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Nitrogen and phosphorus discharge from small agricultural catchments predicted from land use and hydroclimate

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A B S T R A C T

Excess nutrients cause eutrophication of freshwaters all over the world. Decision-support tools are needed to assess nutrient discharges from catchments. This paper used a 28-year nutrient-discharge, hydroclimate and land-use history of small rural catchments to calibrate a simple nitrogen (N) and phosphorus (P) runoff model. The N and P runoffs declined following the post-Soviet collapse of agriculture, and stabilised at low output during the 1990s and early 2000s. Introduction of the European Union Common Agricultural Policy (CAP) re-intensified the agriculture and somewhat rebounded the N and P discharges. Thus, the history of the catchment represents a broad range of land-management systems. Our objective was to explain annual nutrient runoffs from small rural catchments by five factors: hydroclimate, soil type, land-use type, fertilisation and the autumn soil-nutrient stock. Our model independently predicted the eight-year mean N and P losses from a test set of small agricultural catchments in Estonia. This shows the impact of political decisions on agricultural contamination of waters. We can suggest our robust model as a decision-making tool for land-use management in small agricultural catchments.

1. Introduction

Agricultural land-use change is a major driver of habitat loss and ecosystem functions (Vitousek et al., 1997; Tilman, 1999). Europe is experiencing a mix of land-use changes – a decline of agricultural area in some regions and an increase in others (Prishchepov et al., 2013; Jepsen et al., 2015). A major driver of land-use change was the collapse of socialism and subsequent transition from state-controlled to market economies in Eastern Europe in the early 1990s (Prishchepov et al., 2013; Jepsen et al., 2015). The disintegration of collective agriculture had substantial economic and ecological impacts (Mander and Palang, 1994). Estonian agricultural production plummeted during the early 1990s (Mander and Palang, 1994) and fertiliser use declined fivefold (Statistics Estonia, 2017). That did not reduce nutrient discharges immediately (Stålnacke, 1996; Löfgren et al., 1999). However, towards the mid-1990s, the nutrient discharges stabilised at low levels (Stålnacke et al., 2002; Iital et al., 2005; Mourad et al., 2006; Nõges et al., 2010). Similar trends have been described in heterogeneous catchments elsewhere (Larsen et al., 1998; Grimvall et al., 2000; Chang, 2008). During the 2000s, there were no general trends in N and P runoff from agricultural catchments in Estonia and Latvia (Iital et al., 2010; Jansons et al., 2011; Iital et al., 2014).

To evaluate land management effects on nutrient losses, spatial computer models have been developed. The emphasis of the research has shifted from empirical models such as USLE (Wischmeier and Smith, 1978), its modifications MUSLE and RUSLE (EPA, 1992) to more sophisticated simulation models, such as ANSWERS (Beasley et al., 1980), HSPF (Bicknell et al., 1984), AGNPS (Young et al., 1987), PULSE (Bergström et al., 1987), SWAT (Arnold et al., 1993; Santhi et al., 2001; Oeurng et al., 2011; Arnold et al., 2012), HBV-NP (Arheimer and Wittgren, 1994), MONERIS (Behrendt et al., 2000), Riverstrahler (Billen and Garnier, 2000), PolFlow (de Wit, 2001), INCA (Wade et al., 2002), STICS-MODCOU (Ledoux et al., 2007; Beaudoin et al., 2016), TNT2 agro-hydrological model (Ferrant et al., 2013); SPARROW water-quality model of the USGS (Alexander et al., 2007; Schwarz et al., 2011; Preston et al., 2011, 2013; García et al., 2016), and HYPE (Lindström

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et al., 2010; Arheimer et al., 2011). However, land-use managers usually lack high-resolution monitoring data on multiple environmental variables from large areas, such as soil temperature, light, groundwater flow, or water-retention time, to feed the models. Often enough such data are too expensive and the setup of and the data assemblage for dynamic models takes too much time for decision making in river basin management. Therefore, parsimonious models that provide reasonably accurate results and are based on readily available data would be more feasible for the purpose (Dupas et al., 2013, 2015, 2017). Such a model would require a minimum number of independent input and transport factors (as explained by Pärn et al., 2012). Sandner et al. (1993) proposed a parsimonious catchment N-runoff model using two dimensionless input factors (fertilisation and land use), two dimensionless transport factors (soil texture and precipitation) and an initial value (20 kg N ha^{-1} y^{-1}). Mander et al (2000) applied the model for the Porijõgi catchment, Estonia, and adapted the model for P runoff (using 0.5 kg P ha^{-1} y^{-1} as the initial value) with an excellent predictive skill for N and fair predictive skill for P runoff. Henine et al. (2017) argued that the best initial value for catchment N-runoff models is the autumn nitrate pool of the arable land in the catchment. Nonetheless the predictive skill of a parsimonious N or P model has not been tested on a variety of agricultural catchments across an entire state. The objective of our study was to predict long-term N and P discharges from agricultural catchments in Estonia from a parsimonious model including soil nutrient pool, fertiliser inputs, land use, hydroclimate and soil texture.

2. Material and methods

2.1. The model

In our choice of model, we relied on our literature overview (see Introduction). Thus we used the simple empirical model of Sandner et al (1993) which had provided reasonably accurate results before (Mander et al., 2000) and can be based on readily available data. The model was used as follows:

$$Q = F_1 \times F_2 \times F_3 \times F_4 \times c, \quad (1)$$

where:

Q was modelled annual N or P runoff, kg ha^{-1} y^{-1},

F_1 was a discrete factor characterising the dominant land-use pattern in the catchment,

F_2 was a discrete soil factor (remained constant during the whole study period),

F_3 was a discrete fertilisation factor,

F_4 was a hydroclimate factor for annual precipitation or, where available, streamflow divided by the whole-period average, and

c was the initial NO_3–-N or available P pool in the plough layer of the arable land (estimated 20 kg NO_3–-N ha^{-1} and 1‰ of plough-layer total soil P = 0.5 kg P ha^{-1} for the years 1987–1991 and adjusted according to actual soil N and P measurements during the later periods; Henine et al., 2017).

The parameter values used by Mander et al., 2000 are presented in Table 1.

2.2. Calibration methods

In order to test the suitability of the model (Sandner et al., 1993; Mander et al., 2000; Eq. (1); Table 1) for not only historical but also contemporary agricultural catchments, we calibrated the model with data from the Porijõgi and its subcatchments under various agricultural settings: Soviet-style collective agriculture (1987–1991), non-intensive agriculture (1992–2000), intensive agriculture (2001–2015), and organic farming (the Sipe stream subcatchment in 2001–2015). For the calibration we calculated N and P losses from the Porijõgi river catchment and its subcatchments in Estonia under various land-use regimes using two basically different methods: 1) the model (Eq. (1)) and 2) direct streamflow and nutrient-concentration measurements. As goodness-of-fit criteria between the the model and the direct measurements we calculated the root mean square error (RMSE) and a Nash-Sutcliffe index for each subcatchment.

Table 1
Nutrient runoff model parameters (adapted from Sandner et al., 1993 and Mander et al., 2000).

Factors	Specification, description		Values	
F_1	Grassland, forest, < 20% arable		1.0	
Dominant	Grassland, < 40% less intensively cultivated arable lands		1.5	
land use	Mixed grasslands and arable land > 50%		2.0	
	Arable lands > 50%		2.5	
	Intensively cultivated arable land > 60%		3.0	
	> 75% intertilled or winter crops		3.5	
F_2			Automorphic	Hydromorphic
Soil	Sand		1.0	0.7
	Loamy sand		0.7	0.5
	Loam		0.5	0.3
F_3	N (kg ha^{-1} y^{-1})	P (kg ha^{-1} y^{-1})	N	P
Fertilisation	10	< 5	0.1	0.1
(Average for	100	50	0.5	0.5
arable land)	250	70	1.3	1.3
	300	100	1.7	1.7
F_4	Annual precipitation P, mm y^{-1} or streamwater discharge Q, m^3 y^{-1} standardised to long-term annual precipitation or streamwater discharge		$F_4 = P_{annual}/P_{long-term\ annual}$ or $Q_{annual}/Q_{long-term\ annual}$	
Hydrology				

2.3. Calibration sites

The Porijõgi is a tributary of the Emajõgi river (Fig. 1). The Porijõgi flows into Lake Peipus. The river's 258 km^2 catchment lies at the border between two landscape regions: the Southeast-Estonian till plain and Otepää Heights (Varep, 1964; Arold, 2005). Its central and northern parts lie in a ground till plain. The altitude is mainly between 30–60 m with an undulated relief and intersecting primeval valleys (0.1–3 km wide and down to 40 m deep) formed by streams during the Pleistocene and transformed by glaciers during the last glaciation. Portions of the valleys are filled with glaciofluvial sand and gravel. The southern part of the drainage basin lies on the northern slope of the Otepää heights, which are composed of kames with a great variety of glacial deposits. The altitude of this region goes up to 120 m; relative height reaches 30–35 m (Varep, 1964; Mander et al., 1998; Arold, 2005). The bedrock is formed by red Devonian sandstone (compact sandstone with clay and siltstone layers of the Aruküla and Burtnieki stages) overlaid by loamy sand-till of the Weichselian glaciation or glaciofluvial and glaciolacustrine sands and gravel. The Devonian sandstone lies at a depth of 2 m (lower course) to 60 m (Otepää Heights). Groundwater table varies from the ground to −20 m depending on the relief. The river and its tributaries flow in deeply cut glacial valleys. Upland soils are predominantly podzoluvisols, planosols and podzols on loamy sand and fine sandy loam with a surface soil organic matter content of 1.6–1.9%. Soil pH is 5.6–6.5 with a declining trend during recent decades, due to the intensive fertilisation that was practiced up to the end of the 1980s (150 kg N, 70 kg P and 100 kg K ha^{-1} y^{-1} on arable lands and cultivated grasslands; Mander et al., 1989) which resulted in Ca and other cation leaching. On the other hand, the Ca content in podzoluvisols and

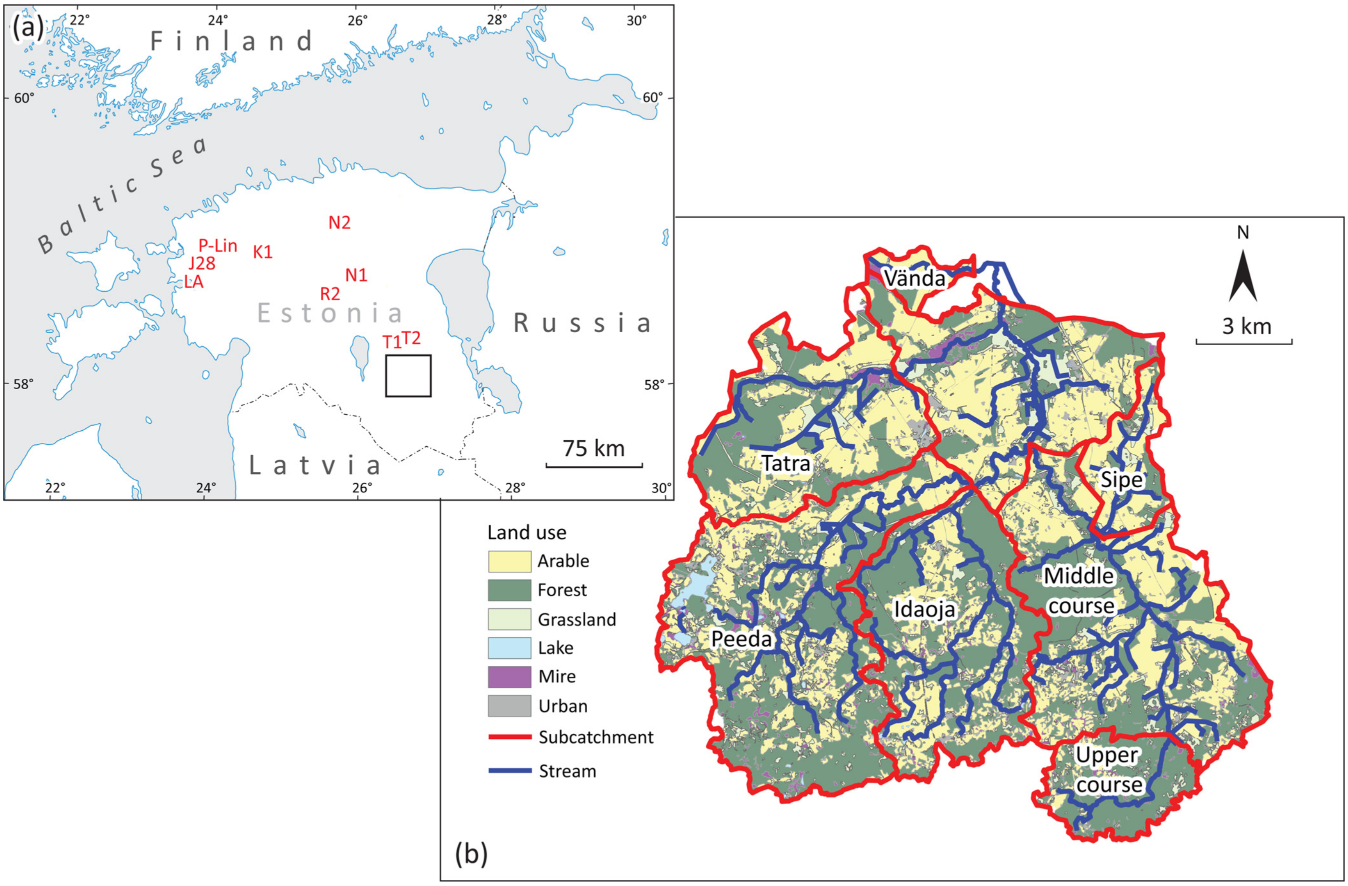

Fig. 1. (a): Location of the calibration catchment (inset) and the validation catchments (see details in Section 2.7); (b): land use in the calibration catchments according to the Estonian Basic Map of 2012.

podzols is normally low (0.1–1.0% CaO) and is enriched by liming in the agricultural fields. The long-term leaching has resulted in elevated Ca content in groundwater (80–160 mg L^{-1}) and the accumulation of spring tufa deposits in valleys where the seeping groundwater is well buffered by natural riparian and hyporheic zones.

About 45% of the catchment has the potential to be used as arable land. The main crops are wheat and rapeseed with mainly mineral fertilisers used. In valleys and other depressions, mixed forests, alder stands, willow scrub and various meadows predominate on gleysols and fen histosols. The interfaces between the fens and the uplands form barriers against nutrient runoff (Pärn et al., 2010).

The share of abandoned agricultural land in 1987–1997 expanded from 1.7 to 10.5%, while arable lands decreased from 41.8 to 23.9%. Forested areas, natural and cultivated grasslands increased from 40.0 to 44.8%, from 6.7 to 10.3% and from 6.4 to 6.8% respectively. In abandoned agricultural lands, young forest ecosystems began to develop. For instance, on automorphic soils grey alders *(Alnus incana)* and silver birches *(Betula pendula)* were the predominating pioneer species, while wet meadows were covered by willow *(Salix spp)* scrub or birch *(Betula pubescens)* forest. Due mainly to the deterioration of drainage systems, wetlands increased in area from 3.4 to 3.7%. In subcatchments, land-use change differed notably. The wooded Upper Course subcatchment experienced no significant change, whereas the agricultural Sipe and Vända showed a remarkable transition similar to the entire catchment. In the Sipe subcatchment, the proportion of arable land fell from 58.5 to 19.1%, and the amount of abandoned agricultural land increased from 1.2 to 27.2%, whereas forested areas, wetlands, and grasslands showed a slight increase. Natural strips under fen, grey alder, and willow were dominant in the Sipe floodplain while the Vända had no riparian buffer zones. In the latter about 90% of the arable land became seminatural and cultivated grassland (a rise from 0.5 to 49.5% and from 0.9 to 24.6% respectively; Mander et al., 2000). Between 1997 and 2001 land use stabilised at low intensity (Kull et al., 2005) while the nutrient discharges decreased substantially (Mander et al., 1998). Only the wooded Upper Course subcatchment showed no significant change in annual nutrient runoff (Kull et al., 2005). Between 2001 and 2006 the extensification of the previous decade was reversed (Fig. 2). Abandoned agricultural land decreased from 11% (Mander

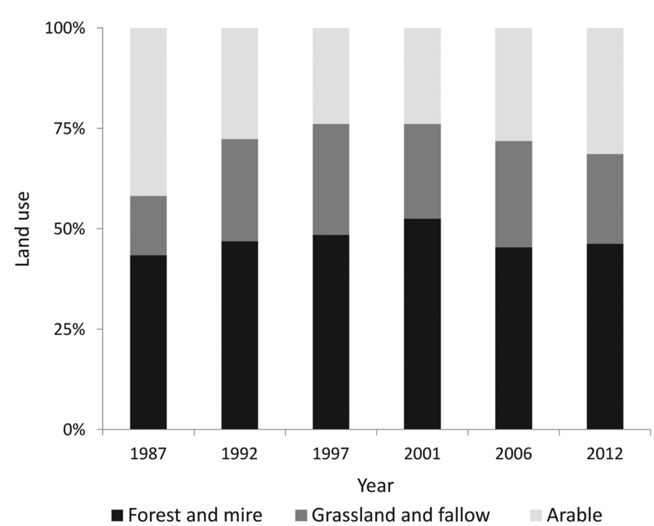

Fig. 2. Land-use dynamics in the Porijõgi catchment according to field surveys by Mander et al. (2000) and CORINE Land Cover databases of 2001, 2006 and 2012.

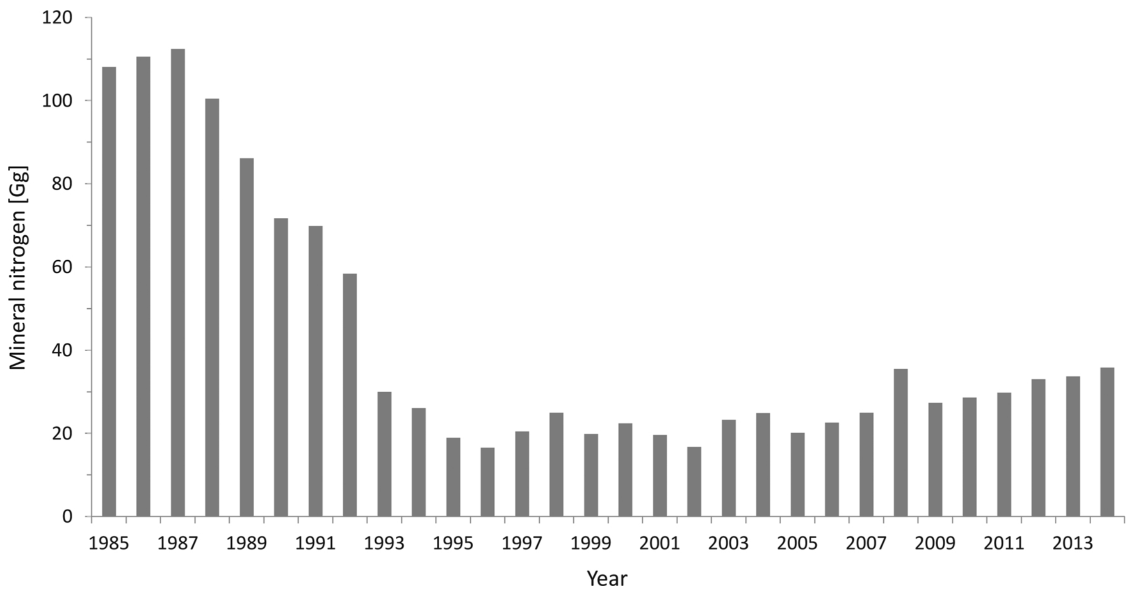

Fig. 3. Mineral N application dynamics in Estonia. Data from Statistics Estonia (2017).

et al., 2000) to 6%, while grassland and arable land surged to 26% and 28%, respectively. Forest and wetland remained at 45%.

After the introduction of the EU CAP in the early 2000s, the agricultural Sipe and Vända subcatchments returned to their previous state. In Sipe, arable land rose to 55% and abandoned agricultural land fell close to zero, whereas forested area depleted. In Vända, the grasslands were returned to the arable land raising its proportion to 62% of the catchment. The changes were followed by an increase in fertiliser use. While almost no P or N was added to the fields between 1992 and 2000, the rates rose to an average of 100 kg N ha^{-1} y^{-1} and 40 kg P ha^{-1} y^{-1} in 2007–2009 and 110 kg N ha^{-1} y^{-1} in the intensively used agricultural lands of the Porijõgi in 2013–2015. Annual dynamics of mineral fertilisation in the catchment as documented in the farmers' books followed the fertilisation history of Estonia (Statistics Estonia, 2017; Fig. 3). In the Sipe catchment, however, the arable land was mostly used for organic farming, and the fertilizer addition equals approximately 50 kg N ha^{-1} y^{-1} from leguminous crops.

2.4. Streamflow data for calibration

Daily precipitation data were collected at the Tartu Observatory (15 km from the centre of the catchment). Daily average streamflow [m^3 s^{-1}] of the Porijõgi was determined from the Parshall flume by Estonian Weather Service (EWS) at the Reola gauge. Streamflow was calculated by the EWS from stage-height measurements. For the subcatchments, we gauged streamflow from the closing weirs at the following frequencies during 2007–2013: eleven gaugings from March till December 2007, seven from March till December 2008, once a year at the spring-peak flow between 2009 and 2012, and ten gaugings between April and December 2013. Peak flow was captured in all the years. Streamflow at the Reola gauge correlated strongly with all the subcatchments ($R^2 > 0.8$). Thus linear interpolation was used to fill the gaps in daily streamflow data of the subcatchments. Three subcatchments with data available from 1987 to 1998 were studied more closely: 1) Upper Course (12.3 km^2) on the slope of the Otepää Heights (relatively undisturbed and dominated by forests), 2) Sipe (9 km^2) with high agricultural land use and well-developed riparian buffers along the stream, and 3) Vända (2.2 km^2) with a high proportion of arable land but few riparian buffers. In the Vända cathment, the intensive agriculture is not associated with any significant measures to control nonpoint pollution.

2.5. Water chemistry data for calibration

Bimonthly water chemistry measurements from the Reola gauge were used for the entire catchment during 1987–2015. For the eight subcatchments, measurements by Mander et al. (2000) were used for the years 1987–1998. During 2007–2013 we sampled water from the closing weirs of the subcatchments at the same frequency as the streamflow data (see the previous section). We analysed the samples for ammonium N, nitrate N, total N, and total P following the APHA standards (APHA-AWWA-WEF, 2005). In order to interpolate to the days without measurements, streamflow events were identified on the Reola gauge graph. The nearest nutrient-concentration measurement within the streamflow event was assigned to a day (Rekolainen et al., 1991). Daily N and P runoffs, kg ha^{-1} d^{-1} were calculated. Annual N runoff, kg ha^{-1} y^{-1} was calculated from the daily runoffs as follows:

$$Q_N = \sum_{i=1}^{n} Q_n \times conc_{Nn} \tag{2}$$

where:

Q_N was annual N runoff,

n was the number of days in the year,

Q_n was the average streamflow during the day, m^3 s^{-1}, and

$conc_{Nn}$ was the nearest nutrient-concentration measurement within the streamflow event (Rekolainen et al., 1991)

2.6. Land-use and soil data for calibration

Land-use data for Porijõgi in years 1987 to 1997 was obtained from annual field surveys by Mander et al. (2000). For the years 2000 to 2015 we used CORINE Land Cover data from the years 2001, 2006 and 2012. Location of abandoned agricultural land was checked during field work in May 2008. The following land-use categories were determined: arable land (including cultivated grassland, fertilised, with the sod tilled after each 5–8 years), forest (including shrub), grassland (without fertilisation, extensively mowed or non-mowed, including fallow land), and mire (swamps, fens, bogs, wet meadows). For soil data, the 1: 10 000 Estonian Soil Map was used. Fertilisation data, kg agricultural land ha^{-1} y^{-1}, was inquired from collective farms' statistics during 1987–1991 and private farms' field books during 1991–2013.

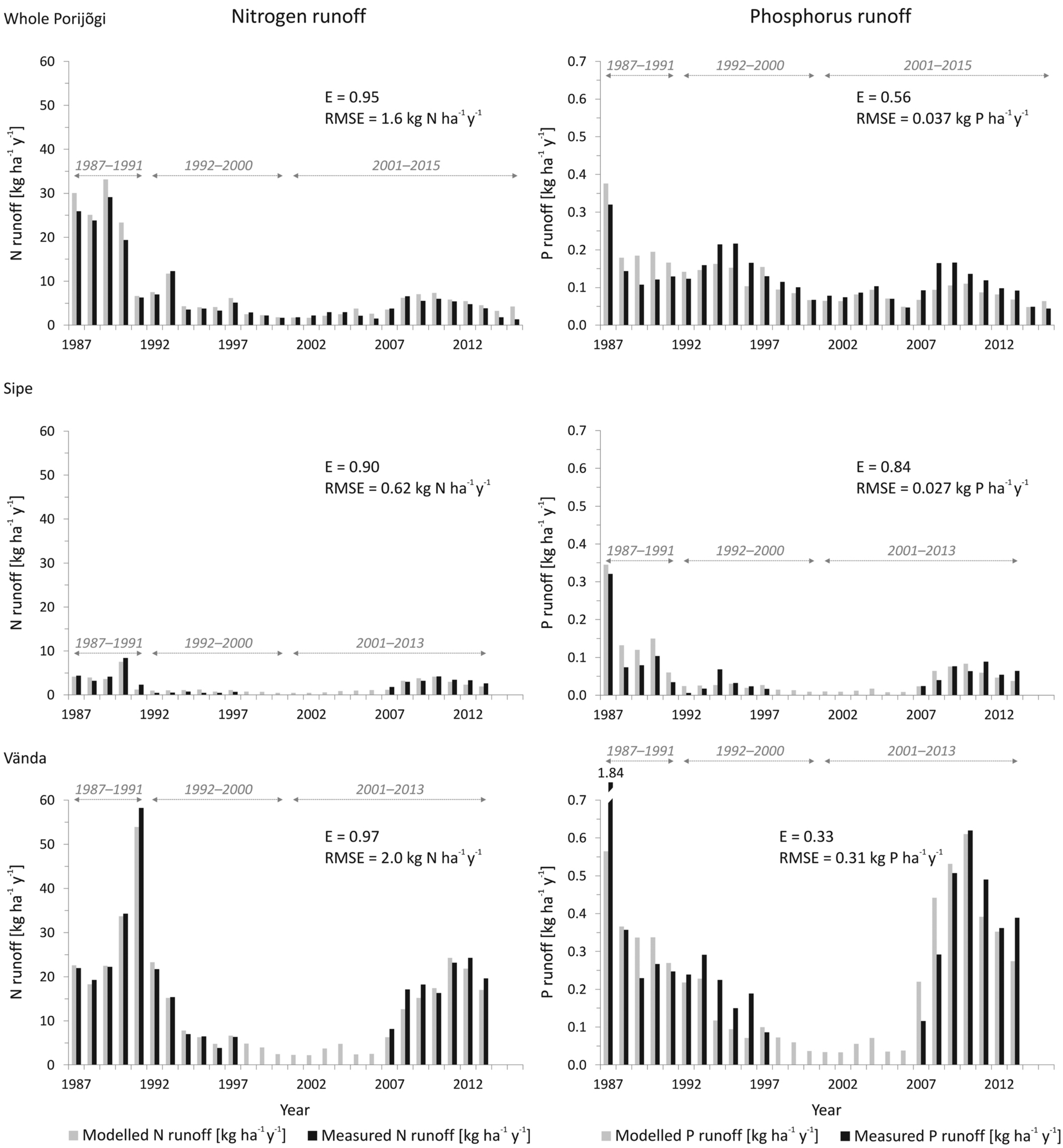

Fig. 4. Nutrient dynamics vs. model predictions in the Porijõgi and two agricultural subcatchments. c = 20 kg N ha^{-1} and 0.5 kg P ha^{-1} (Eq. (2)) except c = 10 kg N ha^{-1} 0.25 kg P ha^{-1} in the whole Porijõgi river and the Sipe stream from 1998 onwards, c = 21.7 kg N ha^{-1} in the Vända ditch from 2011 onwards, and c = 0.584 kg P ha^{-1} in the Vända ditch from 2007 onwards. E: Nash-Sutcliffe index; RMSE: root mean square error. The periods: 1987–1991 – Soviet state and collective farms, 1992–2000 – transition (private farms, free market, no governmental support), 2001–2015 – agricultural subsidies under the EU CAP (Common Agricultural Policy).

2.7. Validation

We used data from the 9 agricultural drainage-water stations in the monitoring and evaluation programme of *Estonian Rural Development Plan 2007–2013* (Fig. 1; Supplementary Table). Exact locations of the individual farmlands were disclosed according to Public Information Act. Size of the catchments was < 31 km^2. Streamflow and concentration of total N and total P was sampled every two weeks under the commission of the Agricultural Research Centre of Estonia during 2007–2014 (ARC, 2016). The ARC measured soil N and P content in October 2007. Soil type and texture class were determined from the 1: 10 000 Estonian Soil Map. N and P fertilisation rates were taken from the farm-gate nutrient balance and use study based on books and interviews conducted by the ARC in the 120 farms using land in the

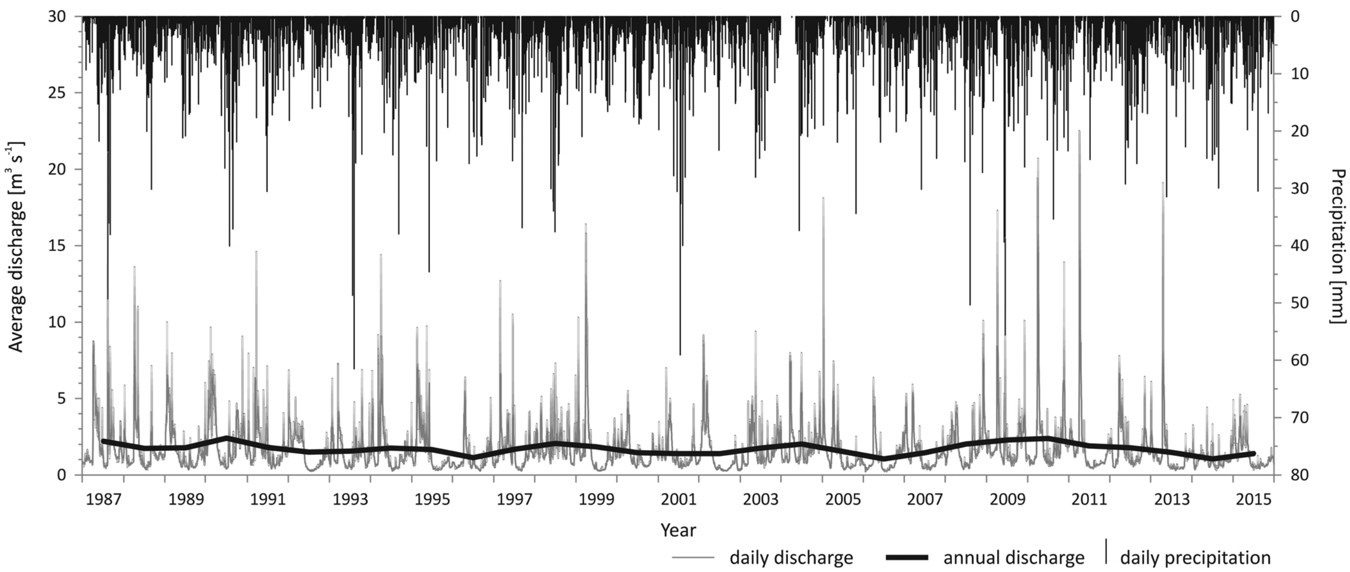

Fig. 5. Precipitation and streamflow dynamics in the Porijõgi river (Reola gauge).

validation catchments.

The J28 catchment was located in a drained calcareous fen in the Oru municipality. The soil was dominantly Mollic Gleysol. Land use alternated between clover, rye, barley, rapeseed, and winter wheat. Fertilisation, both organic and mineral varied from 80–160 kg N ha^{-1} y^{-1}.

The K1 catchment was located in a drained calcareous fen in the headwaters of the Vigala river catchment in the Rapla municipality. The soil was dominantly Mollic Gleysol. Land use alternated between early barley, summer barley, rapeseed and annual leguminous grassland. Mineral fertilisation ranged from 0 (leguminous grassland) to 106 kg N ha^{-1} y^{-1}.

The LA catchment was located in a drained fen in the headwaters of the Rägina ditch in the Martna municipality. The soil was dominantly Eutric Gleysol. Land use alternated between leguminous and poaceous grassland, barley, oats, and buckwheat. Organic fertilisation ranged from 80 to 166 kg N ha^{-1} y^{-1}.

The N1 catchment was located in the headwaters of the Oostriku stream in the Koeru municipality. The landscape was a calcareous moraine plain. The soils were Endocalcaric Cambisols. Land use alternated between barley and annual poaceous grassland. Fertilisation, organic and mineral ranged from 80 to 166 kg N ha^{-1} y^{-1}.

The N2 catchment was located in a drained calcareous fen the Ambla municipality. The landscape was dominated by the floodplain of the Jänijõgi stream headwaters and the surrounding calcareous moraine plain. The soil was Mollic Gleysol and the surrounding uplands were covered by Endocalcaric Cambisols. Land use alternated between barley, bare fallow and annual leguminous grassland. P discharge was not measured in the N2 catchment. Fertilisation, organic and mineral ranged from 80 to 119 kg N ha^{-1} y^{-1}.

The Plin catchment was located in a drained calcareous fen in the Oru municipality. The soil was dominantly Mollic Gleysol. Land use alternated between clover, winter wheat, winter turnip rapeseed, summer barley and summer rapeseed. Fertilisation, organic and mineral ranged from 80 to 166 kg N ha^{-1} y^{-1}.

The R2 catchment was located in a calcareous moraine plain in the headwaters of the Räpu stream in the Kabala and Kõo municipalities. The soils in the floodplains were dominantly Mollic and Molli-Histic Gleysols. In the uplands the soils were Endocalcaric Cambisols. Land use alternated between rapeseed, barley and annual grassland. Fertilisation, organic and mineral ranged from 35 to 230 kg N ha^{-1} y^{-1}.

The T1 and T2 catchments were located in calcareous moraine plains in the Tähtvere municipality. The main soil type was Endocalcaric Cambisol. Land use alternated between rye, barley and winter wheat. Fertilisation, organic and mineral ranged from 80 to 166 kg N ha^{-1} y^{-1}.

We modelled annual N and P runoff between 2007 and 2014 using Eq. (1). For our F_1 factor we used the land use reported by the monitoring programme. The land-use data was presented in one the following categories per year: barley, buckwheat, clover, leguminous grassland, oat, poaceous grassland, rapeseed, rye, timothy, wheat, winter or wheat. We used precipitation data for our F_4 hydroclimate factor (Sandner et al., 1993) derived from the closest automatic rain gauge of Estonian Weather Service. That was preferred to streamflow data for two reasons: 1) they were independent from the N and P runoff measurements calculated from streamflow and 2) public weather data is often readily available from national surveys whereas streamflow.

We checked the independence of all factors (Eq. (2)) calculating correlations between them among the catchments. We validated our model results with annual N and P losses calculated by the ARC from the measured streamflow and N and P concentrations in the drainage water. As the criterion for goodness of fit between the direct measurements and our model we calculated a Nash-Sutcliffe index for the set of 9 catchments.

3. Results and discussion

3.1. Nitrogen and phosphorus dynamics in the calibration site

The rebound of agricultural land use from 2001 to 2010 increased N and P runoff in the Porijõgi river (Fig. 4). We observed no significant rise in streamflow (Fig. 5) which explained only 10% and 14% of annual N and P runoffs from the whole Porijõgi river catchment during 1987–2015 ($R^2 = 0.10$ and 0.14, respectively; $p > 0.05$). As the main change in the streamflow pattern, spring and autumn floods resumed (Fig. 6). While no particular changes in nutrient runoff was observed for the forested Upper Course, nutrient losses increased most in the intensively managed Vända subcatchment. The also intensively managed Sipe subcatchment experienced a milder rise, maybe due to the well-developed riparian buffers. The nutrient runoffs stabilised during 2010–2015.

We observed an important difference between the Soviet-style and the EU-subsidised agriculture in timing and methods of fertiliser application (Bigeriego et al., 1979; Pearce, 1998) as Soviet-style winter fertilisation stopped. This brought the share of nitrate-N in total N discharges during winter (December to March) from 76.9% in the

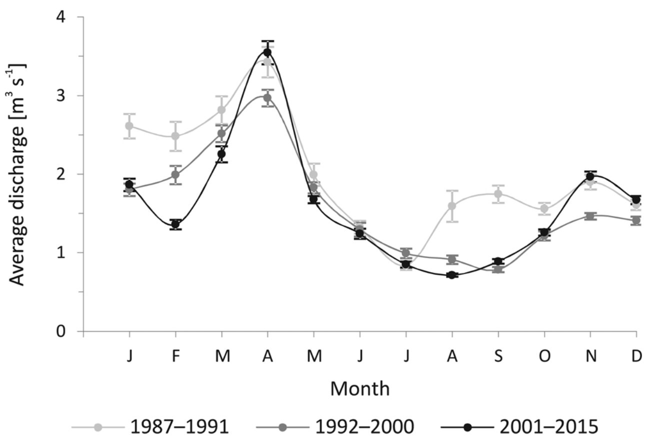

Fig. 6. Changes in monthly average streamflow patterns in the Porijõgi (Reola gauge) in various periods (see explanation in Methods and Fig. 1). The whiskers indicate standard error.

Table 2
Share of ammonium- and nitrate-nitrogen from total nitrogen in various periods (see explanation in Methods and Fig. 1).

Period	Whole year		Winter (December–February)	
	NH_4^+-N/ total N [%]	NO_3^--N/ total N [%]	NH_4^+-N/total N [%]	NO_3^--N/total N [%]
1987–1991	5	61	5	77
1992–2000	5	60	7	60
2001–2015	4	60	4	65

Soviet era down to 60.3% and 64.6% in the modern periods, respectively (Table 2).

3.2. Model calibration results

We compared our models against the measured N and P runoff from the whole Porijõgi, and Upper Course, Sipe and Vända subcatchments. Nash–Sutcliffe index and root mean square error (RMSE) were 0.88 and 1.6 kg N ha^{-1} y^{-1}, and 0.56 and 0.037 kg P ha^{-1} y^{-1} in the whole Porijõgi, which indicated a very good fit for N and a fair fit for P (Fig. 4). F_1 (land use; R^2 with annual N and P runoff 0.89 and 0.15 respectively) and F_3 (fertilisation; R^2 with annual N and P runoff 0.63 and 0.09 respectively) were the most important factors for the nutrient dynamics whereas F_2 (soil texture) remained the same throughout the calibration period, and F_4 (hydrology) apparently also had minor impact on the nutrient dynamics. Likewise, long-term annual nitrogen transport increase has been found due to the intensification of N input in forest/agricultural and residential areas (Boyer et al., 2002; Boithias et al., 2014; Chen et al., 2016). As the main output of the calibration procedure, it showed no need to modify the model parameters used by Mander et al. (2000).

3.3. Model validation results

Intersite variances between our model factors (Eq. (1)) were insignificantly correlated between each other among the validation catchments ($p > 0.1$). Our model failed to explain interannual variance of the nutrients. However our model did predict period-average N and P runoff from the validation catchments (2007–2014) very well. Nash–-Sutcliffe index between period-average measured and modelled N runoffs was 0.88 ($n = 9$ catchments; Fig. 7a). RMSE = 2.3 kg N ha^{-1} y^{-1} which, in a field of 0–22 kg N ha^{-1} y^{-1} may be considered a reasonable deviation. Our model slightly overestimated P runoff by a margin of RMSE = 0.056 kg P ha^{-1} y^{-1}. However the Nash–Sutcliffe index between period-average measured and modelled P runoffs was 0.76 ($n = 8$ catchments, Fig. 7b). Therefore it is fair to assume that our model will work in other small agricultural catchments under a similar climate with no significant point-source pollution. This also shows that a sophisticated hydrological unit (e.g. the retention factor *R* in Nutting; Dupas et al., 2013, 2015, 2017) can be omitted and annual rainfall will suffice as the coefficient of hydroclimate.

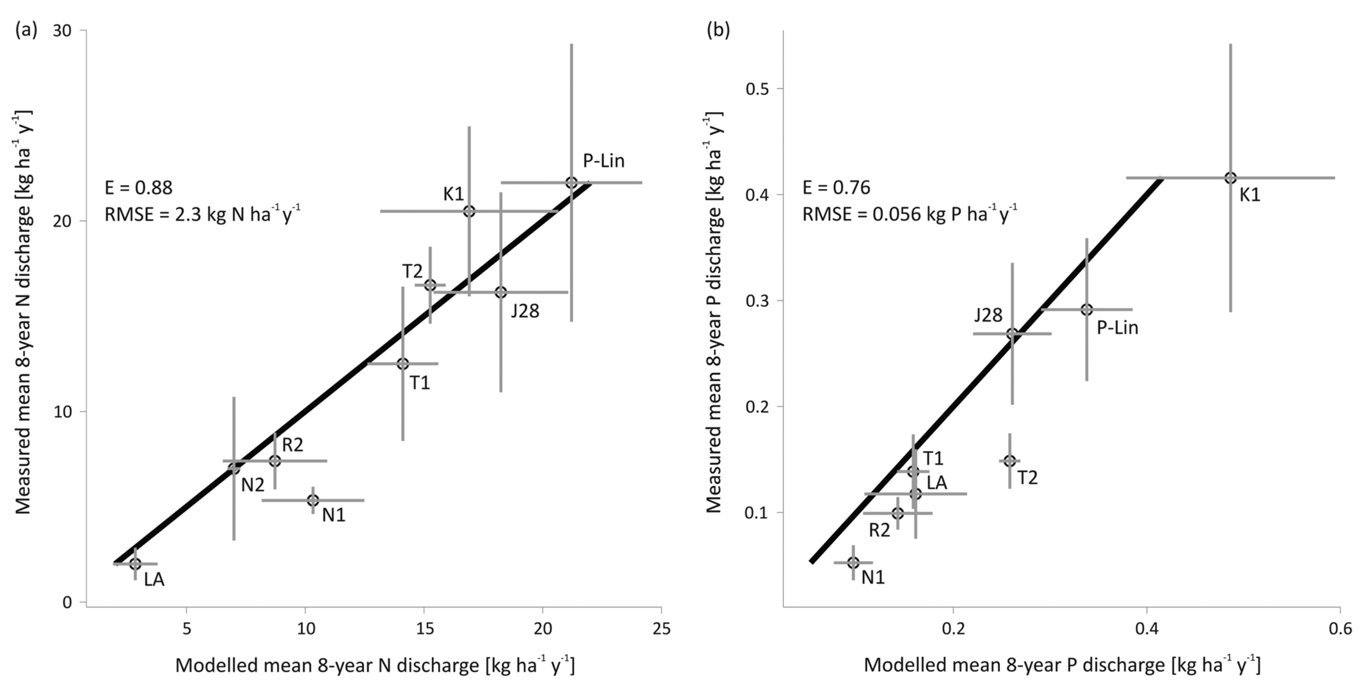

Fig. 7. Model validation. Circles: 2007–2014 average nutrient discharge from catchment; whiskers: standard error of mean. Black line: one-to-one measured nutrient discharge. (a): measured and modelled nitrogen runoff; n = 9 catchments; (b): measured and modelled phosphorus runoff; n = 8 catchments. E: Nash-Sutcliffe index; RMSE: root mean square error. See Section 2.7 for catchment descriptions.

4. Conclusions

N and P discharges from the Porijõgi river declined remarkably with the collapse of the Soviet agriculture. They somewhat rebounded with the re-intensification of agriculture. The broad range of N and P discharges between the three study catchments allowed us to calibrate our simple empirical model using only five easily attainable input variables: land use, soil, hydroclimate, fertilisation and soil nutrient pool. This model predicted differences in long-term N and P discharges from a set of validation catchments all over Estonia very well. This shows the importance of agricultural practices to pollution of waters. We can suggest our model as a tool for land-use planning in small agricultural catchments with similar natural and political history.

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Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.landusepol.2018.03.048.

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