How to Achieve a 50% Reduction in Nutrient Loads from Agricultural Catchments under Different Climate Trajectories?

M. Wynants[1], J. Strömqvist[2], L. Hallberg[1], J. Livsey[1], G. Lindström[2], and M. Z. Bieroza[1]

[1]Department of Soil and Environment, Swedish University of Agricultural Sciences, SE-75007, Uppsala, Sweden.

[2]Hydrological research department, Swedish Meteorological and Hydrological Institute, SE-60176, Norrköping, Sweden.

Corresponding author: Maarten Wynants (maarten.wynants@slu.se)

Key Points:

- Inorganic Nitrogen loads were forecasted to decrease in the future under Representative Concentration Pathways 4.5 and 8.5.
- Target reductions in Total Phosphorus loads could only be achieved by stream mitigation.
- Inorganic Nitrogen loads were backcasted to cumulatively decrease with fertiliser reduction, stream mitigation, and cover crops.

Abstract

Under persistent eutrophication of European water bodies and a changing climate, there is an increasing need to evaluate mitigation measures 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 contrasting agricultural catchments in Sweden 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 mobilisation. Under RCP4.5 and RCP8.5, the IN loads were forecasted to decrease from 16%-26% and 21%-50% respectively, most likely due to temperature-driven increases in denitrification 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 mitigation scenarios, including i) a 20% reduction in mineral fertiliser, ii) introducing cover crops, and iii) stream mitigation by increasing the size of floodplains and wetlands. Target TP load reductions could only be achieved by stream mitigation, which is likely due to legacy effects and secondary mobilisation within agricultural streams. Target IN load reductions were backcasted with a combination of stream mitigation, fertiliser reduction, and cover crops, wherein the required measures depended on the climate. Overall, the diverging responses of nutrients to climate change and mitigation scenarios indicate that water quality management needs to be tailored to the catchment characteristics, and to the spatial and time specific effects of climate change.

Plain Language Summary

The European Union has set a target to reduce nutrient losses from agricultural areas by 50% in 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 actions for improving water quality need to adapt over time. In this study, we simulated the future losses of two major nutrients, inorganic nitrogen and total phosphorus, for two different Swedish agricultural streams. We also modelled potential actions to reach the targeted 50% reduction in nutrient losses. The model predicted that inorganic nitrogen loads will decrease under medium and bad climate change pathways, but increase under the best climate change pathway. We found that targeted reductions in total phosphorus loads could only be achieved by mitigating streams through increasing the space for floodplains and wetlands, making it a critical action. Targeted reductions in inorganic nitrogen loads could be achieved by combining stream mitigation with a 20% reduction in mineral fertiliser, and by protecting the soil with cover crops in winter. This study has shown how we can use water quality models for identifying the required actions for reducing future nutrient loads in agricultural streams.

1 Introduction

In the agricultural regions of Northern and Western Europe, over 80% of water bodies fail to reach good ecological or chemical status, with many being in poor or bad condition according to the Water Framework Directive (Kristensen et al., 2018). Despite nutrient input reductions since the 1990s (Lu & Tian, 2017), the negative impacts from diffuse pollution on aquatic ecosystems are still evident, frequently causing eutrophication and hypoxia in inland and coastal and lake waters (Andersen et al., 2017). Decades of over fertilisation have led to the buildup of legacy nutrients in agricultural unsaturated soils, groundwater, and river sediments (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 resulted into faster nutrient and sediment exports from agricultural catchments (Blann et al., 2009; Castellano et al., 2019). The combination of high nutrient inputs, legacy stores, and artificial drainage continue to negatively impact water quality and aquatic biodiversity (Andersen et al., 2017; Ulén et al., 2007).

The European Green Deal targets to reduce nutrient losses from agricultural areas by 50% in 2030, which is envisioned by decreasing fertilisation by 20% (European Commission, 2020). However, achieving these targets will also require a structured implementation of the locally most effictive catchment mitigation measures (M. Bieroza et al., 2021). EU member states provide land owners with financial incentives for mitigating diffuse nutrient pollution (Boeuf & Fritsch, 2016; Wiering et al., 2020). These measures target three impact pathways: (1) reducing nutrient inputs to fields, (2) reducing erosion and mobilisation of nutrients, and (3) intercepting mobilised nutrient and sediment flows. Despite the large financial investments and efforts by land owners, improvements in water quality are often not observed (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; Frei et al., 2021) that override catchment mitigation efforts. Moreover, mitigation measures are often implemented with inadequate sizes, locations, or designs, based on personal preferences of landowners and financial drivers (Roley et al., 2016; Uusi-Kämppä et al., 2000), 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 mitigation measures 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 spatial and temporally varied impacts of climate change on nutrient loads (Bol et al., 2018; Zia et al., 2022). In this context, the impacts of climate change on future nutrient loads can be forecasted using process-based water quality models (Bartosova et al., 2019; Ockenden et al., 2017; Zia et al., 2022). Moreover, process models also allow to decouple catchment mitigation outcomes from climatic variability (Grimvall et al., 2014) and thereby provide a pathway for backcasting scenarios to reach the desirable future reduction in nutrient loads (Capell et al., 2021; Hankin et al., 2019).

The overall objectives of this study were (i) to forecast the impacts of climate change on future nutrient loads and (ii) to backcast the targeted 50% reduction in nutrient loads with mitigation measures across all impact pathways. Using HYPE (Hydrological Predictions of the Environment) and an ensemble of downscaled future climatic predictions, we modelled the impacts of climate change and catchment mitigation on nutrient loads in two Swedish agricultural catchments with different climate, soil textures and land use. This study focussed on three representative concentration pathways (RCPs): 2.6, 4.5, and 8.5, in the near future (2022-2035), mid future (2050-2065) and distant future (2085-2100). The three mitigation measures we modelled were i) a 20% reduction in mineral fertiliser, ii) introducing cover crops between growing seasons, and iii) stream mitigation by increasing the size of floodplains and wetlands

2 Materials and Methods

2.1 Study catchments

Hestadbäcken catchment is located in central east Sweden, and Tullstorpsån catchment is located in south Sweden. Both study catchments are agriculturally-dominated, but differ in size, cropping regimes, rainfall, and soil type (Table 1, Figure 1, Figure S1). Tullstorpsån is larger overall, but also has a higher percentage of cropland with significant amounts of root crops (8.3%) and spring crops (7.6%) besides its dominant autumn crops (57.1%). The soils are mostly loamy, with smaller pockets of moraine. 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 rainfall and a larger temperature range.

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%)
μ ± σ yearly rainfall (mm)	580 ± 131	790 ± 115
μ ± σ of >15mm rainfall days year^{-1}	6.5 ± 3.6	7.5 ± 3.3
μ ± σ of >30mm rainfall 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

A watershed assessment was performed for both study catchments with the hydrology toolset in ArcMap 10.8.1 (ESRI, 2020) and based on the 2m-resolution Grid2+ digital terrain model (Lantmäteriet, 2019). Land cover was determined by performing a maximum likelihood classification on a 0.25m-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 in autumn crops, spring crops and root crops based on 2021 crop rotation information (Jordbruksverket, 2022). Soil texture classes were derived from the 50m-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. Rainfall and temperature data were obtained from the climate 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. The stream was also sampled flow-proportionally on a fortnightly interval, and samples were analysed for total phosphorus (TP), soluble reactive phosphorus (SRP), particulate phosphorus (PP), inorganic nitrogen (IN), total nitrogen (TN), and suspended sediment

(SS) (Kyllmar et al., 2014). Flow-proportional nutrient concentrations were converted to daily estimates of nutrient loads using the average daily discharges.

Tullstorpsån was monitored intermittently at three different gauge locations (upstream, midstream, and downstream; Figure 1: Location of the study catchments, their land cover distributions and stream gauges.) for either water stage or water quality. Water stage sensors were placed upstream and midstream in May 2020. 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)*).

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

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

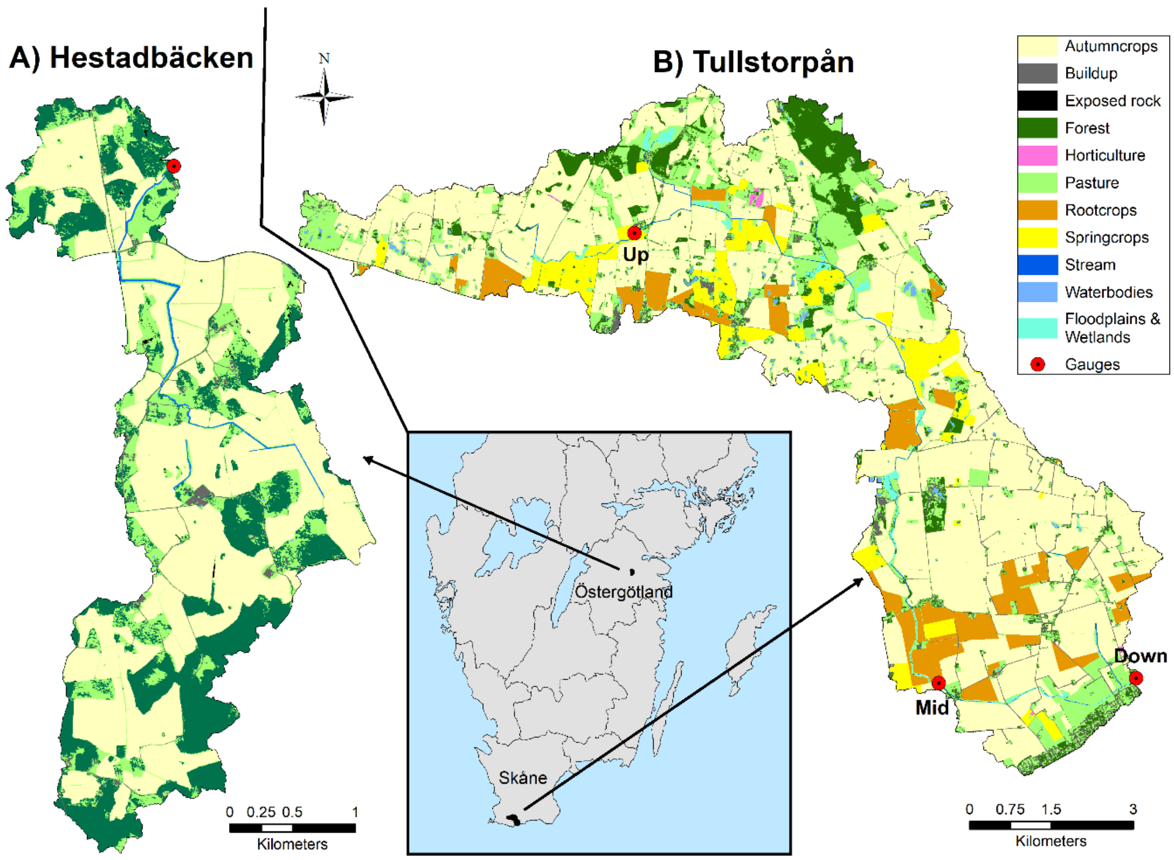

Figure 1: Location of the study catchments, their land cover distributions and stream gauges.

2.3 Model set-up, calibration and validation

The catchment models were set-up in HYPE, which is a semi-distributed and open source hydrological and nutrient transport modelling framework. Detailed information on the set-up procedures, model assumptions, and model characteristics can be found in Text S1. For more information on the representation, parameterisation, 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 was divided in three sub-catchments based on locations of stream gauges. 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 ploughing dates, as well as the amounts and dates of application of mineral fertiliser 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. Starting pools of nutrients in the soil water, organic form, and bound to soil particles were specified for different SLCs and represent legacy stores of nutrients (Strömqvist et al., 2012). Floodplains & wetlands were grouped in one SLC with a defined fraction of the catchment runoff, wherein nutrient removal was modelled using the 'wetland' subroutine with parameters controlling nutrient retention, denitrification, and macrophyte uptake (Arheimer & Wittgren, 2002; Tonderski et al., 2005). Streams were modelled as dynamic pools of sediment and PP, wherein sedimentation, erosion, and resuspension can (im)mobilise 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' measurements (Moriasi et al., 2015), i.e. Nash–Sutcliffe efficiency (NSE; Nash and Sutcliffe (1970)), Kling-Gupta efficiency (KGE; Gupta et al. (2009)), Pearsons's correlation coefficient (R), and relative bias (RE; percentage of difference). The model was optimised using the calibration methodology of Hundecha et al. (2020), wherein we combined manual and automatic calibration based on the Differential Evolution Markov Chain (DE-MC) algorithm of Braak (2006). 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 water quality parameters. A split sample was used to validate the model setups for uncalibrated periods. 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 Centre 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 representative concentration pathways (RCPs) 2.6 (stringent reduction), 4.5 (medium), 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 Centre regional climate model" (SMHI-RCA4) (Strandberg et al., 2015). An overview of the downscaled climate models can be found in Table 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 in this study to model changes in nutrient loads under future climatic conditions. Data analysis was performed in R (R Core Team, 2022). Annual yearly average rainfall, temperature, and amount of high rainfall days (>15mm per day & >30mm 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 1) compare the model predictions with empirical measurements for 2000-2022, 2) evaluate the difference between the climate models and 3) test if the projected changes in rainfall and temperature were significant. 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.

2.5 Catchment mitigation scenarios

Table 2: Overview of catchment mitigation scenarios

Scenario	Abbreviation	Impact pathway	Description
Original	Original	/	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).
1	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
2	20%Fert (Fert)	1	20 % reduction in mineral fertiliser application
3	Stream mitigation (SM)	3	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.
4	Cover crops (CC)	2	All spring crops and root crops get a cover crop in between growing seasons.
5	SM+Fert	1 & 3	Combination of scenarios 2 and 3
6	Fert+CC	1 & 2	Combination of scenarios 2 and 4
7	SM+Fert+CC	1 & 2 & 3	Combination of scenarios 2, 3, and 4

An overview of the mitigation scenarios can be found in Table 2. The selection of scenarios was based on European Green Deal Ambitions, current agronomic practices in Sweden, and stakeholder-supported stream mitigation measures in Hestadbäcken (Malgeryd et al., 2015) and Tullstorpsån (Svensson & Sundin, 2014). Impact pathway 1 (reducing nutrient inputs to fields) measures were modelled by a 20% reduction in the amount of mineral fertiliser applied in a year. Impact pathway 2 (reducing mobilisation of nutrients) measures were modelled by implementing cover crops after spring crops and root crops. Since Hestadbäcken catchment is already dominated by autumn crops, this impact pathway was only modelled in Tullstorpsån. Impact pathway 3 (intercepting mobilised nutrient flows) measures were modelled by increasing the size of floodplains & wetlands (stream mitigation scenario). Their design was modified by adding 30 cm threshold barrier to decouple with the stream and retain inundation water. The baseline area of floodplains & wetlands before recent stream mitigation activities was obtained from historical aerial photographs. Percentage reductions of all mitigation measures were calculated against the baseline situation before catchment mitigation.

3 Results

3.1 Evaluation of water quality model and climate forecasts.

3.1.1 Model discharge and nutrient load calibration and validation outcomes

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). For IN, the Hestadbäcken fit is 'satisfactory' to 'good' in the calibration period, and 'very good' in the validation period. The Pearson's correlation coefficient in both discharge and nutrient loads was high (R > 0.8). There was a slight but consistent overestimation of discharge in Hestadbäcken. Modelled TP loads responded opposite to discharge with a slight but consistent underestimation. The bias for IN in Hestadbäcken was more variable, with alternating periods of underestimation and overestimation. The modelled discharge and TP, and to a lesser extent IN, were most sensitive to the wetland parameters governing its rating curve and outflow threshold (Table S2). 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 modelled 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 in Tullstorpsån is 'very good' for both the calibration and validation periods, with no observed systematic bias and very high correlations (R > 0.9). 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 can be found in Figures S4-S9.

Table 3: Goodness-of-fit values of the calibrated model and validation period showing Nash-Sutcliffe efficiency (NSE), Kling-Gupta efficiency (KGE), Relative error (RE; %), and Pearson's correlation coefficient (R) between simulated and empirical discharges and nutrient loads. C are the calibrated years, while V are the validation years.

			Discharge				**Inorganic Nitrogen**				**Total Phosphorus**			
		Period	**NSE**	**KGE**	**RE**	**R**	**NSE**	**KGE**	**RE**	**R**	**NSE**	**KGE**	**RE**	**R**
Hestad-bäcken	C	2016-2020	0.75	0.81	14.9	0.87	0.61	0.69	-23.4	0.80	0.86	0.69	-25.9	0.93
	V	2021-2022	0.64	0.78	-10.5	0.81	0.82	0.75	-22.8	0.91	0.80	0.72	-3.0	0.95
Tullstorpsån: upstream	C	2020-2022	0.75	0.75	22.2	0.88	/				/			
	V	2022-2023	0.93	0.89	10.0	0.97	/				/			
Tullstorpsån: midstream	C	2020-2022	0.79	0.72	-24.3	0.91	/				/			
	V	2022-2023	0.93	0.87	-5.4	0.98	/				/			
Tullstorpsån: downstream	C	2016-2022	/				0.80	0.80	-16.4	0.90	0.69	0.84	-2.0	0.85
	V	2014-2015	/				0.86	0.90	0.01	0.94	0.60	0.66	-5.7	0.78

3.1.2 Comparison of nutrient load forecasts under different climate models

In the Hestadbäcken catchment, the different downscaled climate models yielded relatively similar predications in average IN loads under original conditions. The only significant or near-significant differences between model load quantifications were found under RCP8.5 (Table S3). Significant differences in TP load estimations were found in 5 of the 27 model comparisons, and near-significant differences in one (Table S4). The most notable difference are the overall higher estimated TP loads and their variability for the KNMI model. In the Tullstorpsån catchment, there were slightly more divergences between model IN load predictions (Table S5), 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 S6), wherein mostly KNMI outcomes were higher. A comparison of the empirical and predicted precipitation and temperatures between different downscaled models is given in Text S2.

3.2 Modelled nutrient loads under different climatic trajectories

3.2.1 Trends in projected rainfall and temperature

Average annual rainfall and high rainfall days were predicted to increase significantly in both Hestadbäcken and Tullstorpsån under RCP8.5 (Figure S2). Under RCP2.6, the climate ensemble predicted a distinct increase in high rainfall days in 2050-2065, but no significant increase in average rainfall for both catchments. Under RCP4.5, the amount of average yearly rainfall and high rainfall days increased but not significantly. The predicted changes in temperature were more uniform between both sites (Figure S3). Under RCP4.5 and RCP8.5, significant increases of respectively 0.7 ˚C and 1.5 ˚C were predicted per period.

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 2). 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.

In Tullstorpsån, the ensemble model predicts no significant change under RCP2.6. Under RCP4.5, the ensemble model predicts a decrease of 11% and 16% in the 2050-2065 and 2085-2100 respectively, while under RCP8.5 a respective decrease of 4% and 21% is forecasted. However, there are some trend divergences between the models, wherein ICHEC predicts a decrease in 2050-2065 under RCP2.6. Under RCP4.5, clear decreases are visible for both periods in the ICHEC and MPI models, however, for the KNMI model, an increase in the IN load is predicted in the 2085-2100 period. Under RCP8.5, the ICHEC model predicts a strong decrease followed by a slight increase, the MPI model predicts a slight increase followed by a strong decrease, and the KNMI model predicts a stability followed by a strong decrease. The percentages of IN load change of the ensemble and individual climate models can be found in Table S7 for Hestadbäcken and Table S9 for Tullstorpsån.

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

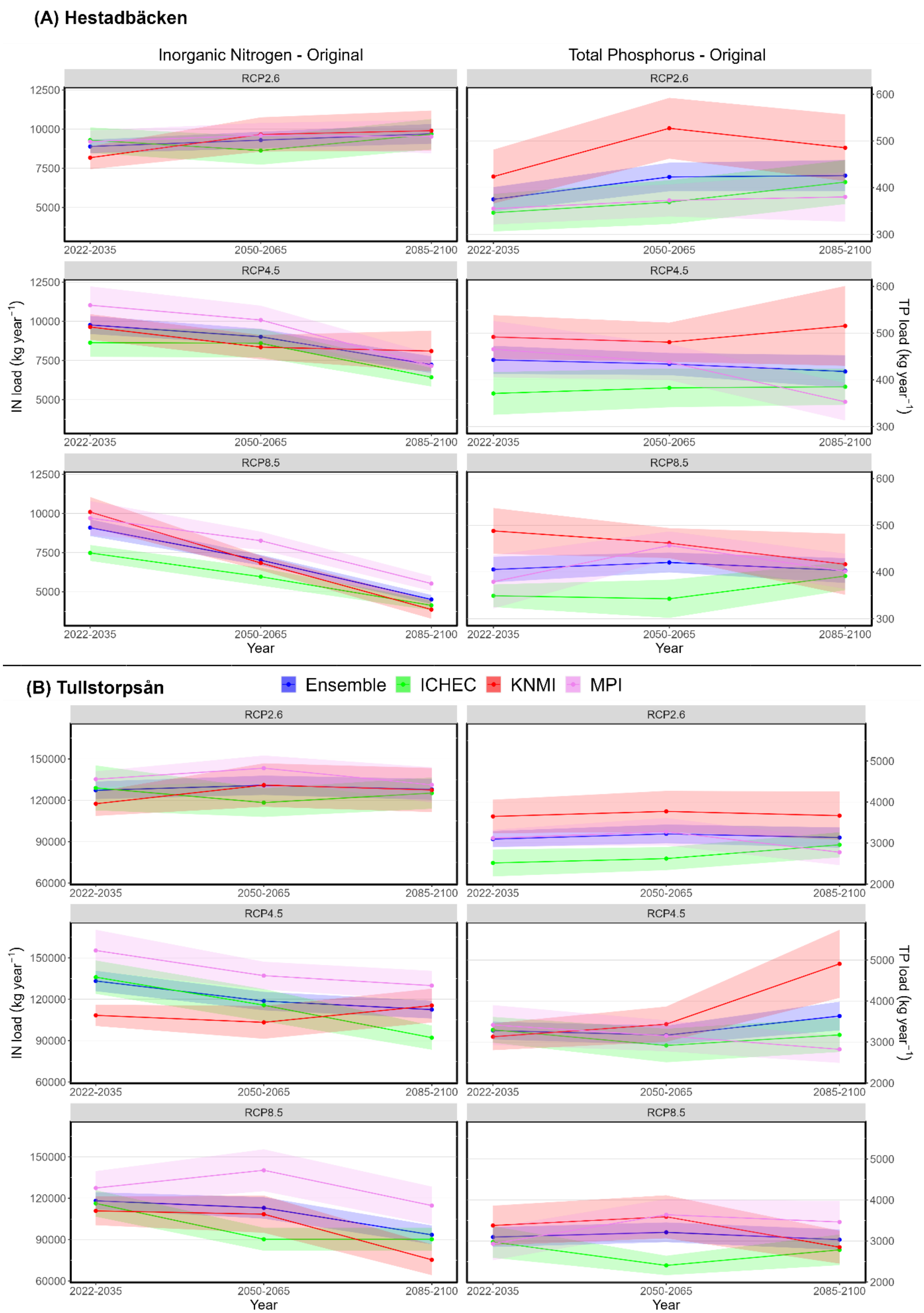

Figure 2: Changes in nutrient loads for the (A) Hestadbäcken and (B) Tullstorpsån catchments under different climate models, the ensemble, and RCPs. Results are for the original catchment situation (see Table 2). Point and lines represent the mean yearly loads for the period, while the ribbons represent one standard deviation.

In Tullstorpsån, the ensemble model predicted no significant changes in TP loads under RCP2.6. 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. The percentage of TP load change can be found in Table S8 for Hestadbäcken and Table S10 for Tullstorpsån.

3.3 Backcasted nutrient loads under catchment mitigation scenarios and climate change

3.3.1 Hestadbäcken

The stream mitigation 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 3). The average changes in loads range between -50.7 to 8.3% for IN and between -7.5 to 8.3% for TP across the RCPs and periods. Moreover, small increases in area for stream mitigation will not have a strong effect on load reductions (Figure S10). A sole reduction of 20% in mineral fertiliser input had strong effects on IN loads, with significant IN load reductions ranging between 27.0% and 68.9%. However, a reduction in mineral P inputs had no effect on the predicted TP loads. A 20% reduction in mineral P fertiliser yielded load changes ranging between a 7.6% reduction and a 10.8% increase, which is not significantly different from loads with current P fertilisation rates. The stream mitigation scenario (floodplains & wetlands increase from 0.03% to 0.8%) was predicted to decrease IN loads between 39.0% and 74.4%. The stream mitigation scenario was found to be the only effective measure for reducing TP loads and yielded reductions ranging between 41.5% and 51.7%. Combining stream mitigation with a 20% reduction in fertilisation led to the highest nutrient IN load reductions, ranging between 56.6% and 83.2%. Catchment mitigation measures were also shown to decrease high nutrient load years and led to a lower range in nutrient yields (Figure 3: Effects of catchment mitigation scenarios on IN and TP loads under different RCPs. 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 line on the plots show the respective median value and 50% reduction of the baseline scenario (no stream mitigation) in the 2022-2035 period.).

3.3.2 Tullstorpsån

The stream mitigation already in place (0.82% of catchment are floodplains and wetlands) was shown to have a significant impact of nutrient loads compared to the baseline situation before stream mitigation (0.13% of the catchment), with load reductions ranging between 6.0% and 28.1% for IN, and between 8.0% and 19.9% for TP (Figure 3). A sole reduction of 20% mineral fertiliser reduced loads ranging between 33.5% and 56.1%. The 20% reduction in mineral P fertiliser only yielded an additional 1% in TP load reduction. The stream mitigation scenario (increasing the size of floodplains & wetlands from 0.8% to 1.0%) resulted in IN load reductions ranging between 28.9% and 50.9%, and TP load reductions between 30.4% and 41.4%. The inclusion of cover crops reduced IN loads with 11.5% to 39.8%, but had almost no effect on TP loads. Combining a 20% decrease of mineral fertiliser and cover crops resulted in decreases of IN loads ranging between 36.2% and 59.0%, and 10.4% to 21.4% decreases in TP loads. Combining a 20% decrease of mineral fertiliser and stream mitigation 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% fertiliser decrease, stream mitigation, and cover crops, yielded IN load reductions between 51.6% and 68.9%, and TP load reductions between 32.2% and 42.5%.

4 Discussion

4.1 Impact of future climate on nutrient loads in agricultural catchments

The response of nutrient loads to climate change differed between inorganic nitrogen and total phosphorus and between both catchments. In Hestadbäcken, the predicated 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 2). These diverging trends indicated dominance of increased soil N mineralisation and precipitation driven IN mobilisation under RCP2.6, versus increased temperature driven denitrification and evapotranspiration under RCP4.5 and RCP8.5. This indicates that future IN loads depend on the interplay between the effects of increased temperature, denitrification and evapotranspiration leading to reduced loads versus the effects of increased precipitation and IN mobilisation leading to increased loads. This pattern is supported by empirical findings of increased denitrification following increasing stream temperatures in the Po river, Italy (Gervasio et al., 2022), 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 denitrification, future denitrification rates will also be controlled by hydrological responses to a changing rainfall distribution (Peterson et al., 2001).

No distinct trends in TP loads could be observed under a changing climate. The variability in TP loads between the years is predicted to increase in the future (Figure 3) due to large projected differences in yearly rainfall and extremes. For Hestadbäcken under RCP2.6, the model forecasted that increased rainfall (intensity) will on average lead to slightly higher TP loads, most likely driven by increased runoff and discharge peaks. Vice versa, under RCP4.5 and RCP8.5, the projected increases in temperature and associated evapotranspiration and reduced mobilisation offsetted this effect. 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.

Overall, the complex responses of IN and TP loads to different RCPs and forecasted periods demonstrated the challenges of predicting nutrient losses and eutrophication in agricultural catchments under a changing climate. The predicted increase in nutrient loads in the near future will require immediate mitigation actions to be undertaken to achieve EU green deal targets. These mitigation measures will need to focus on buffering high flow events with large mobilisation of nutrients. Water quality management in the medium to distant future will be required to adapt in response to the periodic climatic conditions.

4.2 Mitigating future nutrient loads in light of EU Green Deal targets

In both catchments, target 50% reductions in both nutrients could only be achieved by including the stream mitigation scenario. Since proposed activities under the Water Framework Directive and EU Green deal focus on land measures, our results highlight the importance of tackling secondary pollution sources using stream mitigation (M. Bieroza et al., 2021). Yearly TP loads in the study catchments were largely controlled by mobilisation of phosphorus in the stream, irrespective of reduced phosphorus fertilisation or inclusion of cover crops. Mobilisation 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 characterised by large stores of fine sediments. The lack of TP load response to a reduction in phosphorus fertilisation corresponded with results by Capell et al. (2021) in the Baltic sea region, but contrasted with the outcomes from Ockenden et al. (2017) in England. The strong binding of phosphate to fine-textured soils and sediments in the study sites might form an explanation (Sandström et al., 2021). Moreover, legacy phosphorus pools, both in organic form and attached to soil particles, in Swedish topsoils are large compared to annual fertiliser rates (Capell et al., 2021). In this study, the legacy phosphorus thus overrode the impacts of reduced phosphorus fertilisation. However, reduction of phosphorus fertiliser might still be valuable long-term measure in combination with circular management of available legacy stores (Haygarth et al., 2014; Withers et al., 2014). We therefore suggest that water quality models be supplemented with new routines to simulate draw down of legacy nutrient levels through agronomic practices.

Stream mitigation was also effective in reducing IN loads in both study catchments. In Hestadbäcken, the stream mitigation 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, stream mitigation 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 mitigation measures for IN, the reduction in mineral nitrogen fertiliser could be stopped from the mid future under RCP4.5 and RCP8.5. In Tullstorpsån, the tested stream mitigation scenario did not achieve the targeted 50% reduction of TP, but combining stream mitigation with a 20% reduction in IN fertiliser led to the targeted 50% reduction of IN loads. Under RCP8.5 in 2050-2065 and 2085-2100, stream mitigation could also be combined with the use of cover crops to reduce IN loads with approximately 50%.

The different strength of nutrient loads reductions to the tested mitigation scenarios between the two study catchments can be explained by multiple environmental factors. Hestadbäcken is smaller, drier, and has a higher hydrological connectivity compared to Tullstorpsån, thus will respond stronger to changes in driving climatic factors. This also explains the stronger effect of stream mitigation 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 indicate that a uniform 50% reduction in nutrients might not be feasible and mitigation measures should target the highest value for money for each catchment. In the context of these multiple spatial factors, further research is needed to determine to what degree these findings and recommendations can be extrapolated to larger geographical regions.

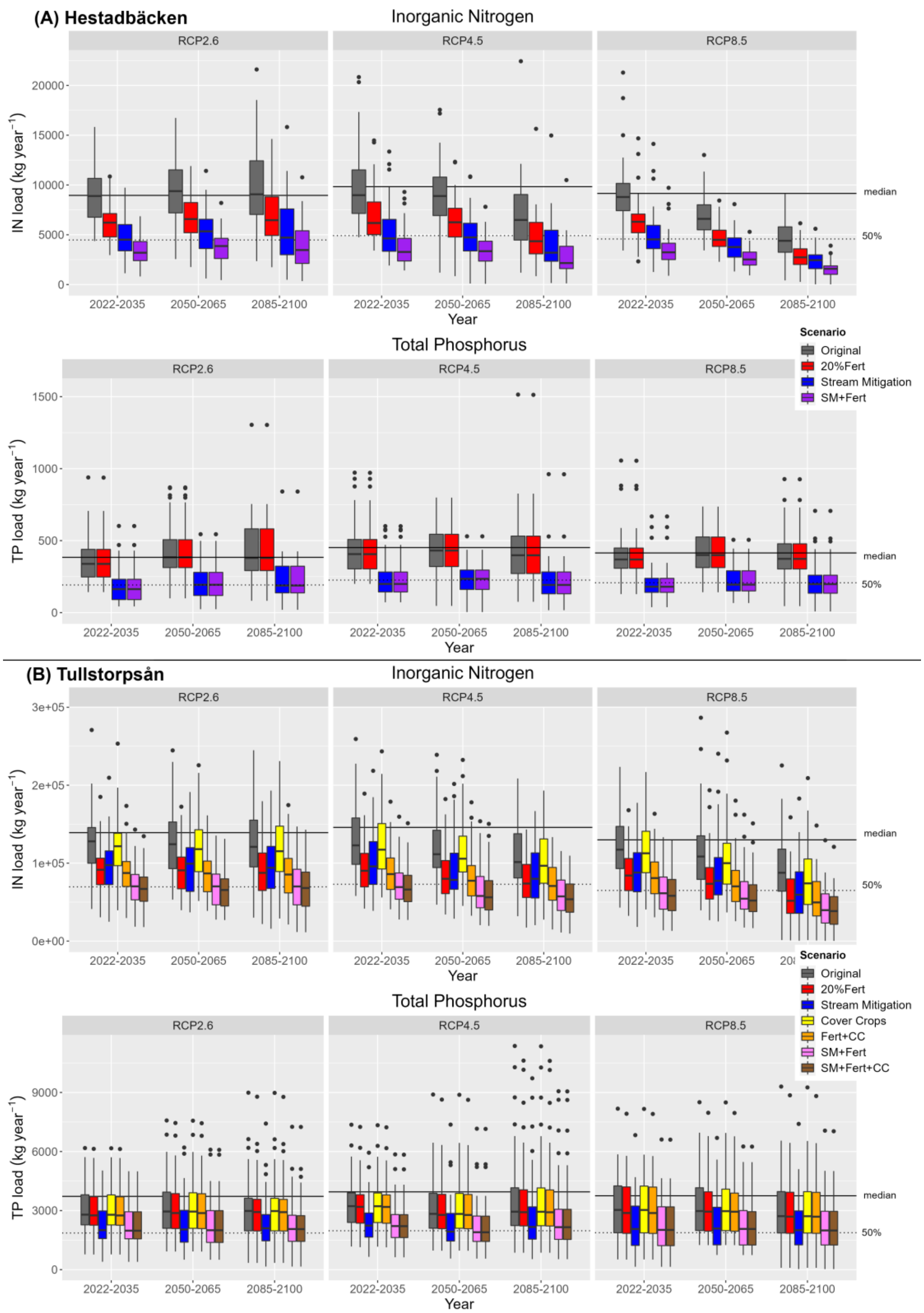

Figure 3: Effects of catchment mitigation scenarios on IN and TP loads under different RCPs. The boxplots depict the outcomes from the ensemble climate model, where median values are shown by the central line, interquartile range by boxes, the range by whiskers, and the outliers by points. The black and dotted line on the plots show the respective median value and 50% reduction of the baseline scenario (no stream mitigation) in the 2022-2035 period.

4.3 Trade-offs and synergies in mitigating nutrient losses

We argue that stream mitigation is critical for reaching water quality targets in the study sites because it is the only measure effective for reducing both TP and IN loads. However, our findings also highlight the importance of the size of stream mitigation zones for achieving significant reductions in nutrient loads (Fig. S10). In both study sites, nearly 1% of the catchment should thus be set aside for floodplains and wetlands to achieve set targets, which corresponds with proposed guidelines for Sweden (Arheimer & Pers, 2017). In Hestadbäcken, the stream mitigation scenario would amount to an area of roughly 61,500 m^2, which would require a stream length of ca. 3,100 m with an average width of ca. 20 m. However, this effect is catchment specific as in Tullstorpån the stream mitigation 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 allow to reach the 50% reduction in TP loads. Modelled stream mitigation scenarios are implemented downstream and are thus likely to reduce productive arable land. As implementation of any mitigation measures is voluntary among landowners, stream remediation could be avoided by landowners who would prefer more soil-based measures, such as cover crops, that have less impact on their crop production. A clear estimation of cost-effectiveness of different measures is thus needed. This should be also supported by financial incentives for landowners who would be required to lose productive land in order to implement more effective stream mitigation (Bol et al., 2018). When estimating cost-effectiveness other ecosystem services enabled by stream mitigation such as flood control, biodiversity improvement, and recreation 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. Moreover, more stringent guidelines and requirements on minimum size and design of stream mitigation are needed to achieve maximum impact and avoid too narrow stream mitigation zones (Arheimer & Pers, 2017; Noe et al., 2013).

An important consideration is the combined effect of multiple measures. In our study, the IN load reductions when combining multiple mitigation measures were lower than the summed reductions of the individual measures, indicating that the impact of these measures is not synergistic (M. Bieroza et al., 2019). This is likely because the efficacy of stream mitigation 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 cover crops. While catchment mitigation measures were found to be effective in reducing total nutrient loads, they were also effective in buffering the nutrient loads during high rainfall years, evidenced by the lower overall variability and range in nutrient loads. This is particularly pronounced for stream mitigation in years with high rainfall (Figure 3), which is important since years with high nutrient loads can destabilise aquatic ecosystems even if average nutrient loads do not increase (A. Yang et al., 2022).

4.4 Uncertainty in modelled nutrient loads

Overall, the models 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 because both catchments are dominated by crop cultivation 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 rainfall 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 mobilised phosphorus. Moreover, the effects of high intensity rainfall events are not fully captured in the daily timestep of the model. The observed underestimation of modelled IN, especially in Hestadbäcken, might be caused by legacy IN 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 (M. Z. 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 (M. 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 mineralisation of bed sediments and crop nutrient uptake are not influenced by rainfall and hydrology. It is expected that potential crop growth in Sweden will increase under the predicted higher temperatures due to a longer growing season (Wiréhn, 2018). However, since crop nutrient uptake can be potentially limited by low water availability during dry summers (Grusson et al., 2021), this could result in higher nutrient loads in winter. Another limitation of HYPE is in its representation of floodplains & wetlands, which are modelled as nutrient buffers between the soil and river. In reality, floodplains & wetlands are characterised by dynamic water tables and redox conditions, which influences denitrification and nutrient mobility. Moreover, stream mitigation influences stream hydraulics and thus sediment and nutrient transport dynamics (Noe et al., 2013). Moreover, stream and wetland parameters were calibrated for their current size, leading to potential unrealistic modulations in the stream mitigation scenarios. 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. Since the outflow threshold was not calibrated but based on field observations or scenarios, and the rating curve parameters were strongly constrained, we argue that the uncertainty around the modelled impacts of stream mitigation is miminal. 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 rainfall in the KNMI model. Uncertainties are expected to decrease with improved climate forecasting under CMIP6 (Jacob et al., 2020; Krysanova et al., 2018). 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 increased denitrification 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 of IN and TP dynamics to climate change.

Moreover, this work successfully demonstrated a methodology for backcasting catchment mitigation 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 is highly effective for reducing IN loads, but has almost no effects on TP loads. Likewise, cover crops showed a promising effect for reducing IN loads, but have almost no effect on TP loads. Increasing the size and design of floodplains & wetlands reduces total export of both IN and TP. These outcomes are likely due to the dominance of stream processes for mobilising TP and large legacy phosphorus stores in soils and sediments. Since TP load reductions only respond to the floodplains & wetlands scenarios, we argue that in these two cases they are critical for catchment mitigation plans. Overall, the diverging outcomes highlight that the optimal mitigation scenarios are dependent on land use, soil type, nutrient form, and the spatial and temporal effects of a changing climate. This study has demonstrated the potential of catchment water quality modelling as a first step in decision support to find the most effective ways to mitigate nutrient loads in agricultural catchments.

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Open Research

All processed geospatial data, water quality and discharge data used for calibration, climate data, model set-up, and model outcomes can be previewed by reviewers on the Wynants et al. (2023) dataset. The HYPE model and HYPE tools can be accessed open access via respectively https://sourceforge.net/projects/hype/files/ and https://github.com/rcapell/HYPEtools.

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