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Title: A site-specific prediction model for nitrogen leaching in conventional and organic farming

Year: 2024

Version: Published version

Please cite the original version:

Joel Kostensalo, Riitta Lemola, Tapio Salo, Liisa Ukonmaanaho, Eila Turtola, Merja Saarinen,

A site-specific prediction model for nitrogen leaching in conventional and organic farming,

Journal of Environmental Management, Volume 349, 2024, 119388, ISSN 0301-4797,

https://doi.org/10.1016/j.jenvman.2023.119388.

Journal of Environmental Management 349 (2024) 119388

Contents lists available at ScienceDirect

Journal of Environmental Management

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

Research article

A site-specific prediction model for nitrogen leaching in conventional and organic farming

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ARTICLE INFO

Keywords:
Nitrogen leaching
Organic farming
Finland
Eutrophication
Life cycle assessment
Emission model

ABSTRACT

Food production has a profound eutrophication impact on waterbodies via nutrient leaching. To provide reliable life cycle assessments of the eutrophication potential of agricultural products, accurate nitrogen leaching models are needed. Although many dynamic nitrogen leaching models are in use, their suitability for farm-level assessments remains limited when their requirements for site specific data or numerous parameters are not met. In Finland, less data intensive leaching models for life cycle assessments have been developed using data from conventional farming, however, the suitability of these models for organic farming remains unknown. In this work, we developed new nitrogen leaching models that are applicable to both conventional and organic production. While this paper does not aim to argue in favor of organic or conventional farming it provides tools that can be used to inform decisions about management practices from the environmental perspective. We utilized up to 16 years of field measurements from two leaching fields in Finland. We developed prediction equations for nitrogen leaching for two soil types: sand soil and clay soil. According to our statistical analysis based on the data, the relevant factors for explaining nitrogen leaching included soil type, rainfall, whether the farming is done organically, and the availability of nitrogen for leaching. Computed nitrogen balance as such was found to be a poor proxy for nitrogen available for leaching, while nitrate nitrogen concentration measurement of the soil carried out in the fall was found to be a valuable predictor. Organic farming, with a crop rotation resembling that of conventional farming, resulted on average in 20% less nitrogen leached per hectare as compared to conventional farming with 95% C.I. [-34%, −3%]. The developed models are suitable for integration into a life cycle assessment framework, and especially the models utilizing nitrate nitrogen were shown to be applicable to a wide range of different crop types, making the model well-suited for plots with diverse crop rotations.

1. Introduction

Food production and consumption have a significant environmental impact with most of it attributable to agricultural production (Garnett, 2011). The loss of nutrients from cultivated soil and their entering into waterbodies is one of the most critical environmental issues related to agriculture (Willett et al., 2019).

Life cycle assessment (LCA) is a commonly used tool for estimating the environmental impact of food products as well as other agricultural outputs (Sala et al., 2017). LCA can be utilized in different situations (McLaren et al., 2021) and is by its very nature a powerful tool for assisting decision making in production chains related to environmental improvements (ISO, 2016a, b). One of the key impact categories assessed in LCA studies is the eutrophication impact, relating to excess nutrients being distributed in the environment, while other frequently considered categories include climate impact and acidification (see e.g., Baldini et al., 2017; McClelland et al., 2018). There are several different impact assessment models (IA models) and impact category indications for estimating the potential eutrophication impact to waterbodies that are compatible with the LCA framework (e.g., Potting and Hauschild, 2006; Hauschild et al., 2013; Bulle et al., 2019). While the spatial resolution of these models varies from regional to global, all of them are

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https://doi.org/10.1016/j.jenvman.2023.119388
Received 3 May 2023; Received in revised form 9 September 2023; Accepted 12 October 2023
Available online 27 October 2023

based on nitrogen (N) leaching and phosphorus (P) emissions into waterbodies in addition to atmospheric emissions. The model-based estimation of these impacts is carried out at the Life Cycle Inventory phase of LCA. Freshwater eutrophication is mainly due to P emissions, while marine eutrophication is typically considered separately as it relates mostly to N emissions (Helmes et al., 2012). EU's Product Environmental Footprint initiative PEF (EU, 2021; Huijbregts et al., 2017) follows this strict division, though some suggest that both P and N impacts should be routinely included (Morelli et al., 2018). In Finland, when assessing the impacts on the Baltic Sea and the waterbodies that flow into it, both P and N contribute significantly and are thus accounted for (Seppälä et al., 2004).

All IA models require leaching estimates as input data. There are different approaches to choose from for approximating the leaching depending on the goals of the assessment and the IA model chosen (McLaren et al., 2021). Several commonly used IA models, such as ReCiPe 2016 (Huijbregts et al., 2017), utilize so-called characterization factors. For example, the model for assessing freshwater eutrophication in Huijbregts et al., (2017) is based on a generic approximation where 10% of P leaches from agricultural soil (Bouwman et al., 2009). This number is then multiplied by region-specific fate and exposure factors (a.k.a. effect factors) to obtain the characterization factors (Morelli et al., 2018). EU's PEF initiative requires the use of ReCiPe's IA method, with robust emission factors available for cases where accurate region-specific factors are not available (EU, 2021). The characterization factors are typically based on large catchment models, e.g., the EUTREND/CARMEN model (Struijs et al., 2009) in ReCiPe 2008 made for Europe, or the Global NEWS 2-DIN model (Cosme et al., 2018) in ReCiPe 2016 developed for other regions. While readily calculated factors for large regions make the assessment easy to carry out, this also results in very limited local accuracy (McLaren et al., 2021). To obtain sufficient local accuracy, agrological or agroecological models specific to certain sites or situations should be applied to obtain leaching estimates instead of only providing emission factors for larger geographical areas. However, because these models consist of several N pools and they typically link N dynamics to carbon (C) dynamics, they require extensive data, and the calculation is often time-consuming (Bhar et al., 2021). This makes the models difficult to exploit in large-scale as well as site-specific farm-scale LCA studies. Another approach is to use regression models based on empirical data to estimate leaching, as has been done in, e.g., the World Food LCA Database for products from non-European countries (Nemecek et al., 2019) where the SQCB-NO3 model (Faist Emmenegger et al., 2009) is applied for regions where robust emission factors are not available. Following this approach, the standard practice in Finnish LCA calculations has been to estimate N leaching with a regression model with N balance as a covariate with separate models fitted for three soil types: sand, clay, peat, as well as arable or grassland cultivation separately (Salo and Turtola, 2006, Saarinen, 2011).

Indeed, since agriculture is the main source of the environmental impacts of food products, in general, the impact assessment models and the emission models behind them should be applicable to individual farms, i.e., they should be able to produce site-specific results, to facilitate the efforts of the farmer to reduce the environmental impact. For example, input data needed for the assessment should be available at the farm level, while the model should include all relevant factors relating to the impact. In addition, when more sustainable cultivation practices are adopted, the emission estimations should reflect and document these improvements. Therefore, simple, accurate, and practical site-specific models at farm level are needed for LCA studies to complement the regional and global estimations.

Promotion of organic production is one of the most used strategies to mitigate the environmental impacts of agriculture. For example, the European Commission has set a goal that 25% of the EU's agricultural land should be under organic farming by 2030. Farming practices differ between organic and conventional farming, and thus do also the environmental impacts; organic farms tend to have higher soil organic matter content and lower nutrient losses, e.g., N leaching and nitrous oxide and ammonia emissions per unit area compared to conventional farming, though not necessarily per unit of food produced according to LCA studies (Tuomisto et al., 2012). Furthermore, there have been some concerns about the overall sustainability (Leifeld, 2012). However, it is still unclear whether LCA modelling can capture the sustainability benefits of organic production sufficiently well (Meier et al., 2015; van der Werf et al., 2020; Chiriacò et al., 2022). Therefore, methodological development of LCA modelling practices for organic farming is necessary. For example, the N leaching models utilized in the Finnish LCA calculations have been developed using only data from conventional farming (Salo and Turtola, 2006).

The aim of this study was to develop accurate prediction equations for estimating N leaching from fields farmed using either organic or conventional methods. Even though this paper does not aim to argue in favor of organic or conventional farming, it does provide tools that can be used to inform decisions about management practices from the environmental perspective, which needs to be considered together with, e.g., crop yield in tandem for fully informed decision making. We also aimed to provide a rough estimate for the effect of choosing organic farming over conventional methods.

2. Materials and methods

2.1. Materials

2.1.1. Field measurements

The experimental data for modeling was collected from 16 sand soil plots (about 0.16 ha each) at Toholampi (63°49′N, 24°09′E, 83 m a.s.l) and from 6 clay soil plots (about 0.5 ha each) at Yöni, Jokioinen (60°49′N, 23°28′E, 85 m) in Finland. Eight of the plots at Toholampi were cultivated organically and eight using conventional methods, while two of the plots at Yöni were cultivated organically and two conventionally with the two remaining plots kept as uncultivated natural grass plots, where grass was not harvested, and fertilizers were not applied. These latter plots were considered to reflect background leaching level from cultivated (non-forested) clay soil. The uncultivated plots were kept free from trees and bushes, though it was rare for these plants to appear on the field.

The sand soil field of Toholampi is classified as fine sand soil (<10% clay, 4.2% organic carbon in 1987) and Gleyic Podsol (IUSS Working Group WRP, 2015; Yli-Halla et al., 2000). The fields have an average slope of 0.5% with the slope ranging from 0.3% to 0.7%. The sixteen plots measure 16 m × 100 m each and were fitted with separate collection of both surface runoff and subsurface drainage from each plot, the latter consisting of drain pipes laid at 16 m intervals at a depth of approximately 1 m, to allow for the measurement and analysis of both the surface runoff and the drainage water from each plot individually and throughout the year. The plots are hydrologically isolated from one another by mounted earth and plastic sheeting. Further details are provided by Salo and Turtola (2006) and Manninen et al. (2018). Four subsequent 4-year crop rotations of organic and conventional farming, mimicking practices of a dairy farm and a cereal farm have been performed on the field in 2001–2016. The four crop rotations (Table 1) have been cultivated in four replicates in blocks as part of a randomized experiment. Prior to the current study period, both organic crop rotations had been in transition to organic farming for four years. Compared to conventional crop rotations, the crop rotation of the organic dairy farm does not receive any mineral fertilizers and has been diversified by including nitrogen-fixing plants red clover (*Trifolium pratense* L.) and common vetch (*Vicia sativa* L.). The organic cycle of the cereal farm does not receive any mineral fertilizers either but has been diversified by one year of timothy (*Phleum pratense* L.) and red clover ley cultivation. During the 4-year crop rotation, the conventional cereal crop rotation plots were ploughed annually, organic cereal rotation plots were

Table 1
Crop rotations representing organic and conventional dairy farming (A) and organic and conventional cereal farming (B) on experimental plots on sandy soil at Toholampi in 2001–2016 as well as crop rotations representing organic and conventional farming on clay soil at Yöni in 2001–2016.

	Sand soil (Toholampi)				Clay soil (Yöni)
Year	Organic (A)	Conventional (A)	Organic (B)	Conventional (B)	Organic & Conventional
1	Barley + under-sown grass	Barley + under-sown grass	Barley + under-sown grass	Barley	Barley + under-sown timothy and clover seed
2	Grass (clover + timothy)	Grass (timothy, meadow fescue	Grass (clover + timothy), rye in the autumn	Barley, rye in the autumn	Grass-clover ley
3	Grass (clover + timothy)	Grass (timothy, meadow fescue)	Rye	Rye	Grass-clover ley
4	Mixture of oats and common vetch	Barley, whole crop silage	Oats	Oats	Rye
5					Oats and pea

ploughed three times, and both dairy farm rotations twice. A detailed description of crop rotations, fertilization and tillage is given in Peltoniemi et al. (2021). The Toholampi leaching field is shown in Fig. 1. Descriptive statistics of the variables from the site that have been used in the fitting of the developed model are presented in Table 2.

The clay soil field of Yöni can be classified as a heavy clay (>60% clay, 4.4% organic C) and a Vertic Stagnosol (IUSS Working Group WRB, 1998; Lilja et al., 2017). The field has an average slope of 0.4%. The experimental set up consists of six separately drained plots, about 0.5 ha (0.44–0.62 ha) each, with subsurface drainage and surface runoff collection systems to allow for the measurement and analysis of the sum of the drainage and surface runoff from each plot throughout the year. The drainage pipes were laid 16.5 m apart and at a depth of about 1 m in 1989. For both organic and conventional farming, similar 5-year crop rotations have been cultivated, however, in organic farming N was applied in the form of manure only (farmyard manure 1995–2011 and cow slurry 2012–2016) and in conventional farming in the form of mineral fertilizers. The N fertilization of the organic crop rotation was based on the yields achieved on the field before 1995, dairy cow feed requirements and the hypothetical subsequent total N excretion by the cows in feces and urine. Yields achieved on the field in 1990–1995 were sufficient to feed 0.5 dairy cows per year. Thus, total N excreted by 0.5 dairy cows (50 kg ha-1 y-1) was applied as manure to the field in a five-year crop rotation. Farmyard manure was applied twice and cow

Fig. 1. The experimental leaching field at Toholampi, Finland. Photograph taken by Jari Lindeman. Starting from the furthest plot (top of the figure), the plots 1,2,6,7,9,10,14, and 15 are organically farmed. The darker plots have a crop rotation mimicking a cereal production farm (half of these plots are organic) and the rest have a crop rotation akin to a dairy production farm.

Table 2
Summary statistics of the data used for fitting the models for the sand soil plots of Toholampi and the clay soil plots of Yöni. The temperature sum is the sum of the average daily temperatures during the year in degrees Celsius. The years here refer to hydrological years, e.g., the year 2001 is defined as June 1st, 2001–May 31st, 2002. Here IQR refers to the interquartile range.

Soil type	Variable	Mean	IQR	Range	Years
Sand soil (Toholampi)	Total N leached (kg ha^{-1} year^{-1})	9.62	9.19	(0.79, 30.18)	2001–2016
	NO_3–N in soil (kg ha^{-1})	8.66	8.72	(0.04, 51.34)	2001–2012
	N input in fertilizer (kg/ha)	70	31	(0, 180)	2001–2016
	Crop N yield (kg ha^{-1})	89	72	(18, 225)	2001–2016
	Average temperature (°C)	3.5	1.5	(1.1, 5.1)	2001–2016
	Rainfall (mm)	671	66	(504, 801)	2001–2016
Clay soil (Yöni)	Total N leached (kg/ha year)	11.56	9.65	(1.80, 44.68)	2002–2016
	NO_3–N in soil (kg/ha)	7.87	14.08	(0.05, 32.84)	2002–2007
	N Input in fertilizer (kg/ha)	48	80	(0, 198)	2002–2016
	Crop N yield (kg/ha)	36	60	(0, 171)	2002–2016
	Average temperature (°C)	5.2	1.9	(3.5, 6.5)	2002–2016
	Rainfall (mm)	616	127	(367, 781)	2002–2016

slurry three times during 5-year crop rotation. While thefarming practices were executed for the period 1995–2016, nutrient leaching measurements were not started until 2001. In both cultivation practices the 5-year rotation consisted of a spring barley (*Hordeum vulgare* L.) under-sown with timothy and clover seed, followed by 2-year grass-clover ley, followed by winter rye (*Secale cereale* L.), and finally a mixture of oats (*Avena sativa* L.) and pea (*Pisum sativum* L.). Both crop rotations were ploughed three times during the five-year rotation to a depth of 20 cm: in August before sowing winter rye and in October after harvesting a rye and a mixture of pea and oats. Further details are provided by Manninen et al. (2018). Descriptive statistics of the variables from the site that have been used in the fitting of the developed model are presented in Table 2.

2.1.2. Measurement and calculation of nitrogen leaching, soil nitrate-nitrogen and nitrogen balance

Total N concentrations were analyzed from the volume-based drainage and surface runoff samples of the experimental plots and multiplied with their respective runoff volumes and summarized for the hydrological year (June 1st – May 31st) to obtain annual total N leached (Table 2). Annual number of individual volume-based water samples varied in Toholampi from 13 to 30 samples and in Yöni from 21 to 107 samples. Water total N concentration was determined with Lachat autoanalyser from unfiltered water samples after oxidation of N compounds to NO_3–N in alkaline solution (ISO 11905–1). Soil NO_3–N measurements (calculated as kg ha^{-1}) from the depth of 0–60 cm were available in Toholampi for autumns 2001–2012 and in Yöni for autumns 2001–2007. Soil NO_3–N was determined after 2 M KCl extraction (1:2.5, w:w) using a Skalar autoanalyzer, and then calculated to give the content in the 60 cm soil layer. For the N balance calculations, N input in fertilizers was calculated using the total N input. N output as crop N yield was calculated from the harvested dry yield and its N concentration as determined by Kjeldahl method. Nitrogen balance was then calculated as

$$N_{balance} = N_{fert} + N_{biological} - N_{yield} \tag{1a}$$

where N_{fert} is the total amount of N applied either through a mineral fertilizer or manure (kg/ha), $N_{biological}$ is the N input by biological N fixation (kg/ha) calculated using the equations developed by Høgh-Jensen et al. (1998), and N_{yield} is the N yield (kg/ha). We also explored a model, where organic N from manure application from the previous year was included, but this did not change the results in a significant way.

2.2. Statistical methods

The average leaching and its 95% confidence intervals were calculated for the organic and conventional plots in both soils. For the clay soil plots average value was also calculated for the uncultivated natural grass plots which were not present in the sand soil data set.

For conventionally and organically farmed plots (i.e., with the uncultivated natural grass plots in the clay soil excluded) the correlations between N leaching with rainfall, average temperature, nitrate N in soil, N leaching, and N balance were explored in order to inform variable selection for prediction models. Based on insights from these comparisons, three alternative models for N leaching are presented:

$$N_i = \beta_0 + \beta_{rain}(R_i - 650) + \beta_{N\ bal.}N_{bal.,i} + \beta_{FT}FT_i + \beta_{Org}O_i + \epsilon_i \tag{1b}$$

$$N_i = \beta_0 + \beta_{rain}(R_i - 650) + \beta_{NO_3N}NO_3N_i + \beta_{FT}FT_i + \beta_{Org}O_i + \epsilon_i \tag{2}$$

$$\log(N_i) = \beta_0 + \beta_{rain}(\log(R_i) - \log(650)) + \beta_{NO_3N}\log(NO_3N_i) + \beta_{FT}FT_i + \beta_{Org}O_i + \epsilon_i \tag{3}$$

where N_i is the amount of nitrogen leached (kg ha^{-1} year^{-1}) for the plot i, R_i is the annual rainfall (mm) centered at 650 mm, $N_{bal.,i}$ is the nitrogen balance (kg ha^{-1}), NO_3N_i is the NO_3–N amount (kg ha^{-1}) measured in the fall from the soil, FT_i is a dummy variable for farm type, which is 1 for the cereal farm plots in Toholampi and 0 otherwise, O_i is 0 for conventional farming and 1 for organic farming, β s are the regression coefficients to be fitted, and ϵ_i is an independent and normally distributed error term. A linear model including both NO_3–N amount and N balance was also explored but based on Akaike Information Criterion (AIC) it performed worse than Model 2 in both data sets and was thus excluded. Interactions between organic farming and rainfall, or organic farming and NO_3–N, or organic farming and N balance were also not found to improve the model.

Model performance was assessed by a cross-validation procedure, where data from one year was held out as a test set at a time, while the model was trained with data from the other years (10, 11, or 15 depending on the data set). We report the RMSE of cross validation (RMSECV), as well as the rank-order correlation (Spearman correlation, r_s) of the predicted values and field measurements, as well as the pair correlation (Kendall correlation, r_k), which are reasonable metrics when comparing models with different variance assumptions. Furthermore, we also report the mean relative error (MRE), where relative error is defined as the ratio of absolute error to the true value. Given the large variation of the predicted values, this reflects the model performance well, since, e.g., an error of 5 kg ha^{-1} year^{-1} in leaching, when the true value is 1 kg ha^{-1} year^{-1}, is much more problematic than when the true value is 50 kg ha^{-1} year^{-1}.

All analyses were carried out using the statistical software R (R Core Team, 2022). The model fitting was done using the lm-function from base R.

3. Results

3.1. Nitrogen leached in organic and conventional farming

In the conventionally farmed sand soil, average N leaching of 10.7 kg ha^{-1} year^{-1} (95% C.I. (6.5, 15.0) kg ha^{-1} year^{-1}) was observed while for organic farming the leaching was on average 8.5 kg ha^{-1} year^{-1}

(95% C.I. (5.2, 11.8) kg ha^{-1} $year^{-1}$). For the clay soil the estimate for average leaching was 17.0 kg ha^{-1} $year^{-1}$ (95% C.I. (11.0, 23.0) kg ha^{-1} $year^{-1}$) with conventional farming, 11.6 kg ha^{-1} $year^{-1}$ (95% C.I. (7.3, 15.8) kg N ha^{-1} $year^{-1}$) with organic farming, and 6.2 kg ha^{-1} $year^{-1}$ (95% C.I. (4.8, 7.6) kg ha^{-1} $year^{-1}$) for uncultivated natural grass.

On the logarithmic scale, the difference between organic and conventional farming was −0.203 (95% C.I. (−0.419, 0.012)) for the sand soil and −0.296 (95% C.I. (−0.714, 0.120)) for the clay soil. By calculating the weighted average of these results, where inverse squared error is used as a weight, we find that organic farming resulted in a −20% (95% C.I. (−34%, −3%)) reduction in leached N per unit area.

3.2. Nitrogen balance and nitrate-nitrogen in soil

After a logarithmic transformation to mitigate heteroscedasticity, the N balance calculated as described above and the annual N leaching in the sand soil correlated at r = − 0.16 for conventional farming and at r = −0.28 for organic farming, and in the clay soil at r = 0.16 for conventional farming and at r = 0.11 for organic farming. Alternatively, N available for leaching can be estimated by measuring the NO_3–N in the soil. For NO_3–N measurements carried out in October the numbers are much more consistent with NO_3–N correlating with N leached in the sand soil at the level r = 0.18 for conventional farming and at r = 0.35 for organic farming, and in the clay soil at r = 0.85 for conventional farming and at r = 0.86 for organic farming.

3.3. Mean temperature and rainfall

In the sand soil of Toholampi the annual rainfall was highly correlated with annual N leaching with r = 0.77 on the logarithmic-logarithmic scale. Mean temperature correlated with N leaching at r = 0.56. Since mean temperature and rainfall correlated at r = 0.77 as well, we would expect the mean temperature to correlate at r = 0.77^2 = 0.59 if the mean temperature is not otherwise linked to N leaching than indirectly through the correlation with rainfall.

In the clay soil of Yöni, rainfall and N leaching were not correlated with r = 0.01, while mean temperature and N leaching had a weak negative correlation of r = − 0.25. Mean temperature and rainfall were also positively correlated in the Yöni data set with r = 0.54.

3.4. Prediction models

The model fits for developed Models 1–3 are presented in Tables 3–5, respectively. The predicted values plotted against the observed values are given in Fig. 2 for the sand soil plots and in Fig. 3 for the clay soil plots. The Model 1, i.e., a linear model with N balance has the worst performance with R^2 = 0.52 for the sand soil and R^2 = 0.09 for the clay soil. For sand soil, rainfall, organic farming, and farm type (dairy vs. crop), and N balance are statistically significant at the $p < 0.05$ level. More rainfall is associated with higher N leaching, while organic farming and dairy farming are associated with less N leaching compared to conventional farming and cereal farming, respectively. In the clay soil none of the covariates are statistically significant except the one for organic farming, which is associated with 5.17 kg ha^{-1} $year^{-1}$ less nitrogen leached compared to organic farming. Nitrogen balance is not statistically significant for the clay soil, while in the sand soil the association is negative, with 1 kg higher N balance being associated with 0.016 kg less nitrogen leached per hectare. The model also produces negative leaching values for some sand soil plots, which is impossible in reality and thus reflects poor model specification. The correlations between the predicted values and the field measurements are $r_s = 0.69$ and $r_k = 0.50$ for the sand soil, and $r_s = -0.23$ and $r_k = -0.18$ for the clay soil. The corresponding mean relative errors are 93% and 136%, respectively.

Model 2 performs better than Model 1 with R^2 = 0.62 for the sand soil and R^2 = 0.68 for the clay soil. While the results regarding other regression coefficients are similar to Model 1, the soil NO_3–N values are clearly statistically significant (p < 0.0001) for both soil types. For the sand soil the association is not as strong with 1 kg NO_3–N ha^{-1} being associated with an increase in annual N leaching of 0.22 kg N ha^{-1} $year^{-1}$, while in the clay soil the relationship is almost one-to-one with 1 kg NO_3–N ha^{-1} being associated with an increase in annual N leaching of 0.81 kg N ha^{-1} $year^{-1}$. While the uncertainties are smaller than for Model 1, the problem of negative predicted values in the sand soil plots still persists. The correlations between the predicted values and the field measurements are $r_s = 0.84$ and $r_k = 0.64$ for the sand soil, and $r_s = 0.47$ and $r_k = 0.32$ for the clay soil. The corresponding mean relative errors are 108% and 119%, respectively.

When looking at the results on the log-log scale, the fits of Model 3 appear to perform the best with $R^2 = 0.72$ for sand soil and $R^2 = 0.76$ for clay soil. However, these numbers are not directly comparable to those of Model 1 and 2, since the response variable is on a different scale and in differentunit.The correlations between the predicted values and the field measurements are $r_s = 0.88$ and $r_k = 0.69$ for sand soils, and $r_s = 0.77$ and $r_k = 0.55$ for clay soils. The corresponding mean relative errors are 67% and 90%, respectively. With the log-log model all predicted values, regardless of the input covariates, are always positive so that the issue of negative predicted values is not a problem with Model 3.

4. Discussion

The goal of this paper was to develop N leaching models which are sufficiently accurate to be used for estimating the annual leaching of individual plots managed using conventional or organic methods. The average N leaching in the cultivated plots ranged from 8.5 to 17.0 kg N ha^{-1} $year^{-1}$, which is in line with observations from the modeling study, which reported values ranging from 5 to 30 kg N ha^{-1} $year^{-1}$ averaging 17.4 kg N ha^{-1} $year^{-1}$ in 1990 and having decreased to 14.4 kg N ha^{-1} $year^{-1}$ by 2019 (Huttunen et al., 2023). These values are also consistent with those reported by Tattari et al. (2017) where the average leaching in Finnish fields since 1981 was estimated to have averaged 15.5 kg N ha^{-1} $year^{-1}$. The preferred models had mean relative errors of 67% and

Table 3
Regression coefficients and fit quality measures for Model 1. Rainfall has been centered so that the intercept value corresponds to a rainfall of 650 mm $year^{-1}$. For cross validation, the correlation between the predicted values and the field observations (r_s = Spearman correlation, r_k = Kendall correlation), RMSE from cross validation (RMSECV), and the mean relative error (MRE) are presented.

	Sand soil			Clay soil		
	Estimate	95%-C.I.	p-value	Estimate	95%-C.I.	p-value
β_0	9.08	(7.46, 10.70)	<0.0001	16.21	(12.27, 20.14)	<0.0001
β_{Rain}	0.066	(0.057, 0.074)	<0.0001	−0.004	(-0.028, 0.019)	0.71
β_{Org}	−2.12	(-3.36, −0.88)	0.0009	−5.17	(-10.08, −0.26)	0.04
$\beta_{N\ bal.}$	−0.016	(-0.028, −0.004)	0.01	0.02	(-0.025, 0.06)	0.47
β_{FT}	2.54	(1.05, 5.03)	<0.0001			
Fit quality	$R^2 = 0.52$	$RMSE = 5.02$		$R^2 = 0.09$	$RMSE = 9.23$	
Cross validation	$r_s = 0.69$	$RMSECV = 5.57$		$r_s = -0.23$	$RMSECV = 10.7$	
	$r_k = 0.50$	$MRE = 93\%$		$r_k = -0.18$	$MRE = 136\%$	

Table 4
Regression coefficients and fit quality measures for Model 2. Rainfall has been centered so that the intercept value corresponds to a rainfall of 650 mm year^{-1}. Cross validation results are listed as in Table 3.

	Sand soil			Clay soil		
	Estimate	95%-C.I.	p-value	Estimate	95%-C.I.	p-value
β_0	6.63	(5.36, 7.90)	<0.0001	8.65	(3.42, 13.88)	0.003
β_{Rain}	0.070	(0.061, 0.078)	<0.0001	0.02	(-0.01, 0.04)	0.15
β_{Org}	−1.77	(-3.05, −0.48)	0.007	−2.26	(-8.18, 3.66)	0.44
$\beta_{NO_3N.}$	0.22	(0.13, 0.30)	<0.0001	0.81	(0.54, 1.09)	<0.0001
β_{FT}	1.54	(0.24, 2.85)	0.02			
Fit quality	$R^2 = 0.62$	$RMSE = 4.34$		$R^2 = 0.68$	$RMSE = 6.91$	
Cross validation	$r_s = 0.84$	$RMSECV = 5.41$		$r_s = 0.47$	$RMSECV = 16.7$	
	$r_k = 0.64$	$MRE = 108\%$		$r_k = 0.32$	$MRE = 119\%$	

Table 5
Regression coefficients and fit quality measures for Model 3. Rainfall has been centered so that the intercept value corresponds to a rainfall of 650 mm year^{-1}. For fit quality the results are presented on the log-log-scale (log) and after a back transformation to original units (BT). Cross validation results are listed as in Table 3.

	Sand soil			Clay soil		
	Estimate	95%-C.I.	p-value	Estimate	95%-C.I.	p-value
β_0	1.47	(1.31, 1.64)	<0.0001	2.19	(1.88, 2.49)	<0.0001
β_{Rain}	6.59	(5.96, 7.23)	<0.0001	0.75	(-0.16, 1.66)	0.10
β_{Org}	−0.17	(-0.31, −0.02)	0.03	−0.09	(-0.48, 0.30)	0.63
$\beta_{NO_3N.}$	0.19	(0.11, 0.26)	<0.0001	0.37	(0.27, 0.47)	<0.0001
β_{FT}	0.15	(0.00, 0.30)	0.06	–	–	–
Fit quality	$R^2 = 0.72(log)/0.50(BT)$	$RMSE = 0.50(log)/5.04(BT)$		$R^2 = 0.76(log)/0.62(BT)$	$RMSE = 0.46(log)/6.82(BT)$	
Cross validation	$r_s = 0.88$	$RMSECV = 7.34$		$r_s = 0.77$	$RMSECV = 19.7$	
	$r_k = 0.69$	$MRE = 67\%$		$r_k = 0.55$	$MRE = 90\%$	

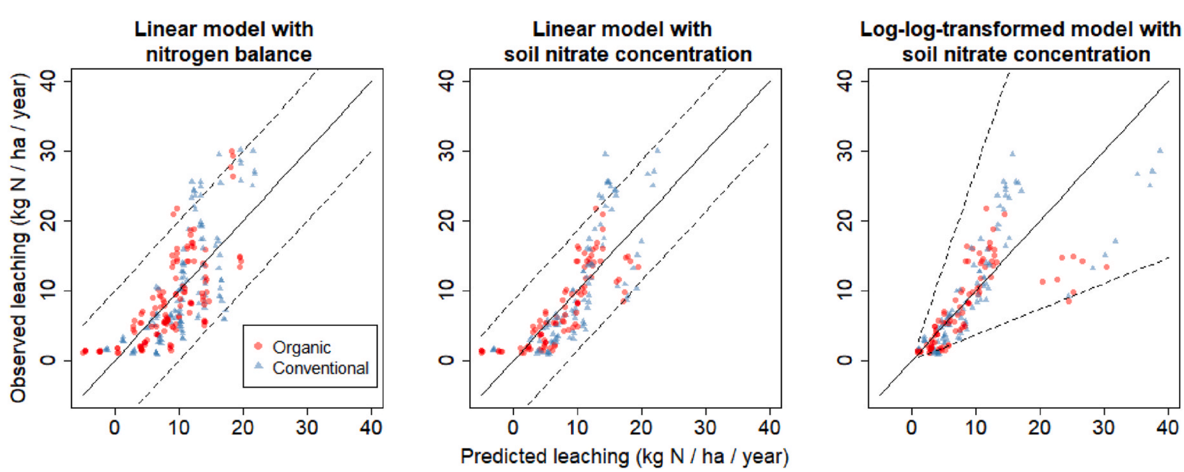

Fig. 2. Predicted and observed N leaching (kg ha^{-1} year^{-1}) with Model 1 (left), Model 2 (middle), and Model 3 (right) for the sand soil plots of Toholampi. The solid line represents a perfect fit, and the dashed lines indicate the 95% C.I. of the model. The organic and conventional plots are denoted by the shape and color.

90% for the sand soil and the clay soil, respectively. While these errors may seem large, it should be noted that in the sand soil the ratio of the largest and smallest leaching varied between 12 and 36 within the same plots, and between 5 and 23 in the clay soil. In other words, the same plot with the same treatment (organic/conventional, same crop rotation) could have 3600% higher leaching in one year compared to another, due to factor such as the particular crop being cultivated in a given year and rainfall. Thus, the prediction errors are small in comparison. This huge variation also underlines the fact that long time series are needed in order to give reliable assessments regarding N leaching. The quality of the model in this case is easier to understand in terms of pair correlations, which were $r_k = 0.69$ and $r_k = 0.55$. This means that if plot A has a larger annual leaching than plot B, the probability that the model predicts the order of these two plots correctly is roughly 85% for the sand soil, and 78% for the clay soil.

In the sand soil plots located in Toholampi, rainfall was the most important predictor of annual N leaching, while in the clay soil plots of Yöni rainfall did not correlate with annual leaching. High correlations between precipitation and N leaching have been previously reported in, e.g., Eltun and Fugleberg (1996). Nitrogen balance was not a statistically significant predictor of N leaching in Yöni, while there was a small statistically significant negative association in Toholampi. This is in line with the observation of Salo and Turtola (2006) where models using N balance as a predictor were not successful at predicting annual leaching in conventional farming, though predictive performance was better, when average leaching over longer periods of 4–10 years were considered. The analysis of Salo and Turtola (2006) utilized the conventionally farmed plots from Toholampi though only data up to 2001 was then used.

When averaging over several years, N balance (Δ N) has been previously found to be associated with N leaching. However, the strength of this association and even its direction vary widely in the literature. For

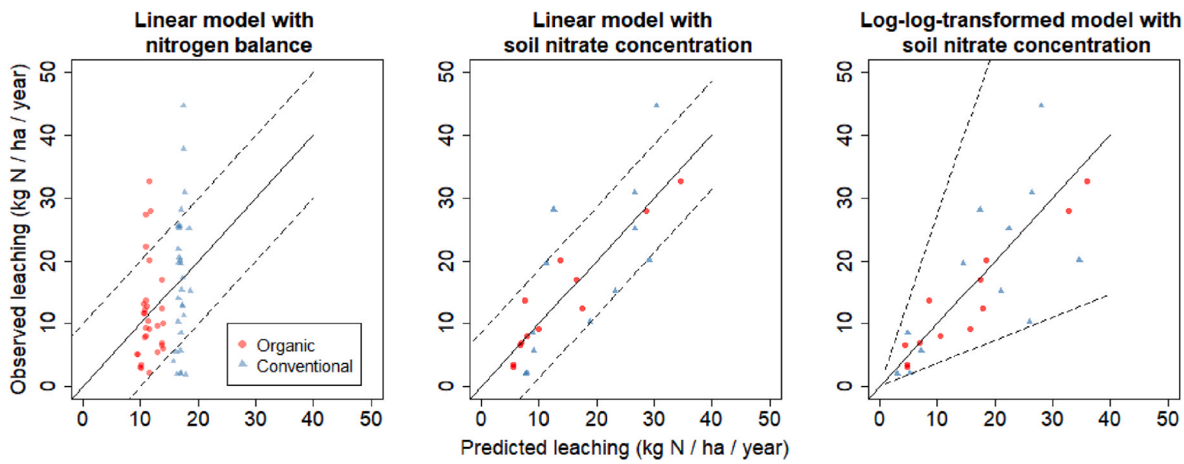

Fig. 3. Predicted and observed N leaching (kg ha^{-1} $year^{-1}$) with Model 1 (left), Model 2 (middle), and Model 3 (right) for the clay soil plots in Yöni. The solid line represents a perfect fit, and the dashed lines indicate the 95% C.I. of the model. The organic and conventional plots are denoted by the shape and color.

example, Uhlen (1989) gives an equation for annual leaching of 39 + 0.27 Δ N, Webb et al. (2000) 16 + 0.46 Δ N, Korsaeth and Eltun (2000) 31 + 0.15 Δ N, while Salo and Turtola (2006) give three equations: 13 + 0.57 Δ N (bare fallow, clay), 12–0.11 Δ N (green fallow, clay), and 5 + 0.20 Δ N (grass, sand). Just by considering the case where Δ N = 0 one can observe that the estimates vary widely, and the effect of the N balance varies greatly in magnitude. The wide variation in the models utilizing N balance points to the fact that such models cannot be reliably applied outside of the data, where they were fitted. The variability in these results may be due to length of the measurement periods of 3–8 years that may be too short for reliable estimates when there is huge annual variation. We also explored including organic N from the previous year's manure applications as a predictor in the model (see, e.g., Frick et al., 2022; Fuchs et al., 2023), but this did not result in improved model fits; the negative effect of N balance was actually slightly larger in Toholampi with this included.

Calculating the N balance is not trivial as it requires quantitative knowledge of all the yield components and their N concentrations. Thus, models using NO_3–N measurements of soil are not necessarily more difficult to use in practice, though they require soil sampling and chemical analyses. In Finland, for example, this might well be possible in practice, because springtime mineral N (NO_3–N and NH_4-N) measurement has previously been optional for farms that have joined the voluntary environmental support system for assessing the need for N fertilization which is part of the EU's common agriculture policy (CAP). Currently this measure is recommended but not subsidized. In this work we have shown that a model utilizing NO3–N works with a variety of different types of crops, meaning that it can be reliably applied to diversified crop rotations, which are becoming increasingly more common at the expense of monoculture farming.

Having reliable estimates for the effects of adopting organic farming or in other ways reducing the amount of fertilizer applied could provide a practical incentive for the farmer to alter cultivation practices in order to reduce leaching, and thus participate in achieving environmental goals while simultaneously mitigating the economic costs of N fertilizer. However, farmers often lack this information. In the future, it would be advisable for farmers to measure soil NO3–N levels regularly and use this information to predict the leaching risk.

Based on our analysis, organic farming as practiced in this experiment resulted in 20% less N leached per hectare as compared to conventional farming with 95% C.I. [−34%, −3%]. However, lower leaching per hectare does not necessarily mean that organic farming results in less N leaching per kilogram of yield as has been noticed in the LCA literature previously (Tuomisto et al., 2012). On the other hand, this emphasizes the fact that increasing yields is important in organic farming, though achieving this solely by increased fertilization may be counterproductive from N leaching point of view. In the end, there probably remains a difference in the eutrophication impact of organic and conventional farming, though the difference may not be very large. Further research with different cultivation practices, crop species, and climates are needed before generalizable conclusions can be drawn. However, the lower per hectare leaching from organic cultivation implies the potential of organic farming in solving localized eutrophication problems.

5. Conclusion

In site-specific LCA-based comparisons of organic and conventional products, it is important to use reliable models to estimate the eutrophication potential and to support decision making regarding choice of organic farming practices vs. conventional ones. Our study shows that it is possible to develop relatively simple yet accurate models for predicting N leaching in organic and conventional farming.

The relevant factors explaining N leaching include soil type, rainfall, mean temperature, organic farming, amount of N fertilizer applied, and the crop N yield. If soil NO3–N measurements from the fall are available, the models can explain most of the variation in annual N leaching, with $R^2 = 0.72$–0.76 on the log-log scale, depending on the soil type. Nitrogen balance, in turn, was not found to be a statistically significant predictor of N leaching in the data set including crop rotations with annual and perennial crops and legumes.

Authors' contributions

The initial conception of the study for the model development was from Merja Saarinen and the design of formal analysis from Joel Kostensalo. All authors contributed to the study design. Material preparation and data collection were done by Riitta Lemola and Tapio Salo. The setup of the field experiments was designed by Eila Turtola and later managed by Riitta Lemola. Formal analysis and modeling were performed by Joel Kostensalo. The first draft of the manuscript was written by Joel Kostensalo, Liisa Ukonmaanaho, and Merja Saarinen. All authors commented on previous versions of the manuscript. All authors participated in the editing of the manuscript, as well as read and approved the final manuscript.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests:

financial support was provided by Business Finland.

Data availability

The authors do not have permission to share data.

Acknowledgements

The research leading to these results received funding from Business Finland under Grant Agreement No 8685/31/2021.

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