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How to Achieve a 50% Reduction in Nutrient Losses From Agricultural Catchments Under Different Climate Trajectories?

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Key Points:

- Inorganic nitrogen (IN) loads were forecasted to decrease under future climate change, while no distinct trends in total phosphorus (TP) were observed
- Target reductions in TP loads rely on more widespread implementation of stream mitigation (SM) to tackle secondary pollution sources
- Target reductions in IN loads can be achieved by combining SM with fertilizer reductions and/or cover crops

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract

Under persistent eutrophication of European water bodies and a changing climate, there is an increasing need to evaluate best-management practices for reducing nutrient losses from agricultural catchments. In this study, we set up a daily discharge and water quality model in Hydrological Predictions of the Environment for two agricultural catchments representative for common cropping systems in Europe's humid continental regions to forecast the impacts of future climate trajectories on nutrient loads. The model predicted a slight increase in inorganic nitrogen (IN) and total phosphorus (TP) loads under RCP2.6, likely due to precipitation-driven mobilization. Under RCP4.5 and RCP8.5, the IN loads were forecasted to decrease from 16% to 26% and 21%–50% respectively, most likely due to temperature-driven increases in crop uptake and evapotranspiration. No distinct trends in TP loads were observed. A 50% decrease in nutrient loads, as targeted by the European Green Deal, was backcasted using a combination of management scenarios, including (a) a 20% reduction in mineral fertilizer application, (b) introducing cover crops (CC), and (c) stream mitigation (SM) by introducing floodplains. Target TP load reductions could only be achieved by SM, which likely results from secondary mobilization of sources within agricultural streams during high discharge events. Target IN load reductions were backcasted with a combination of SM, fertilizer reduction, and CC, wherein the required measures depended strongly on the climatic trajectory. Overall, this study successfully demonstrated a modeling approach for evaluating best-management practices under diverging climate change trajectories, tailored to the catchment characteristics and specific nutrient reduction targets.

Plain Language Summary

The European Union has set a target to reduce nutrient losses from agricultural areas by 50% by 2030 to improve the quality of its water bodies. However, we argue that climate change will have a strong impact on nutrient dynamics, implying that the required management actions for improving water quality need to adapt depending on the climate trajectory. In this study, we simulated the future losses of two major nutrients, inorganic nitrogen (IN) and total phosphorus (TP), for two Swedish agricultural streams representative of major crop-growing regions. We also modeled best management practices to reach the targeted 50% reduction in nutrient losses. The model predicted that IN loads will decrease under moderate and severe climate change pathways, but increase under the mild climate change pathway. We found that targeted reductions in TP loads could only be achieved through SM. Targeted reductions in IN loads could be achieved by combining SM with a 20% reduction in mineral fertilizer and/or protecting the soil with cover crops (CC) in winter. This study demonstrated how to apply water quality models for identifying the required management actions to reduce future nutrient losses from agricultural catchments.

1. Introduction

In the agricultural regions of Northern and Western Europe, over 80% of water bodies fail to reach good ecological status according to the Water Framework Directive (Kristensen et al., 2018). Despite continuous efforts to reduce nutrient inputs since the 1990s (Lu & Tian, 2017), the negative impacts from diffuse pollution on aquatic ecosystems are evident, causing eutrophication and hypoxia in inland and coastal waters (Andersen et al., 2017). Decades of over fertilisation have led to the buildup of legacy nutrient stores in agricultural landscapes (Basu et al., 2022; Bouwman et al., 2013). Extensive drainage networks have been installed throughout European farming areas in the form of open ditches, straightened streams, and subsurface tile drains (Schultz et al., 2007). While these have improved crop growing conditions, the increased hydrological connectivity has also exacerbated nutrient and sediment losses from agricultural catchments (Blann et al., 2009; Castellano

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et al., 2019). The combination of high nutrient inputs, legacy stores, and artificial drainage continues to negatively impact water quality and aquatic biodiversity (Andersen et al., 2017; Ulén et al., 2007).

The European Green Deal aims to reduce nutrient losses from agricultural areas with 50% by 2030 through decreasing fertilisation by 20% (European Commission, 2020) and implementation of best management practices to reduce nutrient losses along delivery pathways. Existing management practices include measures to (a) reduce nutrient inputs to fields, (b) reduce erosion and mobilization of nutrients, and (c) intercept and immobilize nutrient and sediment flows. The first group of measures relies on reducing fertilizer application rates, adjusting them to crop needs and better nutrient management (Drinkwater & Snapp, 2007; Quemada et al., 2013). The second group of measures aims to protect the soil surface from erosion and reduce nutrient leaching through implementation of autumn crops, CC, structured liming, and reduced till (Blomquist et al., 2018; Kaye & Quemada, 2017; Kleinman et al., 2009). The third group includes a wide range of SM measures such as bank stabilization, buffer strips, sedimentation ponds, re-meandering, constructed floodplains, and stream ponding (D'Ambrosio et al., 2015; Djodjic et al., 2022; Lammers & Bledsoe, 2017).

Despite the large financial investments and efforts by land and water managers, improvements in water quality are often insignificant (Destouni et al., 2017; Wiering et al., 2020). This can be partly explained by climatic variation and extreme weather (Mellander et al., 2018), and by the widespread prevalence of nutrient legacies in agricultural catchments (Basu et al., 2022; Zia et al., 2022) that override catchment management impacts. Moreover, management practices are often implemented with inadequate sizes, locations, or designs, based on personal preferences of landowners and financial drivers (Djodjic et al., 2022; Roley et al., 2016), which does not reflect the spatial variability in catchment processes that govern nutrient transport and removal (Basu et al., 2023; Hallberg et al., 2022; Walton et al., 2020). There is thus a need for a catchment-specific evaluation of best management practices that are required to achieve set water quality goals and integrate those in a decision support strategies before committing and investing in specific measures (Hogan et al., 2023). However, many uncertainties remain about the interactions between management actions, and the spatial and temporally varied impacts of climate change on nutrient loads (Bol et al., 2018; Zia et al., 2022).

The IPCC has developed multiple Representative Concentration Pathways (RCPs) for the greenhouse gas concentration trajectories, which are used as a basis for climate modeling (Van Vuuren et al., 2011). Downscaling these global climatic models has yielded regional forecasts of temperature and precipitation (Jacob et al., 2020). Process-based water quality models can describe the fluxes of water and nutrients within a catchment based on a simplified mathematical representation of complex hydrological and nutrient processes (La Follette et al., 2021). By using regional forecasts of precipitation and temperature as forcing data for process-based water quality models, the impacts of climate change on future nutrient loads can thus be estimated (Bartosova et al., 2019; Ockenden et al., 2017; Zia et al., 2022). Process models also allow to decouple catchment management outcomes from climatic variability (Grimvall et al., 2014) and thereby provide a pathway for backcasting scenarios to reach the desired future reduction in nutrient losses (Capell et al., 2021; Hankin et al., 2019).

The objectives of this study were to (a) forecast the impacts of climate change on future nutrient loads and (b) backcast the targeted 50% reduction in nutrient loads with best management practices across multiple impact pathways. Using an ensemble of downscaled future climatic predictions, we modeled the impacts of climate change and catchment management practices on nutrient loads in two agricultural catchments representative for common cropping systems in Sweden. This study evaluated RCP 2.6, RCP 4.5, and RCP 8.5, in the near future (2022–2035), mid future (2050–2065) and distant future (2085–2100). The three backcasted management practices were (a) a 20% reduction in mineral fertilizer, (b) introducing CC between growing seasons, and (c) SM by increasing the size of floodplains. The selection of scenarios was based on the European Green Deal, current agronomic practices and stakeholder-supported SM measures in the study sites (Malgeryd et al., 2015; Svensson & Sundin, 2014).

2. Materials and Methods

2.1. Study Catchments

The selected catchments represent the dominant crop-growing regions of Sweden; Hestadbäcken catchment is located in the humid continental zone of central east Sweden (Beck et al., 2018), and Tullstorpsån catchment is located in the coastal zone of south Sweden. Both study catchments are agriculturally dominated, but differ in

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Table 1
Overview of Current Catchment Characteristics With μ as Mean and σ Standard Deviation

Catchment	Hestadbäcken	Tullstorpsån
Catchment area (km2)	7.6	62.1
Elevation range (m)	44–85	3–101
Dominant land use types	Autumn crops (54.7%), Forest (25.5%), Pasture (16.9%)	Autumn crop (57.1%), Pasture (15.2%), Root crops (8.3%), Forest (7.6%), Spring crops (7.1%)
Floodplain and wetland area	2500 m^2 (0.03%)	507,311 m^2 (0.82%)
Dominant soil classes	Moraine (29.2%), Silty clay (23.7%), Clay loam (19.7%), Clay (8.7%)	Loam (41.8%), Sandy loam (29.6%), Clay loam (6.4%), Moraine (6.0%)
$\mu \pm \sigma$ yearly precipitation (mm)	580 ± 131	790 ± 115
$\mu \pm \sigma$ of >15 mm precipitation days $year^{-1}$	6.5 ± 3.6	7.5 ± 3.3
$\mu \pm \sigma$ of >30 mm precipitation days $year^{-1}$	0.9 ± 0.8	1.0 ± 1.3
Temperature (μ and range in °C)	7.9; −12.8 to 23.3	8.7; −7.9 to 23.1

size, cropping regimes, total precipitation, and soil type (Table 1; Figure 1 and Figure S1 in Supporting Information S1), resulting in different nutrient dynamics and water quality risks. Tullstorpsån is larger overall, but also has a higher percentage of cropland with significant coverage of root crops (8.3%) and spring crops (7.6%) besides its dominant autumn crops (57.1%). The soils are mostly loamy. Hestadbäcken is smaller and is dominated by autumn crops (54.7%) cultivated on clay soils. It also has larger areas of forest (25.5%) and pasture (16.9%), which are mostly developed on the moraine soils. Tullstorpsån is on average wetter and warmer compared to Hestadbäcken, which has less precipitation and a larger temperature range. Hestadbäcken and the upstream gauge of Tullstorpån are hydrologically more flashy (Hallberg et al., 2022).

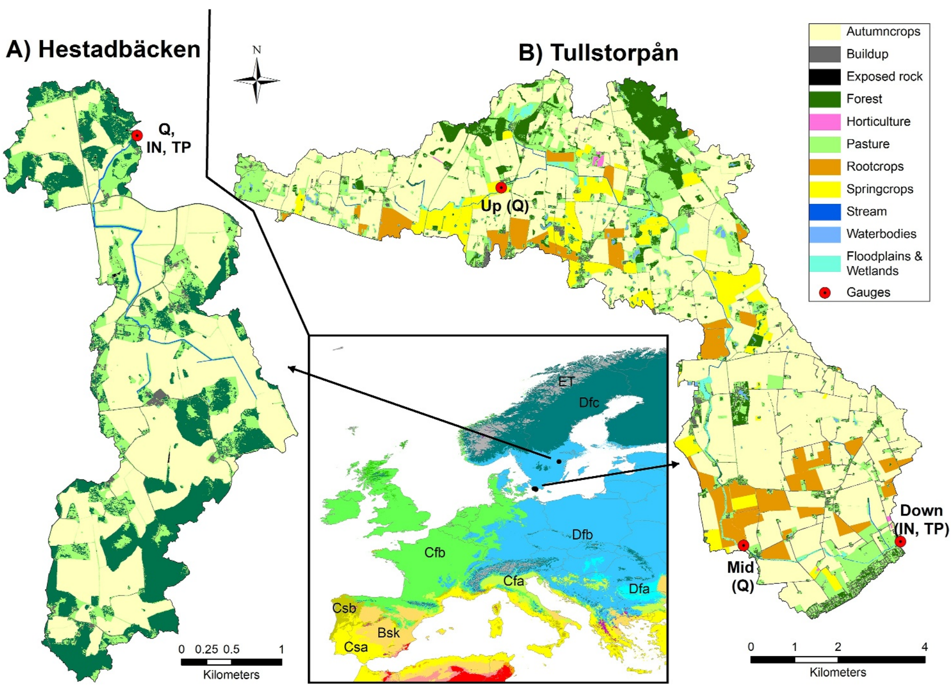

Figure 1. Location and climatic context of the study catchments within Europe under baseline conditions (Beck et al., 2018). Detailed land cover maps of (a) Hestadbäcken and (b) Tullstorpsån catchments, with location of the stream gauges used for calibration and validation.

A watershed assessment was performed for both study catchments with the hydrology toolset in ArcMap 10.8.1 (ESRI, 2020) and based on the 2 m-resolution Grid2+ digital terrain model (Lantmäteriet, 2019). Land cover was determined by performing a maximum likelihood classification on a 0.25 m-resolution RGBI orthophoto from 2021 (Lantmäteriet, 2021) following the methodology of Wynants et al. (2018) to the following land use classes: forest, pasture, cropland, build-up areas, open water, stream, floodplains & wetlands, and rock outcrops. For each field, the cropland class was subdivided into autumn crops, spring crops and root crops based on 2021 crop rotation information (Jordbruksverket, 2022). Soil texture classes were derived from the 50 m-resolution Digital Soil Map of Sweden for arable land (Piikki & Söderström, 2019), the Quaternary Deposits map from the Geological Survey of Sweden (Sveriges Geologiska Undersökning, 2014), and from the land cover classification for build-up areas, rock outcrops, water and stream classes. Precipitation and temperature data were obtained from the meteorological stations of the Swedish Meteorological and Hydrological Institute (2023b) and from the PTHBV grid (Berg et al., 2016; Swedish Meteorological and Hydrological Institute, 2023a).

2.2. Measurements of Discharge and Nutrient Concentrations

Hestadbäcken was gauged continuously for discharge on sub-hourly time steps using a small basin with V-notch, which values were averaged to daily timesteps. Tullstorpsån was monitored intermittently at three different gauge locations (upstream, midstream, and downstream; Figure 1) for either water stage or water quality. Water stage sensors were placed upstream and midstream in May 2020, and manual discharge measurements were performed 8 times over the course of two years. Continuously recorded water depths (S) were converted to stream discharge (Q) using stage-discharge relationships. In the upstream, we used an exponential relationship (Equation 1; $R^2 = 0.831$), while in the midstream we used a split relationship, where an exponential function was followed by a linear (Equation 2). The better fit of a split curve midstream reflects the two-stage design of the stream.

$$Q_{\text{up}} = 0.6863\ x\ S_{\text{up}}^{2.813} \tag{1}$$

$$Q_{\text{mid}} = f(S_{\text{mid}}) = \begin{cases} 30.211\ x\ S_{\text{mid}}^{4.3839} & S_{\text{mid}} < 0.384 \\ 3.208\ x\ S_{\text{mid}} - 0.8418 & S_{\text{mid}} \geq 0.384 \end{cases} \tag{2}$$

Water from Hestadbäcken and Tullstorpån downstream was also continuously sampled in cooled containers, wherein the sampling volume was proportional to the streamflow. The composite samples were collected fortnightly and analyzed for total phosphorus (TP), soluble reactive phosphorus, particulate phosphorus (PP), inorganic nitrogen (IN), total nitrogen, and suspended sediment (Kyllmar et al., 2014). This method was chosen because it captures the entire spectrum of flow events and therefore gives a better representation of the nutrient loads compared to grab sampling (Facchi et al., 2007). Flow-proportional nutrient concentrations were converted to daily nutrient loads using the daily discharge values.

2.3. Model Set-Up, Calibration and Validation

The catchment models were set-up in Hydrological Predictions of the Environment (HYPE), which is a semi-distributed and open source hydrological and nutrient transport modeling framework. Hydrological Predictions of the Environment has been designed for Swedish and European environments and has demonstrated ability to model water quantity and water quality dynamics in these systems (Lindström et al., 2010; Strömqvist et al., 2012). Moreover, HYPE has routines that allow the backcasting of desired reduction in nutrients under future climate forecasts (Bartosova et al., 2019; Capell et al., 2021). Detailed information on the set-up procedures, model assumptions, and model characteristics can be found in Text S1 in Supporting Information S1. For more information on the representation, parameterization, and sensitivity of hydrological and nutrient processes, we refer to Lindström et al. (2010), and Santos et al. (2022).

The catchment models were built on combinations of Soil type and Land use classes (SLCs), with 27 SLCs in Hestadbäcken, and 60 SLCs in Tullstorpsån. Tullstorpsån had one model setup, and was further divided in three sub-catchments based on locations of stream sampling points. The models were forced with daily average precipitation and temperature data. For each SLC, the soil system was classified with up to three soil layers with defined depths. Specific crop types and tile drain depths were assigned to the agricultural SLCs. Each crop type was assigned planting, harvesting, and plowing dates, as well as the amounts and dates of application of mineral fertilizer and manure, based on agricultural monitoring programs at nearby catchments (Kyllmar et al., 2014). Besides fertilisation, nutrients enter the catchment through atmospheric deposition and rural sewage, and leave through crop uptake, denitrification, and with stream water discharges. Potential crop nutrient uptake is based on a logistic growth equation throughout the growing season. The actual nutrient uptake is limited by the available nutrients and a temperature function. Starting pools of nutrients in the different soil layers, as active compounds, in organic form, and bound to soil particles were specified for different SLCs. The values were based on regional trends from the Swedish national soil databases comprising thousands of soil samples (Strömqvist et al., 2012). Floodplains were grouped in one SLC with a defined fraction of the catchment runoff, wherein nutrient removal was modeled using the HYPE's "wetland" subroutine (Arheimer & Wittgren, 2002; Tonderski et al., 2005). The "wetland" subroutine includes water retention coefficients, macrophyte nutrient uptake coefficients and production depths, outflow thresholds, sedimentation rates of particles, and return of nutrients to the soil by degrading macrophytes. Streams were represented as dynamic pools of sediment and PP, wherein sedimentation, erosion, and resuspension can (im)mobilize sediment and PP (Bartosova et al., 2021).

The simulation results were evaluated by comparison with observations of daily discharge and nutrient loads using a set of goodness of fit (GOF) measures (Moriasi et al., 2015), that is, Nash–Sutcliffe efficiency (NSE; Nash & Sutcliffe, 1970), Kling-Gupta efficiency (KGE; Gupta et al., 2009), and relative bias (RE; percentage of difference). The model was optimized using the calibration methodology of Hundecha et al. (2020), wherein we combined manual and automatic calibration based on the Differential Evolution Markov Chain algorithm of Braak (2006). For calibration of hydrology in Tullstorpån, the GOF values of upstream and midstream gauges were averaged before optimisation. Calibration of water quality in Tullstorpån relied on the downstream gauge alone. The calibration was carried out following a stepwise approach by identifying key parameter groups and calibrating these together within possible ranges, while keeping other parameters fixed to reduce potential equifinality (Strömqvist et al., 2012). Because nutrient transport is largely governed by hydrology, the model was initially calibrated for discharge, and subsequently calibrated for the nutrient loads. We calibrated against nutrient loads, and not concentrations, because of the flow-proportional sampling approach and the focus on reducing downstream loads. A smaller subsample was used to temporally validate the model setups for uncalibrated periods, prioritizing the available data for calibration to yield more robust models (Shen et al., 2022). For Tullstorpån, the multi-gauge approach also allowed for a reciprocal spatial validation of the model (Krysanova et al., 2018). Hestadbäcken stream was remediated in 2014 and it was therefore decided to select data from 2016 to current, yielding a 5 years calibration period (January 2016–December 2020), and a 2.5 years validation period (January 2021–June 2023) for both discharge and nutrient loads. For Tullstorpån, the more recent installment of stage sensors yielded a 2 years calibration period (May 2020–June 2022) for discharge, and 1 year validation period (June 2022–June 2023). The nutrient load calibration in Tullstorpån utilized 6.5 years of data (January 2016–June 2022), while the validation period was 2 years (January 2014–December 2015). The model always had a warm-up time of 10 years before calibration, validation, and simulations. The model fit was evaluated using the simplified model evaluation thresholds of Moriasi et al. (2015) and the evaluation guidance for KGE as in Knoben et al. (2019). A simple sensitivity analysis was performed on wetland parameters by measuring the NSE variation during Monte Carlo simulations with random perturbations of parameters between set intervals (Santos et al., 2022).

2.4. Future Climate Trajectories

We used an ensemble of three general circulation models (GCM) from the Coupled Model Intercomparison Project Phase 5 (CMIP5): the "Met Office Hadley Center ESM, HadGEM2-ES" model (Jones et al., 2011), the "Max Planck Institute ESM-LR" (Popke et al., 2013), and the "ICHEC-EC-EARTH" (Hazeleger et al., 2010). These GCMs simulate RCPs 2.6 (stringent reduction), 4.5 (moderate), and 8.5 (business as usual) (Collins et al., 2013). As described in Jacob et al. (2014), the GCMs have been downscaled to a 5 km grid over the northern European regions using the "KNMI regional atmospheric climate model (RACMO) version 2" (van Meijgaard et al., 2008) and the "SMHI Rossby Center regional climate model" (SMHI-RCA4) (Strandberg et al., 2015). An overview of the downscaled climate models can be found in Table S1 in Supporting Information S1. Outcomes from these downscaled climate models were statistically scaled against a reference temperature and precipitation data -set using the distribution-based scaling algorithms of W. Yang et al. (2010). The resulting daily temperature and precipitation data were used to model changes in nutrient loads under future climatic conditions. All data analyses were performed in R Core Team (2022). Annual yearly average precipitation, temperature, and amount

Table 2
Overview of Catchment Management Scenarios

Scenario name	Description
Current	In Hestadbäcken, ca. 2500 m^2 of floodplain & wetland (0.03% of catchment). In Tullstorpsån, ca. 0.51 km^2 of floodplain &wetland (0.82% of catchment)
Baseline	In Hestadbäcken, 200 m^2 (0.003% of catchment) of floodplain & wetland. In Tullstorpspån, 0.08 km^2 (0.13% of catchment) of floodplain & wetland
20% Fert (Fert)	20% reduction in mineral fertilizer application
Cover crops (CC)	All spring crops and root crops get a cover crop in between growing seasons
Stream mitigation (SM)	In Hestadbäcken, floodplains & wetlands increase in size to ca. 61,100 m^2 (0.80% of the catchment) with a barrier of 30 cm between stream. In Tullstorpån, floodplains & wetlands increase to 0.62 km^2 (1.0% of catchment) with a barrier of 30 cm between stream
Fert + CC	Combination of scenarios 20% Fert and Cover Crops
SM + Fert	Combination of 20% Fert and Stream Mitigation
SM + Fert + CC	Combination of 20% Fert, Cover Crops, and Stream Mitigation

of high precipitation days (>15 mm per day & >30 mm per day) for the period 2000–2022, 2022–2035, 2050–2065, and 2085–2100 were estimated using the *aggregate* function. We subsequently performed *t-tests* in R to (a) compare the model predictions with empirical measurements for 2000–2022, (b) evaluate the difference between the climate models and (c) test if the projected changes in precipitation and temperature were significant. Agricultural management and growing season were assumed to remain the same in the different climate trajectories. Model calculations of daily nutrient loads were summed to total yearly nutrient loads. For each period and RCP, basic statistics (mean, median, standard deviation, and interquartile range) were calculated for both individual models and the ensemble of models. T-tests were performed between the nutrient load predictions of different models, RCPs, and periods to evaluate the differences. The percentage of change between the periods was calculated to evaluate trends in nutrient loads. To provide indication of the influence on changing stream discharges on the changing loads, we calculated the best fitting nutrient load-discharge relationships for the original catchment scenario and all of the RCPs and periods combined.

2.5. Catchment Management Scenarios

An overview of the catchment management scenarios can be found in Table 2 and visualized in Figure 2. These included a 20% reduction in mineral fertilizer, CC, and SM Since Hestadbäcken catchment is already dominated by autumn crops, the CC were only modeled in Tullstorpsån. Stream mitigation was modeled by increasing the size of floodplains and by adding a 30 cm bar between the floodplain and mainstream to retain inundation water over longer periods (Figure 2). The baseline area of floodplains before recent SM activities was obtained from historical aerial photographs. Percentage reductions of all management practices were calculated against the baseline situation before any management practices were implemented.

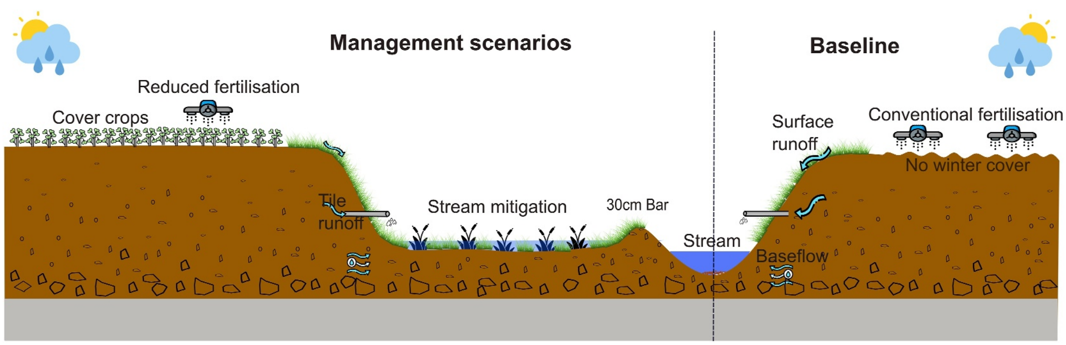

Figure 2. Schematic representation of the flow pathways and modeled management scenarios compared to the baseline.

Table 3
Goodness-Of-Fit Values of the Calibrated Model and Validation Period Showing Nash-Sutcliffe Efficiency, Kling-Gupta Efficiency, and Relative Error (RE; %) Between Simulated and Empirical Discharges and Nutrient Loads

			Discharge			Inorganic nitrogen			Total phosphorus		
		Period	NSE	KGE	RE	NSE	KGE	RE	NSE	KGE	RE
Hestad-bäcken	C	2016–2020	0.75	0.81	14.9	0.61	0.69	−23.4	0.86	0.69	−25.9
	V	2021–2023	0.77	0.82	−5.6	0.74	0.79	1.4	0.78	0.73	2.1
Tullstorpsån: upstream	C	2020–2022	0.75	0.75	22.2		/			/	
	V	2022–2023	0.93	0.89	10.1		/			/	
Tullstorpsån: midstream	C	2020–2022	0.79	0.72	−24.3		/			/	
	V	2022–2023	0.94	0.88	−5.1		/			/	
Tullstorpsån: downstream	C	2016–2022		/		0.80	0.80	−16.4	0.69	0.84	−2.0
	V	2014–2015		/		0.83	0.92	−15.1	0.60	0.66	−5.7

Note. C are the calibrated years, while V are the validation years.

3. Results

3.1. Evaluation of Water Quality Model and Climate Forecasts

3.1.1. Model Outcomes for Discharge and Nutrient Load Calibration and Validation

The fit of discharge and TP in Hestadbäcken can be described as "good" to "very good" for both the calibration (2016–2020) and validation (2021–2022) periods (Table 3) with a slight but consistent overestimation of discharge and underestimation of TP. For IN, the Hestadbäcken fit was "satisfactory" to "good" in the calibration period, and "very good" in the validation period. The bias for IN in Hestadbäcken was more variable, with alternating periods of underestimation and overestimation. The modeled discharge and TP, and to a lesser extent IN, were most sensitive to the SM parameters governing water retention through rating curve and height of the outflow bar (Table S2 in Supporting Information S1). The TP loads were also sensitive to variations in sedimentation velocity.

In Tullstorpsån, the fit of discharge in the calibration period (2020–2022) was "good" to "very good" for both the upstream and midstream gauges, and "very good" both during the validation (2022–2023). Flow was overestimated upstream and underestimated midstream. The main bias in the upstream discharge occurred during the falling limbs of high-flow events, wherein modeled discharge receded slower than observed discharges. At midstream, the main bias originated from missing smaller peaks in between larger events. The IN fit for downstream Tullstorpsån was "very good" for both the calibration and validation periods, with no observed systematic bias. The fit for TP is "very good" in the calibration period and "good" in the validation period, without any systematic bias. Daily flow and load comparison plots of the calibration and validation periods for both catchments can be found in Figures S4–S9 of Supporting Information S1.

3.1.2. Comparison of Nutrient Load Forecasts Under Different Climate Models

In the Hestadbäcken catchment, the downscaled climate models yielded similar predictions in average IN loads, wherein the only differences between load quantifications of different climate models were found under RCP8.5 (Table S3 in Supporting Information S1). Significant differences in TP load estimations were found in 5 of the 27 model comparisons, and near-significant differences in one (Table S4 in Supporting Information S1). Significant differences in yearly discharge were found in 6 of the 27 comparison, and near-significant difference in 1 (Table S5 in Supporting Information S1). The most notable difference are the overall higher estimated discharges and TP loads, as well as their variability, for the KNMI climate model. In the Tullstorpsån catchment, there were slightly more divergences between IN load predictions of different climate models (Table S6 in Supporting Information S1), which were caused by the higher load predictions of the MPI model. Significant differences in TP load estimations were found in 3 of the 27 model comparisons, and near-significant differences in another 3 (Table S7 in Supporting Information S1), while for discharge 8 of the 27 model comparisons were significantly different (Table S8 in Supporting Information S1). Like in Hestadbäcken, the KNMI outcomes for discharge and TP were higher. A detailed comparison of the empirical and predicted precipitation and temperatures between different downscaled models is given in Text S2 in Supporting Information S1.

3.2. Modeled Nutrient Loads Under Different Climatic Trajectories

3.2.1. Trends in Projected Precipitation and Temperature, and Forecasted Discharge

Average annual precipitation and high precipitation days were predicted to increase significantly in both Hestadbäcken and Tullstorpsån under RCP8.5 (Figure S2 in Supporting Information S1). The increase was most pronounced in the winter months. Under RCP2.6, the climate ensemble predicted a distinct increase in high precipitation days in 2050–2065, but no significant increase in average precipitation for both catchments in de mid and distant future. Under RCP4.5, the amount of average yearly precipitation and high precipitation days increased but not significantly. The predicted changes in temperature were more uniform between both sites (Figure S3 in Supporting Information S1). Under RCP4.5 and RCP8.5, significant increases of respectively 0.7°C and 1.5°C were predicted per period.

The predicted changes in precipitation and temperature resulted in differences in forecasted discharge for both rivers (Figure 3; Tables S9 and S10 in Supporting Information S1). Under RCP2.6, a slight increase (9%) in discharge was forecasted for Hestadbäcken for the mid and distant future, while no change was predicted in Tullstorpån. Under RCP4.5, a 5%–10% decrease was forecasted for Hestadbäcken in the future, while Tullstorpån was predicted to decrease with 5% and then move back up to the near future discharges. Under RCP8.5, the discharge in Hestadbäcken was forecasted to increase with 3% in the mid future, followed by a 6% decrease in the distant future, while in Tullstorpån, the discharge was forecasted to remain stable in the mid future and decrease with 8% in the distant future.

3.2.2. Trends in Forecasted Inorganic Nitrogen Loads

For Hestadbäcken, the ensemble model forecasted a 5% and 9% increase in IN loads under RCP2.6 for the 2050–2065 and 2085–2100 periods respectively (Figure 3; Table S11 in Supporting Information S1). Decreases of respectively 8% and 26% in IN loads were found under RCP4.5 and decreases of respectively 23% and 50% under RCP8.5. The only trend divergence between the different climate models was observed for the ICHEC model under RCP2.6, where the loads slightly decreased in 2050–2065, and for the KNMI model under RCP4.5, where the loads remained stable in 2085–2100. The observed trends in IN loads can only be partly explained by discharge, where a linear IN load-discharge relationship (R^2 of 0.56) was found for the current management scenario under all climate trajectories and periods. Crop IN uptake was forecasted to increase, ranging from 1% to 11%, in the mid and distant future under RCP4.5 and RCP8.5, while denitrification was predicted to decrease, ranging from +2 to −17% (Table S21 in Supporting Information S1).

In Tullstorpsån, the ensemble model predicted no significant change under RCP2.6 (Figure 3; Table S12 in Supporting Information S1), a decrease of 11% and 16% in the 2050–2065 and 2085–2100 periods respectively under RCP4.5, and a decrease of 4% and 28% under RCP8.5. However, there were some trend divergences between the models, wherein the MPI model predicted higher IN loads, and the ICHEC model predicted stronger decreases in IN loads compared to the other models. Likewise, the observed trends in IN loads can be partly explained by discharge, where a linear IN load-discharge relationship (R^2 of 0.52) was found. Crop IN uptake was forecasted to increase slightly, ranging from 0% to 5%, in the mid and distant future under RCP4.5 and RCP8.5, while denitrification was predicted to decrease slightly, ranging from −1% to −7% (Table S22 in Supporting Information S1).

3.2.3. Trends in Forecasted Total Phosphorus Loads

In Hestadbäcken, TP loads were forecasted to increase with 13% under RCP2.6 in the 2050–2065 period, and with 14% in 2085–2100 period, compared to the 2022–2035 period (Figure 3; Table S13 in Supporting Information S1). These predictions corresponds to trends in discharge, which is due to the strong linear TP load-discharge relationship ($R^2 = 0.93$) for all climate trajectories and periods. Under RCP4.5 and RCP8.5, the ensemble climate model did not forecast any significant changes in the mid nor distant future. Under RCP2.6, all models show a similar increasing trend (5%–24%). Under RCP4.5, the KNMI showed a distinct increase in 2085–2100, while the MPI showed a distinct decrease in 2085–2100, and the ICHEC remained stable. Under RCP8.5, the ICHEC and

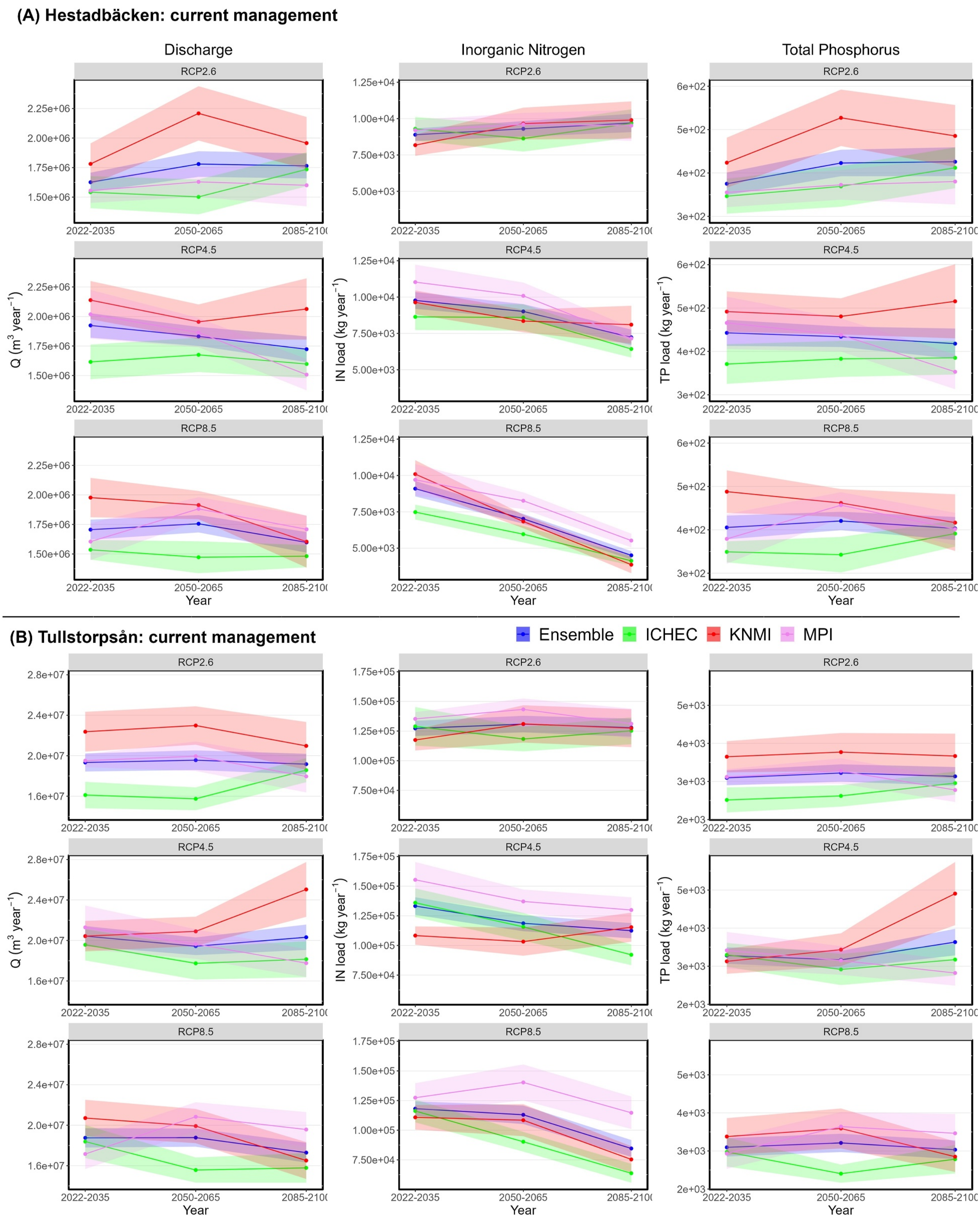

Figure 3. Changes in discharge and nutrient loads for the (a) Hestadbäcken and (b) Tullstorpsån catchments under different climate models and Representative Concentration Pathways. Results are for baseline management practices only. Point and lines represent the mean yearly loads for the period, while the ribbons represent one standard deviation.

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MPI models showed an increase, however with different periods of increase, while the KNMI showed a decreasing trend.

In Tullstorpsån, the ensemble model predicted no significant changes in TP loads under RCP2.6 (Figure 3; Table S14 in Supporting Information S1). Under RCP4.5, the ensemble approach predicted an insignificant 3% decrease in the mid future followed by a distinct 11% increase in the distant future. This was mostly forced by the KNMI model, which predicted a 9.8% increased followed by a very strong 57% increase, while the other two models predicted slightly decreasing TP loads. Under RCP8.5, the ensemble model predicted no significant changes in TP loads between the near and distant future. Overall, the ensemble approach in Tullstorpsån did not forecast any distinct changes in TP loads over time under the different RCPs. However, there was a strong divergence between the outcomes using different climate models, where the KNMI model predicted significant increases under RCP4.5 and RCP8.5. The ICHEC model predicted increased loads under RCP2.6 and decreased loads under RCP4.5 and RCP8.5. The MPI model predicted an increase followed by decrease under RCP2.6, decreasing loads under RCP4.5, and increasing loads under RCP8.5. Overall, predicted trends in TP loads also corresponded with discharge due to the strong power TP load-discharge relationship ($R^2 = 0.849$).

3.3. Backcasted Nutrient Loads Under Catchment Management Scenarios and Climate Change

3.3.1. Hestadbäcken

The SM already in place (0.03% of catchment floodplains and wetlands) only had a minor effect on nutrient load reductions in the catchment compared to the baseline scenario (Figure 4). The changes in average annual discharge ranged between −11.0% and 8.7% (Table S15 in Supporting Information S1). The changes in average loads ranged between −50.7% and 8.3% for IN (Table S16 in Supporting Information S1) and between −7.5% and 8.3% for TP (Table S17 in Supporting Information S1) across the RCPs and periods. Moreover, small increases in area for SM did not have a strong effect on load reductions (Figure S10 in Supporting Information S1). A sole reduction of 20% in mineral fertilizer input had a strong effect on IN loads, with significant reductions ranging between 27.0% and 68.9%. However, a reduction in mineral P inputs had no effect on the predicted TP loads (from −7.6% to 10.8%), which is not significantly different from loads with current P fertilisation rates. The SM scenario (floodplains increase from 0.03% to 0.8%) was predicted to decrease yearly discharge between 34.1% and 46.4%, and IN loads between 39.0% and 74.4%. The SM scenario was found to be the only effective measure for reducing TP loads and yielded reductions ranging between 41.5% and 51.7%. Combining SM with a 20% reduction in fertilisation led to the highest nutrient IN load reductions, ranging between 56.6% and 83.2%. Catchment best management practices were also shown to decrease nutrient loads during wet years and led to a lower range in nutrient loads (Figure 4).

3.3.2. Tullstorpsån

The SM already in place (0.82% of catchment are floodplains and wetlands) was shown to have a significant impact on discharge and nutrient loads compared to the baseline situation before SM (0.13% of the catchment). Discharge was reduced ranging between 4.1% and 12.7%, while load reductions ranged between 6.0% and 28.1% for IN, and between 8.0% and 19.9% for TP (Figure 4; Tables S18–S20 in Supporting Information S1). A sole reduction of 20% mineral fertilizer reduced IN loads between 33.5% and 56.1%, but only an additional 1% in TP load reduction. The SM scenario (increase from 0.8% to 1.0%) resulted in discharge reductions between 27.1% and 34.0%, IN load reductions between 28.9% and 50.9%, and TP load reductions between 30.4% and 41.4%. The inclusion of CC reduced IN loads with 11.5%–39.8%, but had no effect on discharge or TP loads. Combining a 20% decrease of mineral fertilizer and CC resulted in decreases of IN loads ranging between 36.2% and 59.0%, and 10.4%–21.4% decreases in TP loads. Combining a 20% decrease of mineral fertilizer and SM yielded IN load decreases ranging between 49.5% and 66.8%, and TP load decreases ranging between 32.0% and 42.4%. A combination of a 20% fertilizer decrease, SM, and CC, yielded the highest IN load reductions between 51.6% and 68.9%, and TP load reductions between 32.2% and 42.5%.

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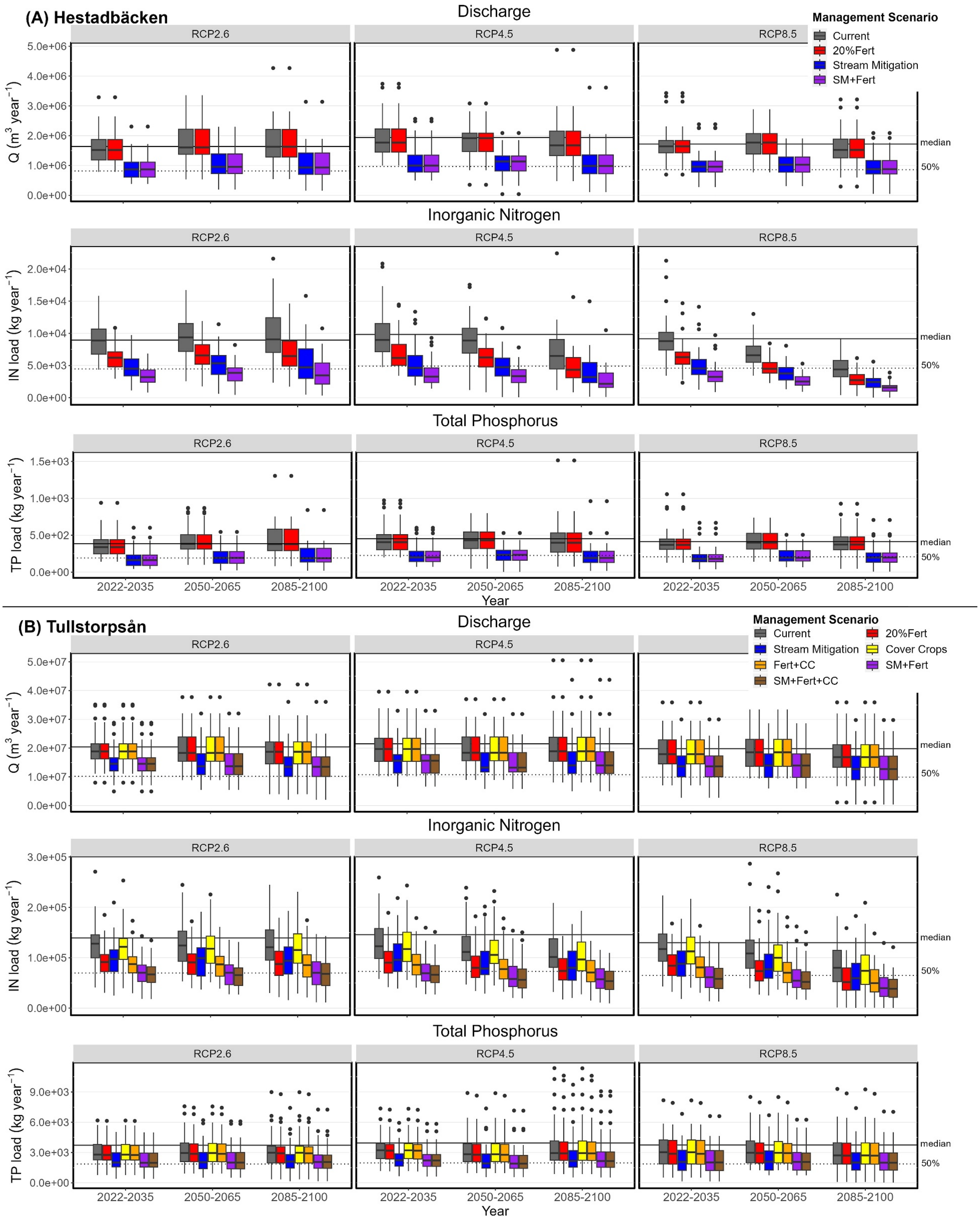

Figure 4. Effects of management scenarios on discharge, IN and total phosphorus loads under different Representative Concentration Pathways. The boxplots depict the outcomes from the ensemble climate model, where median values are shown by the central line, interquartile range by boxes, the range by whiskers, and the outliers by points. The black and dotted lines on the plots show the respective median value and 50% reduction to the baseline management scenario in the 2022–2035 period.

4. Discussion

4.1. Impact of Future Climate on Nutrient Loads in Agricultural Catchments

The response of nutrient loads to climate change differed between IN and TP and between both catchments. TP loads were strongly correlated with changes in discharge, while IN loads were also influenced by other factors. In Hestadbäcken, the predicted trends in IN loads showed a strong similarity between the different climate models, while in Tullstorpån the forecasted trends were slightly more divergent. In most cases, the differences in predicted IN loads between the different climate models were smaller than between RCPs and periods, confirming that our approach is robust for evaluating the impacts of future climate change on IN loads. Inorganic nitrogen loads were forecasted to slightly increase under RCP2.6, decrease under RCP4.5 from the distant future, and decrease under RCP8.5 from the mid future (Figure 3). These diverging trends indicated dominance of increased soil N mineralization and precipitation driven IN mobilization under RCP2.6, versus increased crop uptake (Tables S21–S22 in Supporting Information S1) and evapotranspiration under RCP4.5 and RCP8.5. This indicates that future IN loads depend on the interplay between the effects of increased temperature, crop uptake, denitrification and evapotranspiration leading to reduced loads versus the effects of increased precipitation and IN mobilization leading to increased loads. This pattern is supported by empirical findings of increased crop yield and nutrient uptake following increasing temperatures (Wang et al., 2018), and highlights the complex feedbacks between climate change and water quality (Whitehead et al., 2009). Although increased temperatures in these two catchments are capable of increasing future crop nutrient uptake, these dynamics will also be controlled by hydrological responses to a changing precipitation distribution, wherein water availability during the growing season can be a limiting factor (Grusson et al., 2021; Peterson et al., 2001).

No distinct trends in TP loads could be observed under a changing climate due to the differences between the climate model forecasts. The variability in TP loads between the years was predicted to increase in the future (Figure 4) due to large projected differences in yearly precipitation and discharge. For Hestadbäcken under RCP2.6, the model forecasted that increased precipitation (intensity) will on average lead to slightly higher TP loads driven by increased discharges. Vice versa, under RCP4.5 and RCP8.5 this effect was ofsetted by the projected increases in temperature and associated evapotranspiration that reduced discharge and particle mobilisation. However, the divergences in the predicted TP trends between the different climate models highlighted that the uncertainty of these TP load outcomes remained large in both catchments. Moreover, these trends are based on periodically averaged yearly aggregations of nutrient loads, which could hide seasonal shifts in nutrient dynamics at high flows during storm events. Furthermore, our focus on nutrient loads is not always informative about local water quality dynamics because it is not sensitive to high nutrient concentrations during low flows.

Overall, the complex responses of IN and TP loads to different RCPs and forecasted periods demonstrated the challenges of predicting nutrient losses in agricultural catchments under a changing climate. The predicted increase in nutrient loads in the near future will require immediate management actions to be undertaken to achieve environmental goals for example, EU Green Deal targets. Best management practices will need to focus on buffering high flow events with large mobilisation of nutrients. Water quality management in the mid to distant future will be required to adapt in response to the changing climatic conditions.

4.2. Mitigating Future Nutrient Loads in Light of EU Green Deal Targets

In both catchments, target 50% nutrient reductions could only be achieved with the SM scenario. Since proposed activities under the Water Framework Directive and EU Green Deal focus mostly on land measures, our results highlight the importance of tackling secondary pollution sources using SM (Bieroza et al., 2021). Yearly TP loads in the study catchments were largely controlled by discharge and mobilization of phosphorus in the stream, irrespective of other management actions. Mobilization of phosphorus from eroding stream banks (Fox et al., 2016) or by resuspension of bed sediments (Ballantine et al., 2009) have been shown to be important drivers of TP loads. Field observations also confirm that Hestadbäcken and the upstream reaches of Tullstorpsån are characterized by large stores of fine sediments. The lack of TP response to reduced phosphorus fertilisation rate and CC differs from outcomes of Ockenden et al. (2017) and Martin et al. (2021). It thus seems that in our study sites, the secondary mobilization of existing phosphorus pools overrode the impacts of reduced phosphorus fertilisation and primary mobilization, as was also shown in Muenich et al. (2016) and Van Meter et al. (2021). However, the selection of management practices was based on existing stakeholder action in the catchments and model constraints. The reduction of phosphorus fertilizer might still be impactful as a long-term measure in

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combination with improved application and circular management of existing nutrient stores in the landscape (Haygarth et al., 2014; Withers et al., 2014). We therefore suggest that HYPE is supplemented with new routines to simulate different fertilisation techniques and draw down of organic and bound soil nutrient pools by crops and nature-based approaches.

Stream mitigation was also effective in reducing IN loads in both study catchments. In Hestadbäcken, the SM scenario was sufficient to reduce both IN and TP loads by 50% in most combinations of RCPs and periods compared to the baseline. We found that for target 50% IN load reductions in Hestadbäcken, SM would need to be complemented with a 20% reduction in mineral fertilisation in the 2022–2035 period. However, since climate change will have a strong impact on the required management actions for IN, the reduction in mineral nitrogen fertilizer could be ceased from the mid future under RCP4.5 and RCP8.5. In Tullstorpsån, the tested SM scenario did not achieve the targeted 50% reduction of TP, but combining SM with a 20% reduction in IN fertilizer led to the targeted 50% reduction of IN loads. Under RCP8.5 in 2050–2065 and 2085–2100, SM could also be combined with the use of CC to reduce IN loads with approximately 50%.

The strong reductions of IN and TP loads with stream and wetland mitigation in our work correspond well with similar studies (Arheimer & Pers, 2017; Martin et al., 2021; Tonderski et al., 2005), showing that the remediation impact depends on the stream discharge and area allocated for remediation. It is important to highlight that the modeled outcomes represent idealized implementation of management practices that in reality might be limited by a range of factors (Grimvall et al., 2014; Lintern et al., 2020). The modeled effect of CC on IN loads corresponds with modeled and empirical findings (Ruffatti et al., 2019; Speir et al., 2022). Other differences in strength of nutrient load reductions to the tested management scenarios between catchments can be explained by multiple environmental factors. Hestadbäcken is smaller, drier, and has a higher hydrological connectivity compared to Tullstorpsån, thus it will respond stronger to changes in driving climatic factors and discharge. This also explains the stronger effect of SM in Hestadbäcken, which acts as a buffer for discharge and nutrient flows. The coarser texture of agricultural soils in Tullstorpån explains the slight response to the reduction in mineral phosphorus, compared to no response in Hestadbäcken. These findings highlight that scale and environmental conditions are important precursors to the potential nutrient load reductions (Hambäck et al., 2023). A uniform 50% reduction in nutrients might thus not always be feasible and best management practices should target the highest value for money for each catchment (Djodjic et al., 2022). Therefore, more research and stringent guidelines and requirements on minimum size, location, and design of SM are needed to achieve maximum impact and avoid too narrow and thus ineffective SM zones (Arheimer & Pers, 2017; Noe et al., 2013). In the context of these multiple spatial factors, integration of this approach for backcasting nutrient loss targets in larger modeling frameworks such as S-HYPE is needed to determine to what degree our findings and recommendations can be extrapolated to larger geographical regions. In this context, it is important to note that other Best Management Practices, which were not tested for these study sites, could be more appropriate in other study sites. Nonetheless, this approach is valuable on its own as a blueprint decision support tool for testing the impacts and implementation of farmer-supported best management strategies in different agricultural headwater catchments in the context of climatic variability.

4.3. Trade-Offs and Synergies in Mitigating Nutrient Losses

We argue that SM is critical for reaching water quality targets in the study sites because it is the only measure able to reduce both TP and IN loads. However, our study shows that these reductions can be only achieved with a sufficiently large areas under SM (Figure S10 in Supporting Information S1). In both study sites, nearly 1% of the catchment should be set aside for SM to achieve set targets, which corresponds with proposed guidelines for Sweden (Arheimer & Pers, 2017). In Hestadbäcken, the SM scenario would amount to an area of roughly 61,500 m^2, which would require a stream length of ca. 3,100 m with a total floodplain width of ca. 20 m. However, this effect is catchment specific as in Tullstorpån the SM scenario (additional area of 110,000 m^2, which corresponds to ca. 6,100 m of unmitigated stream length and an average width of ca. 18 m) would still not be able to reach the 50% reduction in TP loads. Modeled SM scenarios necessarily encroach on productive arable and are thus likely to impact landowners financially. The decreases of IN crop uptake (ranging from −12% to −18%) following fertilizer reduction also indicate potential impacts on the crop yields, although these effects might be partly mitigated by increased growing season following climate change.

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As implementation of any management actions is voluntary among landowners, SM or fertilizer reduction could thus be avoided by landowners who would prefer measures such as CC that have less impact on their crop production. While this study provides estimations of nutrient load reductions, a clear estimation of cost-effectiveness of different measures is thus also needed. This should be supported by financial incentives for landowners who would be required to lose productive land and yields in order to achieve better water quality (Bol et al., 2018). However, since most agricultural streams in Sweden are bordered by 10 m grass buffer strips, which are often not fit-for-purpose in their current form (Stutter et al., 2021), we argue that these could partly be partly replaced by SM zones. When estimating cost-effectiveness other ecosystem services enabled by SM such as flood control, biodiversity improvement, and social benefits should be taken into account (Hambäck et al., 2023). Stream mitigation can achieve a higher cost-effectiveness compared to measures strictly focusing on one ecosystem service or water quality problem (Bieroza et al., 2024; Djodjic et al., 2022). For example, SM is the only tested measure that also significantly impacted discharge dynamics, which is promising for mitigating floods and droughts under climate change (Figure 4).

An important consideration is the combined effect of multiple measures. In our study, the IN load reductions when combining multiple management actions were lower than the summed reductions of the individual measures, indicating that the impact of these measures is not synergistic (Bieroza et al., 2019). This is likely because the efficacy of SM for nutrient retention and denitrification is also partly determined by the incoming nutrient loads (Hallberg et al., 2022; Noe et al., 2013), which will be lower with reduced fertilisation or CC. While the tested management practices were found to be effective in reducing total nutrient loads, they were also effective in buffering the nutrient loads during high precipitation years, evidenced by the lower overall variability and range. This is particularly pronounced for SM in years with high precipitation (Figure 4), which is important since years with high nutrient loads can destabilize aquatic ecosystems even if average nutrient loads do not increase (A. Yang et al., 2022).

4.4. Uncertainty in Modeled Nutrient Loads

Overall, the model performed well to predict flow discharge and nutrient loads, both on event-scale and yearly scale. The main source of error in discharge is likely due to both catchments being dominated by dynamic crop cultivation regimes with associated soil management and tile drainage. The drainage efficiency and depth of tile drains was assumed similar in all SLCs, while it is highly variable in reality. Small-scale precipitation events are also not always well represented in the meteorological inputs to the model and this might be a likely explanation for the absence of some discharge events in the model. The overestimation of discharge in the upstream part of Tullstorpsån and underestimation in the midstream part also indicated challenges in representing hydrological connectivity on different spatial scales.

Stochastic contribution of point sources, such as rural sewage or runoff from livestock stables, are a likely overall source of error in the nutrient loads. The most likely explanation for the underestimation of simulated TP in Hestadbäcken are the large amounts of fine sediment in the stream, which could act as a store of easily mobilized phosphorus. Moreover, the effects of high intensity precipitation events are not fully captured in the daily timestep of the model. The observed underestimation of modeled IN, especially in Hestadbäcken, might be caused by IN pools in the saturated zone and changing importance of subsurface delivery to the river discharge. Subsurface water discharged from tile drains and groundwater contributes more proportionally to the base flows and the falling limbs of storm events, thereby disproportionally increasing the IN loads (Bieroza et al., 2018). Finally, the model also did not always pick up the "first flush effects" (September 2018 in Hestadbäcken), wherein a dry summer led to lower nutrient uptake by crops, as well as exposure of dried out stream bed sediments, resulting in disproportionally high nutrient loads during the first high flow event (Bieroza et al., 2019). The larger catchment area and higher average precipitation throughout the year in Tullstorpsån make it less vulnerable to these effects.

This latter issue exposes a limitation of HYPE in flashy headwater catchments, wherein mineralization of bed sediments and crop nutrient uptake functions are fixed and not influenced by changes in precipitation and hydrology. While the model predicted strong impacts of IN availability and temperatures on crop IN uptake, HYPE is not a crop model and thus does not account for the complex interactions with other environmental variables during crop growth. For example, crop nutrient growth will also be impacted by an elongation of the growing season in Sweden under climate change (Wiréhn, 2018), which is currently not possible in HYPE. It is also likely that farmers will respond to the changing climate by adapting crop types and crop rotations, which would have an impact on nutrient dynamics. Crop growth could also be impacted by low water availability during dry summers (Grusson et al., 2021), which could result in lower nutrient uptake during summer and higher nutrient loads in winter. We also did not consider changes to rural sewage inputs in the future scenarios since it is unlikely to change, nor have a strong impact in our study sites, however, this might be impactful in more densely populated areas as shown by Capell et al. (2021). Another limitation of HYPE is in its representation of headwater SM, which is modeled as nutrient buffers between the soil and river. In reality, stream floodplains are characterized by dynamic water tables and redox conditions, which influences denitrification and nutrient mobility (Hallberg et al., 2022) that are not fully represented in the model wetland subroutine. Moreover, SM influences stream hydraulics and thus sediment and nutrient transport dynamics (Noe et al., 2013). Stream mitigation also comprises a much wider range of measures, such as bank stabilization, riparian buffer zones, remeandering, and stream ponding, which all have unique impacts on discharge and nutrient transport processes (Lammers & Bledsoe, 2017). Moreover, SM parameters were calibrated for their current size and current climate, leading to potential unrealistic modulations in the SM scenarios (Lintern et al., 2020). There is thus a need to develop additional subroutines for different mitigation measures with reciprocal feedback loops to include hysteresis effects on nutrient removal efficiency. However, the sensitivity analysis revealed that only large changes in the rating curve and outflow threshold parameters resulted into significant NSE variance for discharge and TP loads (Table S2 in Supporting Information S1). Since the outflow threshold was kept constant based on field observations or scenarios, and the rating curve parameters were strongly constrained during calibration, we argue that the uncertainty around the modeled impacts of SM is acceptable.

The used ensemble climate forecast approach provided an uncertainty range of the forecasted nutrient loads, which remained particularly large for TP loads driven by the higher predicted precipitation in the KNMI model. Moreover, climate models are still struggling to predict extreme precipitation events (Shamekh et al., 2023), which has its implications on predicted TP loads. Uncertainties are expected to decrease with improved climate forecasting under CMIP6 (Jacob et al., 2020; Krysanova et al., 2018). Finally, our model was calibrated with 2–6 years of discharge and nutrient records in current climatic conditions. These periods are relatively short, particularly for discharge in Tullstorpån, which is partly alleviated by the multigauge approach that constrains the parameter space during calibration and reciprocally validates the model setup (Krysanova et al., 2018). Moreover, the high responsivity of these headwater systems to variable weather conditions allow for a wide range of processes included in the calibration. Nonetheless, it remains uncertain how representative the current model calibration will be for future climatic conditions. Particularly non-linear responses to extreme weather, such as thresholds and hysteresis effects, are hard to predict under future climates (Krysanova et al., 2018; La Follette et al., 2021). In the context of these complex interrelations between climate dynamics, vegetation growth, and nutrient dynamics, a promising next step in modelling future water quality would be to couple hydrological models such as HYPE with other disciplinary models (e.g., hydraulic, dynamic crop growth, nutrient legacies, and land use), into a systems dynamic model (Duran-Encalada et al., 2017).

5. Conclusions

This study forecasted nutrient load exports up to year 2100 using three climate trajectories. IN loads were predicted to decrease under RCP4.5 and RCP8.5 due to temperature-driven increases in crop nutrient uptake and evapotranspiration. Under RCP2.6, IN loads were forecasted to increase in Hestadbäcken, while remaining stable in Tullstorpsån. The response of TP loads to climate change was found to be highly variable and a significant increase only occurred under RCP2.6 in Hestadbäcken. These findings highlighted the divergent responses to climate change of IN, which responds to temperature and discharge, and TP, which responds mostly to discharge.

Moreover, this work successfully demonstrated a methodology for backcasting best management scenarios to achieve the European Green Deal ambition of 50% reduction in nutrient exports from agricultural catchments under a changing climate. A reduction in mineral fertilisation was highly effective for reducing IN loads, but had almost no effects on TP loads. Likewise, CC showed a promising effect for reducing IN loads, but had almost no effect on TP loads. Increasing the size and design of SM reduced total export of both IN and TP. These outcomes can be explained by the dominance of stream processes for mobilizing TP and large phosphorus stores in soils and sediments. Since TP load reductions only respond to SM, we argue that in these two cases they are critical management actions for mitigating high nutrient loading. Overall, the diverging outcomes highlight that the best management strategies are dependent on land use, soil type, nutrient form, and the spatial and temporal effects of a changing climate. This study therefore demonstrated the potential of catchment water quality modeling as a first step in decision support to find the most effective ways to decrease nutrient loads in agricultural catchments.

Data Availability Statement

All processed geospatial data, water quality and discharge data used for calibration, climate data, model set-up, and model outcomes are accessible open access (Wynants, 2024). The HYPE model (Pers et al., 2022), HYPE tools (Capell & Brendel, 2023), and R software (R Core Team, 2022) can be accessed open access. ArcGIS is available per paid subscription (ESRI, 2020). Meteorological data was obtained from open data sets (Swedish Meteorological and Hydrological Institute, 2023a, 2023b). Used satellite imagery (Lantmäteriet, 2019, 2021), geospatial data on agricultural cropping regimes (Jordbruksverket, 2022) and soil type (Sveriges Geologiska Undersökning, 2014) is available freely for Swedish institutions.

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