffersen et al., 2020; Liu et al., 2021). Therefore, reducing easily releasable P in soil by reduced application is an important measure for reducing transport of DRP to streamwater.

4.3. Need of continued monitoring and research

The long-term monitoring has shown considerable variability in annual mean DRP concentrations (Fig. 5C). Notably, the annual mean DRP concentrations trended to decrease during the period 2016/2017–2020/2021. With the existing data, it is impossible to tell whether it is natural variability or a decrease due to declining P balances that

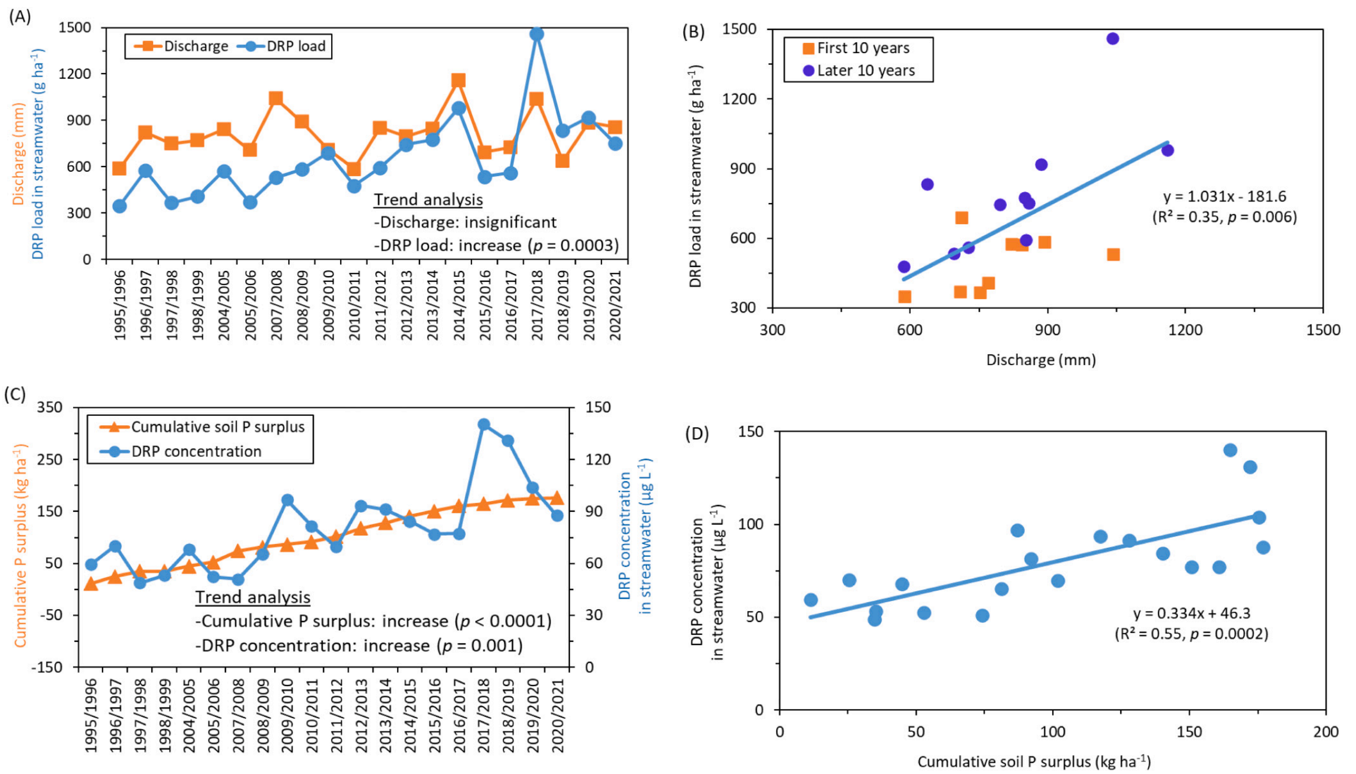

Fig. 5. Long-term trends of DRP in streamwater, and their relationships with influential variables. (A) Temporal trends of annual discharge and annual DRP load; (B) Correlation between annual DRP load and annual discharge ($n = 20$); (C) Temporal trends of cumulative soil P surplus and flow-weighted annual mean DRP concentration; and (D) Correlation between flow-weighted annual mean DRP concentration and cumulative P surplus ($n = 20$).

have happened since 2012/2013 (Fig. 1). In a review of many studies, Meals et al. (2010) found that it might take 5–30 years before soil test P and runoff P had significantly responded to changes in fertilizer/manure management. Therefore, long term monitoring is essential to confidently identify the improvement of water quality.

The P balance data were derived from information given by farmers and were subject to large uncertainty, especially regarding the water content of applied manure and the yield amount, dry matter and concentration of nutrients. However, the average P balance of 8.8 kg ha^{-1} in our study seems reasonable when being compared to other estimated results for farm-gate P balances in Rogaland County (Fig. 1). For example, Volden et al. (2022) estimated an average surplus of 6.4 kg P ha^{-1} on seven farms in 2021, and Halland et al. (2022) estimated an average surplus of 16.4 kg P ha^{-1} year^{-1} on 20 farms during the years of 2018–2020.

The positive correlation between flow-weighted annual mean DRP concentrations in the streamwater and cumulative P surplus in the soil (Fig. 5D) suggests the need to reduce P surplus in livestock-intensive regions to protect water quality. Several strategies have been proposed in literature, besides decreasing livestock density, for reducing P surplus, e.g., improving livestock feeding management (Kleinman et al., 2020), increasing crop P removal (Van der Salm et al., 2009; Vandermoere et al., 2021), transporting manure to non-livestock-intensive areas (e.g., Liu et al., 2016), and refining manure to concentrate P and facilitate transport (Persson and Rueda-Ayala, 2022).

In this study, we focused the analysis on long-term trends and average risks of DRP loss and did not examine event-level manure application effects and processes. Nevertheless, manure applications affect water quality not only through building up soil nutrients over a long term but also through their incidental effects, i.e., the losses shortly after manure applications (Withers et al., 2003). Also, point sources from e.g., leakage of manure storage can affect water quality. The event-level effects of manure applications and possible point sources should be further studied, in combination with detailed field-scale risk assessment in order to best target management strategies.

5. Conclusions

In livestock-intensive regions, P surplus is a big concern for water quality. Our study showed that the buildup of soil test P by P surplus over time plays an important role in affecting water quality. We found significant correlations between soil P balance and soil P-AL at the field level, and between cumulative soil P surplus and flow-weighted annual mean DRP concentration in streamwater at the catchment level. A moderate P surplus of 8.8 kg ha^{-1} year^{-1} was high enough to significantly increase the annual mean concentration and annual load of DRP over the 20-year study period. The results pointed to soil P as the main source for DRP runoff, and, thereby, the need to reduce the easily releasable P content in soil when aiming at reduced transport of DRP to streamwater.

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2023.165470.

Author contributions statement

J.L.: conceptualization, data analyses, results interpretation, and manuscript writing. M.B.: conceptualization, fund acquisition, catchment monitoring, and help with data analyses, results interpretation and manuscript writing. H·O.E.: database management and data compiling, and help with manuscript writing. A.F.Ø.: help with results interpretation and manuscript writing.

Declaration of competing interest

The authors declare no conflict of interest.

Data availability

Data will be made available on request.

Acknowledgement

This study was funded by the Norwegian Agricultural Environmental Monitoring Programme (JOVA), which received financial supports from Ministry of Agriculture and Food. We thank the funds and many people (including cooperative farmers) who helped with all kinds of work in the study catchment over the decades.

References

Andersen, H.E., Blicher-Mathiesen, G., Bechmann, M., Povilaitis, A., Iital, A., Lagzdins, A., Kyllmar, K., 2014. Mitigating diffuse nitrogen losses in the Nordic-Baltic countries. Agric. Ecosyst. Environ. 195, 53–60.

Bakken, A.K., Steinshamn, H., 2022. Grovfôravlinger i Norge: En gjennomgang av datakilder. NIBIO Rapport 8 (91), 38 (In Norwegian).

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

Bechmann, M., 2014b. The effect of phosphorus application and balance on concentrations in streams from agricultural dominated catchments in Norway. Acta Agric Scand. B Soil Plant Sci. 63, S162–S171.

Bechmann, M., Deelstra, J., 2013 (Ed.). Agriculture and environment – long term monitoring in Norway. Akademika publishing, Trondheim, Norway. 392 pp.

Bechmann, M., Greipsland, I., Øgaard, A.F., 2013. Phosphorus use in agriculture. In: Bechmann, M., Deelstra, J. (Eds.), Agriculture and Environment – Long Term Monitoring in Norway. Akademika Publishing, Trondheim, Norway, pp. 69–82.

Bechmann, M., Stenrød, M., Kværnø, S.H., Eggestad, H.O, 2021. Erosjon og tap av næringsstoffer og plantevernmidler fra jordbruksdominerte nedbørfelt - Sammendragsrapport fra Program for jord- og vannovervåking i landbruket (JOVA) for 1992—2019. NIBIO Report 7 (135), 106 pp (In Norwegian).

Bergström, L., Kirchmann, H., Djodjic, F., Kyllmar, K., Ulén, B., et al., 2015. Turnover and losses of phosphorus in Swedish agricultural soils: long-term changes, leaching trends, and mitigation measures. J. Environ. Qual. 44, 512–523.

Bos, J., Williams, M., Smith, D., Armstrong, S., Harmel, D., 2021. Long-term effect of poultry litter application on phosphorus balances and runoff losses. J. Environ. Qual. 50, 639–652.

Cassidy, R., Jordan, P., Bechmann, M., Kronvang, B., Kyllmar, K., Shore, M., 2018. Assessments of composite and discrete sampling approaches for water quality monitoring. Water Resour. Manag. 32, 3103–3118.

Daugstad, K., Kristoffersen, A.Ø., Nesheim, L., 2012. Næringsinnhald i husdyrgjødsel: Analyser av husdyrgjødsel frå storfe, sau, svin og fjørfe 2006–2011. Bioforsk Repart 7 (24), 29 (In Norwegian).

Deelstra, J., Stenrød, M., Bechmann, M., Eggestad, H.O., 2012. Discharge measurement and water sampling. In: Bechmann, M., Deelstra, J. (Eds.), Agriculture and Environment – Long Term Monitoring in Norway. Akademika Publishing, Trondheim, Norway, pp. 83–104.

Egner, H., Riehm, H., Domingo, W.R., 1960. Untersuchungen uber die chemische Bodenanalyse als Grundlage fur die Beurteilung des Nahrstoffzustandes der Boden. II Chemische Extraktionsmethoden zur Phosphor- und Kaliumbestimmung. Kunlinga Landboukshogskolans Annaler 26, 199–215 (In German).

Ekholm, P., Turtola, E., Grönroos, J., Seuri, P., Ylivainio, K., 2005. Phosphorus loss from different farming systems estimated from soil surface phosphorus balance. Agric. Ecosyst. Environ. 110, 266–278.

European Commission, 2000. Directive 2000/60/EC of the European Parliament and of the council of 23 October 2000: establishing a framework for community action in the field of water policy. Off. J. Eur. Union L327, 1–93.

European Economic Community, 1991. Protection of water against pollution by nitrates from agriculture. Off. J. Eur. Union L375, 1–8.

FAO/ISRIC/ISSS, 1998. World reference base for soil resources. World Soil Resour. Rep. 84, 92 pp.

Gerber, P., Chilonda, P., Franceschini, G., Menzi, H., 2005. Geographical determinants and environmental implications of livestock production intensification in Asia. Bioresour. Technol. 96, 263–276.

Halland, A., Bjugan, M., Dombu, S.V., Haukås, T., Bonesmo, H., 2022. Foretaksøkonomiske konsekvenser av forslag til endringer i gjødselregelverk. NIBIO Report 8 (47), 91 (In Norwegian).

Hanrahan, B.R., King, K.W., Williams, M.R., Duncan, E.W., Pease, L.A., et al., 2019. Nutrient balances influence hydrologic losses of nitrogen and phosphorus across agricultural fields in northwestern Ohio. Nutr. Cycl. Agroecosyst. 113, 231–245.

Hauken, M., Kværnø, S., 2013. Agricultural management in the JOVA catchments. In: Bechmann, M., Deelstra, J. (Eds.), Agriculture and Environment – Long Term Monitoring in Norway. Akademika Publishing, Trondheim, Norway, pp. 19–42.

Heje, K.K., 2000. Håndbok for jordbruket 2001. Landbruksforl 109, 256 (pp (In Norwegian).

Johnes, P.J., Heathwaite, L., Spears, B., Brownlie, W., Elser, J.J., Haygarth, P.M., Macintosh, K., Withers, P., 2022. Phosphorus and water quality. In: Brownlie, W.J., Sutton, M.A., Heal, K.V., Reay, D.S., Spears, B.M. (Eds.), Our Phosphorus Future: Towards Global Phosphorus Sustainability. Centre for Ecology and Hydrology, Edinburgh, United Kingdom, pp. 159–217.

Kleinman, P.J.A., Sharpley, A.N., Withers, P.J.A., Bergström, L., Johnson, L.T., Doody, D. G., 2015. Implementing agricultural phosphorus science and management to combat eutrophication. Ambio 44, S297–S310.

Kleinman, P.J.A., Spiegal, S., Liu, J., Holly, M., Church, C., Ramirez-Avila, J., 2020. Managing animal manure to minimize phosphorus losses from land to water. In: Waldrip, H.M., Pagliari, P.H., He, Z. (Eds.), Animal Manure: Production, Characteristics, Environmental Concerns, and Management. American Society of Agronomy, Madison, Wisconsin, United States, pp. 201–228.

Kleinman, P.J.A., Spiegal, S.A., Silviera, M.L., Baker, J.M., Dell, C.J., et al., 2022. Envisioning the manureshed: toward comprehensive integration of modern crop and animal production. J. Environ. Qual. 51, 481–493.

Kolle, S.O., Oguz-Alper, M., 2020. Bruk av gjødselressurser i jordbruket 2018: Metodebeskrivelse og resultater fra en utvalgsbasert undersøkelse. Statistics Norway, Oslo, Norway (124 pp (In Norwegian).

Kristoffersen, A.Ø., Krogstad, T., Øgaard, A.F., 2020. Prediction of available phosphorus in soil: combined use for crop production and water quality protection. J. Environ. Qual. 49, 1575–1584.

Krogstad, T., 1987. Utvikling og vurdering av fosfortilstand i dyrka jord i perioden 1960-85 med hovedvekt på Romerike og Jæren. Norske Jord og Myrselskap 5, 153–163 (In Norwegian).

Li, H., Liu, J., Li, G., Shen, J., Bergström, L., Zhang, F.S., 2015. Past, present, and future use of phosphorus in Chinese agriculture and its influence on phosphorus losses. Ambio 44, S274–S285.

Liu, J., Aronsson, H., Ulén, B., Bergström, L., 2012. Potential phosphorus leaching from sandy topsoils with different fertilizer histories before and after application of pig slurry. Soil Use Manag. 28, 457–467.

Liu, J., Kleinman, P.J.A., Beegle, D.B., Dell, C.J., Veith, T.L., Saporito, L.S., et al., 2016. Subsurface application enhances benefits of manure redistribution. Agric. Environ. Lett. 1, 150003.

Liu, J., Kleinman, P.J.A., Aronsson, H., Flaten, D., McDowell, R.W., Bechmann, M., et al., 2018a. A review of regulations and guidelines related to winter manure application. Ambio 47, 657–670.

Liu, J., Spargo, J.T., Kleinman, P.J.A., Meinen, R., Moore, P.A., Beegle, D.B., 2018b. Water-extractable phosphorus in animal manure and manure compost: quantities, characteristics, and temporal changes. J. Environ. Qual. 47, 471–479.

Liu, J., Macrae, M.L., Elliott, J.A., Baulch, H.M., Wilson, H.F., Kleinman, P.J.A., 2019. Impacts of cover crops and crop residues on phosphorus losses in cold climates: a review. J. Environ. Qual. 48, 850–868.

Liu, J., Elliott, J.A., Wilson, H.F., Macrae, M.L., Baulch, H.M., Lobb, D.A., 2021. Phosphorus runoff from Canadian agricultural land: a cross-region synthesis of edge-of-field results. Agric. Water Manag. 255, 107030.

Meals, D.W., Dressing, S.A., Davenport, T.E., 2010. Lag time in water quality response to best management practices: a review. J. Environ. Qual. 39, 85–96.

Melås, A.M., 2019. Den regionale arbeidsdelingen i landbruket og kanaliseringspolitikken mellom 1990 og 2017. In: Master Thesis. Norwegian University of Science and Technology, Trondheim, Norway, 80 pp (In Norwegian).

Nesme, T., Senthilkumar, K., Mollier, A., Pellerin, S., 2015. Effects of crop and livestock segregation on phosphorus resource use: a systematic, regional analysis. Eur. J. Agron. 71, 88–95.

Neumann, K., Elbersen, B.S., Verburg, P.H., Staritsky, I., Pérez-Soba, M., de Vries, W., Rienks, W.A., 2009. Modelling the spatial distribution of livestock in Europe. Landsc. Ecol. 24, 1207–1222.

NIBIO, 2023. Fosforbehov – korreksjon etter jordanalyser. Norwegian Institute of Bioeconomy Research https://www.nibio.no/tema/jord/gjodslingshandbok/korreksjonstabeller/fosforbehov–korreksjon-etter-jordanalyser?locationfilter=true. Accessed 05.01.2023 (In Norwegian).

Persson, T., Rueda-Ayala, V., 2022. Phosphorus retention and agronomic efficiency of refined manure-based digestate – a review. Front. Sustain. Food Syst. 6, 993043.

Peyraud, J.L., Taboada, M., Delaby, L., 2014. Integrated crop and livestock systems in Western Europe and South America: a review. Eur. J. Agron. 57, 31–42.

Reynolds, C.S., Davies, P.S., 2001. Sources and bioavailability of phosphorus fractions in freshwaters: a British perspective. Biol. Rev. Camb. Philos. Soc. 76 (1), 27–64.

SAS Institute, 2012. SAS User's Guide: Statistics. SAS Inst, Cary, North Carolina, USA.

Schindler, D.W., Carpenter, S.R., Chapra, S.C., Hecky, R.E., Orihel, D.M., 2016. Reducing phosphorus to curb lake eutrophication is a success. Environ. Sci. Technol. 50, 8923–8929.

Sharpley, A.N., Chapra, S.C., Wedepohl, R., Sims, J.T., Daniel, T.C., Reddy, K.R., 1994. Managing agricultural phosphorus for protection of surface waters: issues and options. J. Environ. Qual. 23, 437–451.

Sharpley, A.N., Bergström, L., Aronsson, H., Bechmann, M., Bolster, C.H., et al., 2015. Future agriculture with minimized phosphorus losses to waters: research needs and direction. Ambio 44, S163–S179.

Standard Norway, 2019. Water Quality — Determination of Orthophosphate and Total Phosphorus Contents by Flow Analysis (FIA and CFA) — Part 2: Method by Continuous Flow Analysis (CFA) (ISO 15681-2:2018). Oslo, Norway.

Svanbäck, A., McCrackin, M.L., Swaney, D.P., Linefur, H., Gustafsson, B.G., Howarth, R. W., Humborg, C., 2019. Reducing agricultural nutrient surpluses in a large catchment – links to livestock density. Sci. Total Environ. 648, 1549–1559.

Vagstad, N., 2014. Nutrient management for integrating productivity and environmental concerns – framework of a joint China-Norway research initiative. Acta Agric. Scand. B Soil Plant Sci. 63, S105–S108.

Van der Salm, C., Chardon, W.J., Koopmans, G.F., van Middelkoop, J.C., Ehlert, P.A.I., 2009. Phytoextraction of phosphorus-enriched grassland soils. J. Environ. Qual. 38, 751–761.

Vandermoere, S., Van De Sande, T., Tavernier, G., Lauwers, L., Goovaerts, E., Sleutel, S., De Veve, S., 2021. Soil phosphorus (P) mining in agriculture – impacts on P availability, crop yields and soil organic carbon stocks. Agric. Ecosyst. Environ. 322, 107660.
Volden, H., Hellesø, H., Viste, J., Njærheim, M.G., Lima, S., Pedersen, H., Dahlmo, M., Husveg, N., 2022. Oppdatert faggrunnlag for redusert fosforutslipp fra husdyrproduksjonene i Rogaland – Vurderinger til bruk i arbeidet med gjødselregelverket. Norwegian University of Life Sciences, Ås, Norway, 68 pp (In Norwegian).
Wenng, H., Croghan, D., Bechmann, M., Marttila, H., 2021. Hydrology under change: long-term annual and seasonal changes in small agricultural catchments in Norway. Hydrol. Res. 52 (6), 1542.
Withers, P.J.A., Ulén, B., Stamm, C., Bechmann, M., 2003. Incidental phosphorus losses – are they significant and can they be predicted? J. Plant Nutr. Soil Sci. 166, 459–468.