ANALYSIS OF AGRICULTURAL RUN-OFF MONITORING PROGRAM RESULTS FOR ESTIMATION OF NITROUS OXIDE INDIRECT EMISSIONS IN LATVIA

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Abstract. Indirect emissions of nitrous oxide (N_2O) due to nitrogen leaching and runoff refer to agricultural soils and manure management. One of the most important parameters to calculate indirect nitrous oxide emissions due to leaching and runoff from all nitrogen added to managed soils is represented as fraction of nitrogen that is lost through leaching and runoff or $Frac_{LEACH-(H)}$. Default $Frac_{LEACH-(H)}$ value is 0.3 (kg N) according to the guidelines for the greenhouse gas emission inventory developed by the Intergovernmental Panel on Climate Change (IPCC). This approach of default $Frac_{LEACH-(H)}$ value is generalized in order to allow widespread applicability. However, agricultural practice, climatic conditions, soil properties, hydrological flow processes and other factors play an important role in determining these losses. The potential of emission estimate improvement on national level by revision of $Frac_{LEACH-(H)}$ is based on local data availability. However, there are often limited measurement data on leaching and runoff losses from various soils and agriculture practice. Therefore, agricultural run-off monitoring results related to diffuse pollution were analyzed with the aim to evaluate the nitrogen losses fraction outcome in relation to nitrogen inputs. The results of the research show that $Frac_{LEACH-(H)}$ in conditions of moderately intensive agriculture for croplands ranges close to 0.23 against the default value currently used in Latvia to estimate indirect nitrous oxide emissions from the agriculture sector.

Keywords: nitrous oxide, indirect emissions, agriculture.

Introduction

Nitrous oxide (N_2O) emissions from managed agricultural soils are formed naturally through the microbial processes of nitrification and denitrification. Emissions are related to the soil type, moisture, temperature, organic matter content, fertilizer type, consequently showing large variations [1]. Agricultural activities like use of mineral fertilizers, animal manure, sewage sludge or other organic additions in soils increase the amount of nitrogen available for N_2O emissions. According to the Intergovernmental Panel on Climate Change (IPCC) guidelines of the greenhouse gas emissions (GHG) inventory, emissions of N_2O that result from nitrogen inputs to soils occur through two pathways. Direct pathway includes N_2O emission directly from the soils. N_2O emission through nitrogen leaching and runoff, as well as through volatilization (NH_3, NOx) and subsequent redeposition refer to indirect pathways [2].

Almost all N_2O produced in agricultural topsoil is emitted into the atmosphere as direct N_2O emission. However, indirect N_2O emission associated with leaching and runoff also plays an important role in determining GHG emissions. According to the IPCC methodology, emissions from leaching and runoff are calculated from leaching-runoff fraction of nitrogen ($Frac_{LEACH-(H)}$) that represents the share of nitrogen losses compared to total applied nitrogen. Surplus of nitrogen or excessive rainfall promote nitrogen lost through leaching and surface runoff, producing high N_2O concentrations in the subsurface. N_2O emission occurs after the discharge to surface water [3]. General estimation of indirect N_2O emission from nitrogen leaching and runoff in agro-ecosystems is represented as follows (1):

$$N_2O_{(L)} = N_{input} \cdot Frac_{LEACH-(H)} \cdot EF_5, \tag{1}$$

where $N_2O_{(L)}$ – amount of N_2O emission by leaching and runoff due to nitrogen addition to the soil, kg $N_2O \cdot yr^{-1}$;
N_{input} – amount of nitrogen added to and mineralized in the soil, $kg \cdot N \cdot yr^{-1}$;
$Frac_{LEACH-(H)}$ – fraction of all nitrogen added to and mineralized in the soil that is lost through leaching and runoff, kg N;
EF_5 – emission factor for N_2O emission from nitrogen leaching and runoff, kg N_2O-N, $kg^{-1} \cdot N$.

According to the IPCC methodology nitrogen inputs include: mineral fertilizer, manure, sewage sludge and other organic compounds, urine and dung deposited by grazing animals, crop residues and mineralized nitrogen in mineral soils associated with loss of soil carbon from soil organic matter [2].

The default approach of IPCC assumes that 30 percent (in the range 10-80 %, depending on agricultural practice) of the nitrogen applied to soils is lost in humid regions when rainfall is greater than the half of potential evaporation, [2; 4]. This value largely is based on mass balance studies comparing agricultural nitrogen inputs to nitrogen recovered in rivers. Rainfall causes part of the reactive nitrogen to leach into groundwater or be washed away to drainage ditches. Surface runoff is promoted when rainfall exceeds a maximum infiltration level of the soil [5]. Also manure storage facilities, if not designed or managed properly, can be potential sources of nitrogen leaching, but this source of nitrogen leaching is accounted by different approach [2]. The total export of nitrogen from agricultural ecosystems to water, as a percentage of fertilizer inputs, may range from 10 to 40 % from loam and clay soils to 25-80 % for sandy soils [6].

Most of European countries use the default $Frac_{LEACH-(H)}$ value for national GHG inventories. However, many studies on national level demonstrate lower values of $Frac_{LEACH-(H)}$ and suggest that the IPCC default value may be too high for most situations [7]. Consequently, same countries already have established lower country specific values of $Frac_{LEACH-(H)}$ since research studies. Figure 1 shows the values of $Frac_{LEACH-(H)}$ implemented in national GHG inventories of European countries, based on submission year 2014 information from the common reporting format (CRF) tables.

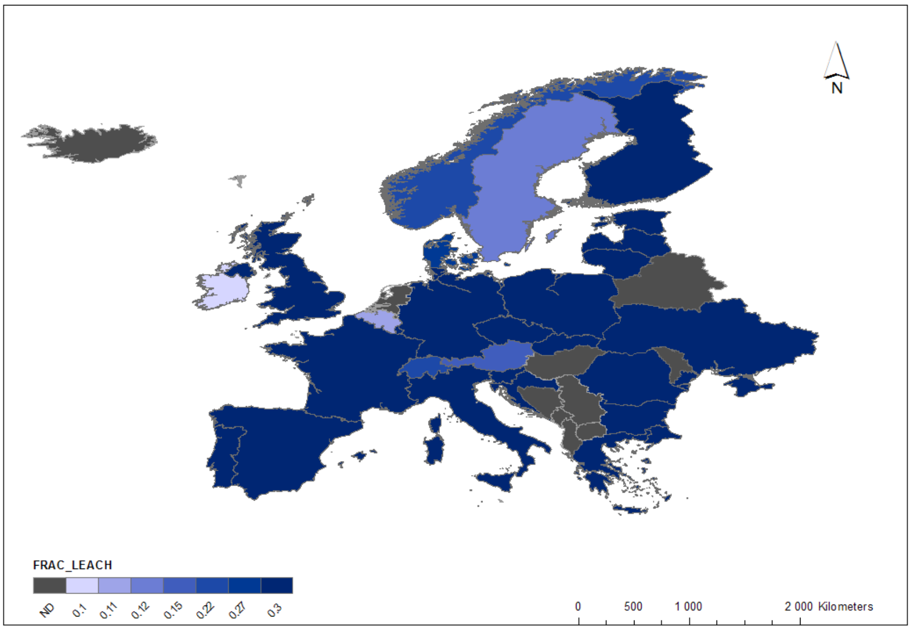

Fig. 1. **$Frac_{LEACH-(H)}$ values implemented in European countries GHG inventory, 2014**

Measures to decrease nitrogen leaching, such as balanced fertilization, have the potential of creating synergistic effects to decrease N_2O emissions. In the European Union (EU) the Nitrates Directive (ND) is one of the main policies that aim to reduce nitrate leaching from agriculture mainly due to the lower nitrogen inputs by fertilizers and manures. The ND also requires to implement measures on periods when the land application of fertilizers is prohibited and limitations to application of fertilizers on sloping soils, during wet conditions, and near watercourses [8]. Nitrate nitrogen (NO_3-N) is the form of nitrogen most likely to be lost from the crop root zone, either to tile drainage and groundwater. All forms of nitrogen added to the soil eventually are converted to NO_3-N [9; 10].

Requirements of the ND are implemented in several legislation documents in Latvia. However, due to agricultural intensification, a rapid growth of mineral fertilizers is observed in the last years. For example, the use of nitrogen with mineral fertilizers increased by 78.0 % 2005-2014, while the sown area increased by 15.0 % for the same period. It is projected that nitrogen inputs to the soil will continue to increase, leading to increased N_2O emissions.

There is a reason to believe that indirect N_2O emissions in Latvia also could be over calculated. The purpose of this paper therefore is to calculate values of $Frac_{LEACH-(H)}$ based on available data to promote the revision of $Frac_{LEACH-(H)}$ used in the national GHG inventory to provide more actual fraction of nitrogen loss for representing indirect soil emissions in Latvia.

Materials and methods

Latvia is situated in a humid and moderately mild climatic region where rainfall exceeds evaporation, resulting in percolation losses from the soil during spring and autumn. An assessment of diffuse agricultural pollution in Latvia has been implemented in three monitoring stations located in small catchments (river basins less than 10 km^2). *Mellupīte* is one of the monitoring stations established to analyze catchment and drainage field nitrogen balances and nitrogen loss in agriculturally influenced watersheds.

Since 1997, *Mellupīte* operates as complete agricultural run-off monitoring station that allows to analyze the nutrient loss by different agricultural practices, including nitrogen application rates. *Mellupīte* catchment represents moderately intensive farming conditions and could be considered as typical for present agriculture in Latvia. The main soil texture is loam with pH 6.7-7.0 in the catchment [11].

In order to estimate the impact of different fertilization treatments on the water quality, within the *Mellupīte* catchment 16 experimental plots with a total area of 2 ha drained with tile drains (depth 1.1-1.2 m, spacing between drains 11-12 m) are located in the territory of the farm *Kaudzītes*. Each plot size is 0.12 ha (30 x 40 m) and drainage system blocks are separated by contour drains [11; 12]. Five treatments in three replicates are applied for experimental plots design, including normal mineral fertilizer rate, high mineral fertilizing rate, manure and slurry applications, as well as, unfertilized plots [12]. The main cultivated crops in the rotation include winter wheat (*Triticum aestivum L.*), spring barley (*Hordeum vulgare L.*) and winter rape (*Brassica napus L.*). In order to evaluate what share of the used nitrogen is leached; the agricultural run-off monitoring data 1998-2014 are used for the study. Detailed information of the plots is given in Table 1.

Table 1

Characterization of plots within catchment of diffuse agricultural pollution monitoring station *Mellupite* (1998-2014)

Plots	Type of fertilization	Average yearly rate of nitrogen application, kg·ha^{-1}
Nr. 2;6;11	unfertilized	0
Nr. 1;8;13	mineral fertilizer	84
Nr. 3;9;14	manure	78
Nr. 4;7;12	slurry	78
Nr. 5;10;15	mineral fertilizer	130
Nr. 16	mineral fertilizer	130

The runoff measurements and water sampling are carried out in drainage field outlets. Data logger measures the water level on the weir and calculates discharge continuously and after certain amount of water bypass sends a signal to the water sampling pump. Based on a flow proportional sampling procedure, monthly composite water samples are collected to analyse the nitrogen concentrations in the samples or calculation of nutrient leakage.

In order to calculate the nutrient run-off for the certain period, the daily nutrient concentrations are multiplied by discharges measured automatically by the data logger (2):

$$N_{run-off} = \sum_{i=1}^{n} C_i Q_i \tag{2}$$

where $N_{run-off}$ – nitrogen run-off, kg·ha^{-1}·year^{-1};
C_i – average daily concentrations, mg·l^{-1};
Q_i –average daily water discharge, mm.

For determination of nitrogen losses in relation to input by mineral or organic fertilizers, total estimated nitrogen run-off should be corrected for other possible sources of nitrogen input (3).

$$N_{loss} = N_{run-off} - N_f \tag{3}$$

where N_{loss} – nitrogen loss from fertilizer and manure application, kg·ha^{-1}·year^{-1};
$N_{run\text{-}off}$ – total nitrogen run-off, kg ha^{-1} year^{-1};
N_f – background nitrogen loss, without fertilizer and manure application, kg·ha^{-1}·year^{-1}

Results and discussion

Nitrogen run-off from the plots applied with mineral fertilizer, manure and slurry is presented in Figure 2. The plots with slurry caused average nitrogen leaching of 10 % higher (20.5 kg·ha^{-1}·yr^{-1}) than leaching from the plot with mineral fertilizer (18.4 kg·ha^{-1}·yr^{-1}) in the study period. The monitoring results demonstrated significant influence of water discharge on nutrient run-off. Precipitation rates determined continuous discharge amount ($r = 0.766$, p-value < 0.05). The results also indicated high variability of leakage within the study period. The highest variability demonstrated the plots with mineral fertilization ($V_{(variation\ coefficient)} = 45$ %), relatively lover variability referred to the plots with slurry application (V = 37 %).

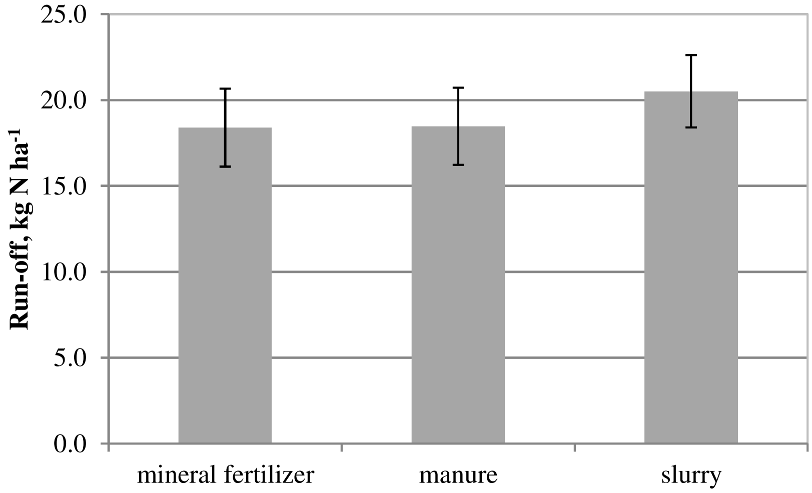

Fig. 2. **Calculated average run-off from plots with different fertilization practice, 1998-2014**

Estimate results of nitrogen loss fraction of total nitrogen input are summarized in Figure 3. The average $Frac_{LEACH\text{-}(H)}$ was calculated by dividing the runoff nitrogen with the total nitrogen inputs. Nitrogen leaching losses also varied widely among the plots and years. Predicted leaching loses represented 5-70 % of nitrogen inputs on plot basis. The research results have shown that, despite of nitrogen concentrations in drainage water and high run-off rate for the plots with mineral fertilizer application, the relevant fraction of nitrogen loss was observed as evidently lower, comparing to nitrogen losses in the plots with manure and slurry application. This could be explained by nitrogen availability for plant uptake. Nitrogen from mineral fertilizer, especially when applied as nitrate, is immediately available for plants and thus prevents leaching.

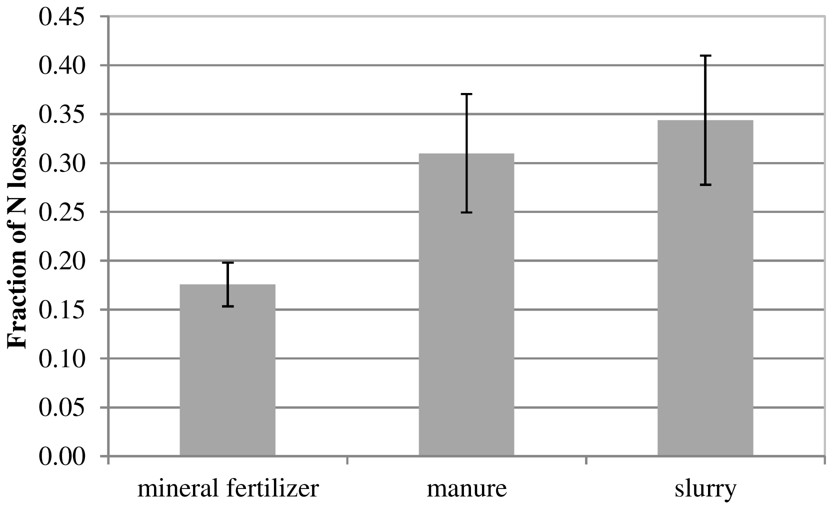

Fig. 3. **Estimated average fraction of nitrogen losses, 1998-2014**

Consequently, the results of N_2O emissions calculation from mineral fertilizer application, based on IPCC default and average observed $Frac_{LEACH-(H)}$, lead to important reduction of indirect emission amount. Figure 4 explains that indirect emission is about 41 % lower if the IPCC default $Frac_{LEACH-(H)}$=0.3 is replaced by average observed $Frac_{LEACH-(H)}$ = 0.18. However, it should be admitted that the calculated fraction of nitrogen losses refers only to mineral fertilizer inputs in the plots. At the field level also other nitrogen inputs should be evaluated, including crop residues. The estimation shows that on the drainage field level in *Mellupīte* monitoring station combined fertilization practice will result in $Frac_{LEACH-(H)}$ = 0.23.

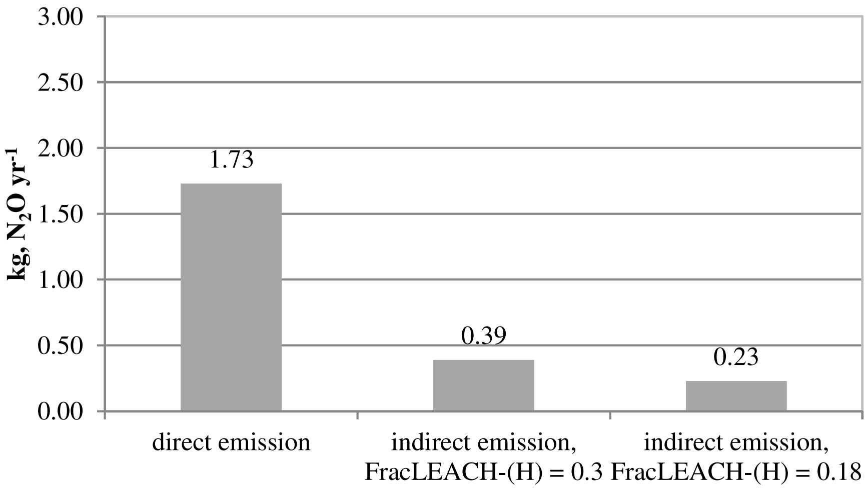

Fig. 4. **N_2O emissions from plots with average mineral fertilizer application rate**

Studies in Austria for both grassland and arable land plots showed smaller values of $Frac_{LEACH-(H)}$ than the default value [7]. For grassland, $Frac_{LEACH-(H)}$ values of 0.02 were found in Austria which varied very little over the entire observation period. For arable sites, $Frac_{LEACH-(H)}$ values were higher (around 0.25) and similarly showed significant variability between years due to variations in crop rotation, fertilization rates, and yields. Similar results of the $Frac_{LEACH-(H)}$ value in grasslands were shown by studies in the Netherlands, it can be explained that the denitrification capacity of grassland is higher than of arable land due to the higher organic matter contents of grasslands, by which the $Frac_{LEACH-(H)}$ of grassland is significantly lower than of arable land [13; 14].

Summarizing the results of the study tended to describe moderately intensive agricultural production influence on nitrogen loss, the $Frac_{LEACH-(H)}$ obtained from *Mellupīte* monitoring station drainage field level is smaller than the default IPCC value. This value could be even smaller at the catchment level, taking into account nitrogen losses not only from arable land, but also from grassland. However, for application on state level GHG inventory $Frac_{LEACH-(H)}$ should be evaluated in details by compressive study of nitrogen looses in conditions of intensive and extensive agricultural production and its distribution characteristics.

Conclusions

1. The results of agricultural run-off monitoring in moderately intensive agricultural production conditions show potential to reduce the default $Frac_{LEACH-(H)}$ value implemented in the national inventory for quantification of indirect N_2O emissions from managed soils in Latvia. Determined $Frac_{LEACH-(H)}$ by the study is 0.23 against default 0.30.
2. A high nitrogen application rate does not result in higher leaching if the nitrogen uptake is also high. It is recommended for Latvia to continue activities for country specific $Frac_{LEACH-(H)}$ development, because the IPCC default methodology does not account for differences in crop nitrogen uptake.
3. It is preferable to update the $Frac_{LEACH-(H)}$ for the GHG inventory each year, because the input parameters for $Frac_{LEACH-(H)}$ determination are highly variable.

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