Chapter 11

Water Quality of Agricultural Drainage Systems in the Czech Republic — Options for Its Improvement

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Additional information is available at the end of the chapter

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1. Introduction

Worldwide, artificial agricultural drainage systems create the optimal conditions for crop planting and soil cultivation by removing of excess water from the root zone and providing better trafficability for farm machinery. Although they are benefitial for agricultural production, the extensively used drainage systems considerably affect the hydrological and hydrochemical regimes of catchments in both positive and in negative ways. The runoff characteristics are some of the most affected catchment parameters: differences in runoff volumes and temporal variations of flow rates in the water courses, the lowered groundwater levels and changes in the surface energy balance have frequently been found [1-4]. Problems such as altered hydrological patterns and impaired drainage water quality in agricultural catchments have often been mentioned. Water quality from agricultural drainage systems (both tiles and ditches) has been discussed by the studies which draw attention to the reduced quality of drainage waters caused by elevated concentrations of nutrients (N, P, C) and/or pesticides. The results of direct monitoring of drainage groups or very small drained catchments [5-11] together with various model approaches have proven that the contribution of agricultural drainage to water pollution in larger areas may be significant [12-15]. In principle, land drainage increases the aeration of the soil profile and thus promotes the mineralization of soil organic matter and reduces denitrification in previously waterlogged soils. Further, the systems are often connected to soil preferential flow paths and so contribute to a rapid movement of water and soluble or particle-bound contaminats to related water bodies [6,10,11,13].

In the Czech Republic (CR), there were more than 1 078 000 ha of land drainage built by 1990, which cover about ¼ of the agricultural land in the CR [16]. Many of these systems are located in slopes. Tile drainage systems, built in slopy areas underlaid by crystalline bedrocks, receive in most cases not only the water infiltrating directly from upper soil horizons, but also water rising up from the groundwater table. In these cases tile drainage is often connected to a spring, a shallow aquifer or to a groundwater body, with a broader contributive area – a drainage subcatchment, see e.g. [12,17]. In such cases, this entire drainage subcatchment must be taken into an account for water balance studies, since a considerable portion of drainage runoff originates outside the drained area [17-19]. In the conditions of hilly landscape and soil environment located upon the rocks of crystalline complex, approx. 40% of the total catchment discharge happens as subsurface flow and 30% as groundwater flow [18]. Within a drainage subcatchment, different catchment zones exhibit different hydrological and hydrochemical roles and responses, influenced predominantly by soil conditions and land use and management rather than by the technical characteristics of the tile drainage-drainage spacing and lodgment, drain diameter, etc. [20,21].

Generally, from a hydrogeological viewpoint, a catchment can be divided into recharge zones, where precipitation infiltrates quickly and then recharges the groundwater body, and discharge zones, where groundwater approaches the land surface or a surface water body [22]. The recharge zones are mainly located in the uppermost areas of the catchment, close to the catchment divide, peaks and ridges. The soils of these zones are typically shallow and stony, with high sand content and high infiltration capacity. The coarse-textured soils of the recharge zones are, with respect to groundwater resources, well suited to growing grass, which, besides benefiting water quality, increases their field capacity and results in virtually complete infiltration of precipitation, including rainstorms [2,18,23]. The recharge zones are assumed to be the most important agent in drainage runoff and water quality formation. Discharge zones can usually be found in the lowest parts of the slopes (also along water courses, lakes) and are prone to surface waterlogging. The dominant soils in the discharge zones are generally deep, with higher clay content and a lower capacity for infiltration. The recharge zones and discharge zones are connected by transient zones, where precipitation is mostly transformed into surface runoff and groundwater flows downslope in a quasi-steady way [18,24]. The transient zones are located mainly in the middle sections of slopes. Groundwater in natural catchments flows from the recharge zones to the discharge zones. The actual spatial distribution of these zones depends on the local geologic and geomorphologic conditions [25, 26, © [2011] - 27]. The aforementioned zones, being described as recharge zones, are delineated by various methods and techniques [28-33]. However, the most common is the *DRASTIC* method [34]. Other methods are e.g. *SINTACS*, which is a modified Drastic approach published by [35], or the *PI* [36] and *COP* [37] methods. In the Czech Republic, the system of soil vulnerability evaluation based on the soil classification system was developed by [38], and later improved by [39], who transformed it into the Method of Critical Source Areas Identification for the Evaluation of Agricultural Non-Point Pollution Sources. Another useful tool is the Synthetic Map of Groundwater Vulnerability Assessment published by [40].

Tile drainage systems represent a shortcut between recharge and discharge zones which significantly reduces catchment water residence time [11,41], hastens the precipitation-runoff reaction, and shortens the time to reach peak discharge during events [4]. By ploughing shallow soils of recharge zones, nitrate concentration increase in soil water, especially during the non-growing season, due to the elevated mineralization of soil organic matter, encouraged by the aeration of soil profile induced by tilling. Subsequent enhanced leaching of nitrates occurs and lasts as long as the drainage system functions [14,42-43]. These facts have led to the assumption, that land use and agricultural management of areas prone to rapid infiltration profoundly affect nitrate concentration in surface as well as groundwaters [31,44-46].

In this chapter, results from two experimental studies are described. These two studies document linkages between land use within certain geomorphological enclaves of a drainage subcatchment and drainage water quality. Both studies were focused on nitrate nitrogen, but in general, they represent a generalised evidence of drainage runoff formation and its influence on water quality. Case study *A* was done on twenty-two tile drainage systems and their subcatchments. Case study *B* was realized on a very small 58 ha tile drained catchment Dehtáře. While case study *A* focused on monitoring the current status of land use within various drainage subcatchment zones and its relationships to drainage water quality, case study *B* was aimed at veryfying the effects of grassing in a catchment *recharge zone* on nitrate concentrations and loads in drainage waters. The main goal of both studies was to get a practical evidence for findings obtained in the CR and abroad by statistical approaches [23, 30, 44, 47] concerning the profoundly mitigative effects that grassing certain catchment areas has on the nitrate burden in drainage and surface waters.

2. Case study *A*

2.1. Materials and methods

2.1.1. Study areas

Case study *A* was carried out on 22 tile drainage systems, mostly built in slopes, which were monitored for water quality ($N\text{-}NO_3$) and related discharge once a fortnight for three years, 2004 – 2006. The plots studied were located in two regions of the Czech Republic (Figure 1); in the Svihov drinking water reservoir basin on the Zelivka river, and in the Southern Bohemian foothills of the Sumava mountains. The drainage systems which were selected and monitored were supposed to be functioning and not be connected to a surface water course or a pond / sewerage / wastewater outlet. All the examined drainage systems and related subcatchments were situated in crystalline complex with granite or paragneiss as parent rocks. The typical soil types were sandyloam to loamy Cambisols and loam to clayloam gleyic Stagnosols (Planosols).

For each examined drainage system, a design document (detailed construction plan) 1:1000 – 1:2000 scale was obtained and georeferrenced (rectified) to specify the exact properties and the positioning of the drainage pipes as well as to determine how the drained land and the related contributive area (subcatchment) were interconnected, using GPS and ArcGIS tools. For these areas, land use characteristics were identified either from digital cadastre maps from land registers or from LANDSAT 7 images with 30 x 30 m resolution, in both cases followed by corrections based on field surveys.

2.1.2. Determination of relative soil infiltration vulnerability

The determination of relative soil infiltration vulnerability and the delineation of infiltration-vulnerable areas was based on an analysis of five-digit codes of valuated soil ecological units (VSEU) using the method established by [38]. The VSEU maps are available as digital layers for the entire Czech Republic at a scale of 1:5 000. The VSEU code serves for the evaluation of soil characteristics according to the following criteria: main soil units, slope, exposure, skeletal character, and soil depth. Based on the categorization of these criteria, soil is classified into five relative groups according to its significance for the infiltration process, with category 1 corresponding to the maximum infiltration capacity. For the subcatchments of all tile drainage systems evaluated in this study, the proportion of land use types (arable land-AR, grassland-GR, forest-FR, built-up-BU) was determined within the infiltration–vulnerable categories I – IV; category V (the lowest infiltration–vulnerable category) was not present in any of the studied localities. An example of a tile drainage subcatchment with delineated infiltration-vulnerable areas / categories and related land use types is depicted in Figure 2.

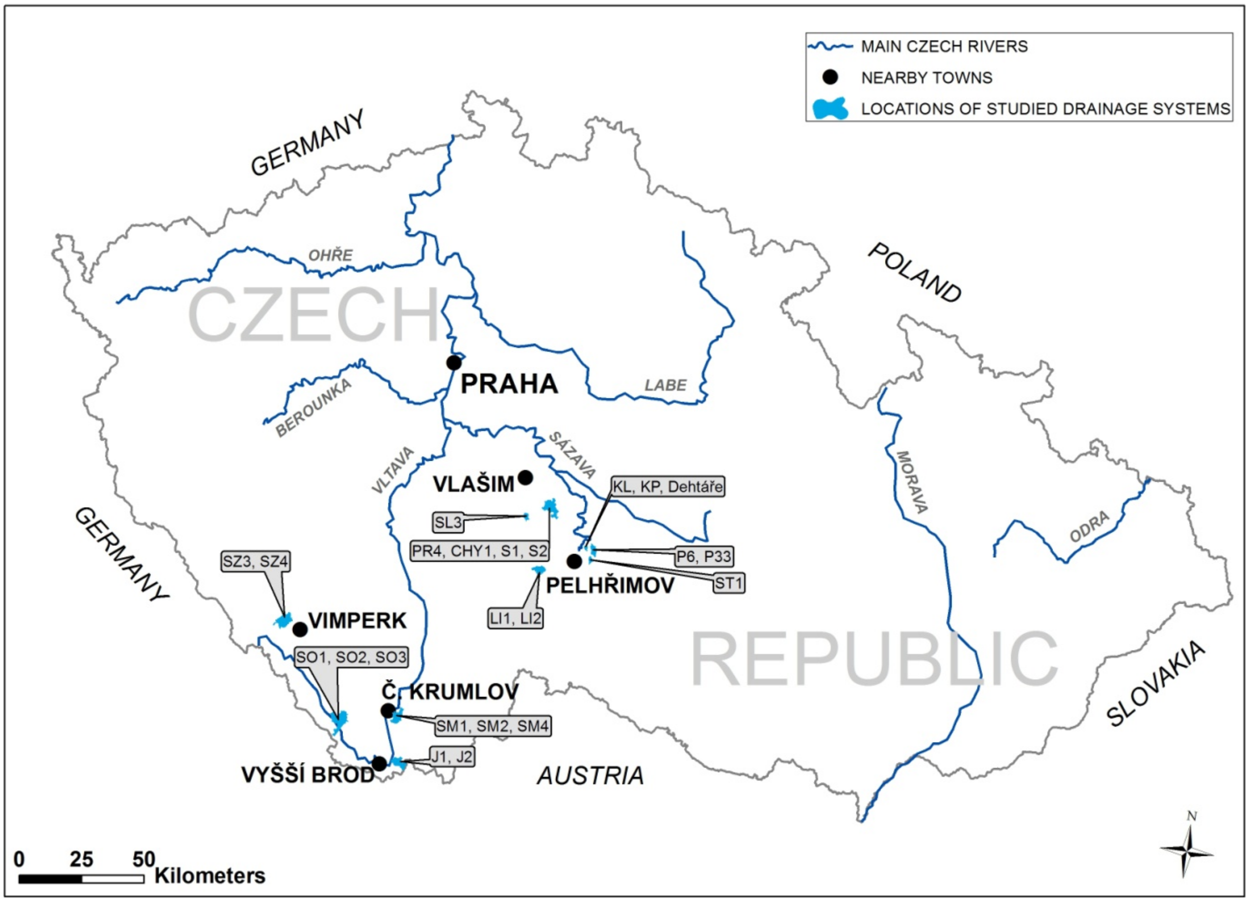

Figure 1. Location of the evaluated tile drainage systems and the Dehtáře catchment within the Czech Republic

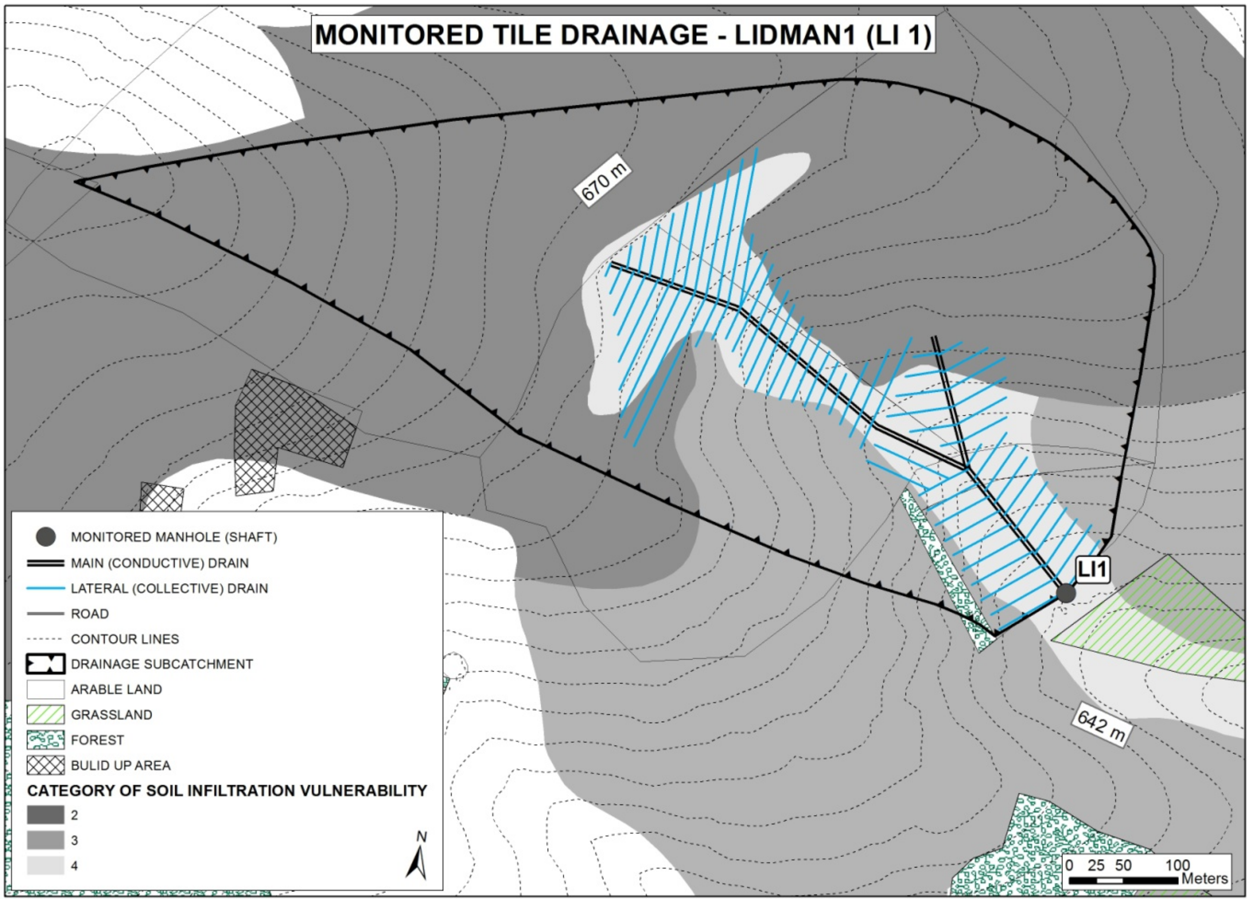

Figure 2. An example of a tile drainage subcatchment, with delineated infiltration-vulnerable areas and related land use types indicated.

2.1.3. Data preparation and employed statistical methods

Basic statistics were calculated for all monitored drainage systems. Due to the close relationships between actual nitrate concentrations and runoff [7,11] as representative values for further analyses, flow–weighted concentrations were taken in order to assess the mutual links existing solely between soil and land use catchment characteristics. Flow-weighted concentration values were computed according to the following formula:

$$Cfw = \frac{\sum (Ci * Qi)}{\sum Qi} \qquad (1)$$

Where Cfw is flow-weighted concentration

Ci is actual solute concentration in the sample

Qi is actual runoff during sample withdrawal.

To assess possible relationships between catchment soil and land use characteristics, and flow-weighetd nitrate concentration values, principal component and multiple stepwise regression (MR) methods were employed. Principal component analysis (PCA) is a widely applied method, which transforms the original data into new orthogonal coordinates in order to express the information and possible relations contained in the original data using a smaller number of variables. As the dependent variable, flow-weighted concentration (Cfw) of N-NO_3 was selected; as independent variables, the proportion of land use types within individual infiltration-vulnerable categories (AR1-4, GR1-4, FR1-4 and BU1-4). In PCA models, the following indicators were assessed: the cumulative variability explained by the first two components (%), and the component weights, showing how influential the original variables were in determining the structure of individual components. In general, factors found in the same biplot quadrants in the PCA are correlated positively; factors in opposite quadrants are correlated negatively. Factors situated around 90° angles are evaluated as independent. In the MR, the classical parameters of model correctness were considered: mutual parameter correlation – multicolinearity, autocorrelation of residuals – heteroskedasticity, detection of significant points – leverage value, deviation points – outliers (DFFITS diagnostics), and the p-values of the model and of the initial parameters. To compare different models describing identical data, the following indicators were considered: F statistics, R^2, R^2adj., (the adjusted coefficient of determination for the number of parameters, or degrees of freedom) and the Mean Square Error (MSE) [48] All analyses were done at the significance level of null hypothesis α=0.05.

2.2. Results

Table 1 shows the results of the basic statistics calculated on the tile drainage systems which were evaluated. Median N-NO_3 concentration values obtained during the period 2004-2006 were from 0.23 (SZ4) to 29.45 (KL) mg/l N-NO_3. Table 2 is an overview of land use type proportions accross four delineated soil vulnerable categories for all the evaluated drainage systems. The PCA and MR described mutual relationships between the assessed variables for the tile drainage subcatchments evaluated. In the PCA biplot (Figure 3), the points represent individual drainage subcatchments, and the beams are variables. In the assessed case, the vectors of variables Cfw and AR2 were positively correlated, as were AR3 and AR4 to a lesser extent. The factors FR4, GR4 and GR1 had a negative mutual relationship with Cfw N-$NO_{3.}$ The variables AR1 and GR3 were located in independent positions, most likely due to the very small area of all land use types in soil infiltration vulnerable category I. The results from various MR analyses are described in Table 3. The best (statistically significant and correct) model is coloured grey. It confirmed the results obtained from the PCA; the positively correlated factors were ratios of arable land within the first two soil infiltration vulnerable categories, and the ratio of grassland within the third soil infiltration vulnerable category appeared to be a mitigative factor on nitrate concentration. The final model from the MR is thus:

$$Cfw_{(N\text{-}NO3)} = 3.59 + 2.05*AR1 + 0.25*AR2 - 0.29*GR3 \quad (2)$$

Drainage system	Number of samples	Min	Max	Mean	10% Quantile	Median	90% Quantile	Flow-weighted concentration
			$N\text{-}NO_3^-$ (mg/l)					
ST1	58	7.68	40.44	25.39	18.55	25.07	34.02	24.66
LI1	53	13.10	30.95	21.81	16.72	21.01	29.01	22.32
LI2	47	14.01	22.82	17.51	14.46	17.12	21.01	18.33
SL3	51	1.13	16.04	2.16	1.13	1.13	3.19	3.16
PR4	57	1.13	23.04	14.47	8.76	15.36	18.84	14.51
CHY1	53	18.75	39.08	27.74	23.76	27.56	33.03	30.41
P6	66	7.48	23.72	15.50	13.33	15.14	18.52	15.28
P33	66	2.71	40.89	26.89	18.51	27.56	34.00	15.58
KL	70	8.81	46.08	27.50	17.85	29.45	34.43	27.01
KP	70	9.71	43.15	23.09	13.96	23.35	30.09	25.40
S1	52	1.13	26.20	7.94	1.45	6.21	13.78	9.59
S2	52	2.26	28.46	15.42	9.98	14.46	21.46	20.67
SM1	27	1.54	24.85	19.23	16.75	20.22	22.26	19.32
SM2	27	6.30	20.20	10.81	7.40	10.55	14.29	11.45
SM4	27	5.24	22.36	17.38	14.05	17.91	20.70	15.77
SO1	27	0.35	1.89	0.82	0.47	0.79	1.12	0.72
SO2	27	0.15	6.33	2.21	1.00	1.99	3.88	1.98
SO3	25	0.65	2.23	1.71	1.13	1.88	2.14	1.43
SZ3	20	0.24	1.83	1.08	0.75	1.08	1.61	1.00
SZ4	19	0.11	1.27	0.38	0.11	0.23	0.86	0.35
J1	26	1.44	6.78	4.76	2.82	5.04	6.29	4.10
J2	26	1.54	4.09	3.01	2.12	3.03	3.87	2.78

Table 1. Basic statistics of N-NO3-concentration values in the studied drainage systems

Drainage system	Area of drainage subcatchment	Soil-vulnerable category I.				Soil-vulnerable category II.			
	ha	Arable land	Grassland	Forest	Built-Up	Arable land	Grassland	Forest	Built-Up
		AR1	GR1	FR1	BU1	AR2	GR2	FR2	BU2
		% of the whole drainage subcatchment							
ST1	34.60	0.00	0.00	0.00	0.00	78.29	0.11	1.00	0.00
LI1	28.31	0.00	0.00	0.00	0.00	79.34	0.00	0.00	0.00
LI2	24.17	0.00	0.00	0.00	0.00	76.60	0.00	0.00	0.00
SL3	8.18	0.00	0.00	0.00	0.00	47.32	36.07	0.84	0.00
PR4	9.05	0.00	0.00	0.00	0.00	71.76	0.00	0.00	0.00
CHY1	1.94	0.00	0.00	0.00	0.00	100.00	0.00	0.00	0.00
P6	15.73	0.00	0.00	0.00	0.00	68.15	0.64	0.00	0.00
P33	19.73	0.00	0.00	0.00	0.00	48.29	0.45	5.14	0.00
KL	29.44	5.37	0.28	0.00	0.00	45.29	17.32	0.00	1.32
KP	28.33	5.02	0.61	1.15	0.40	51.77	2.20	5.60	1.93
S1	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
S2	3.82	0.00	0.00	0.00	0.00	51.60	0.00	0.00	0.00
SM1	15.08	0.00	0.00	0.00	0.00	58.09	0.00	0.60	0.00
SM2	48.40	0.00	0.00	0.00	0.00	18.99	8.41	0.19	0.00
SM4	45.99	0.00	0.00	0.00	0.00	25.51	16.68	27.40	0.00
SO1	21.20	0.00	10.19	0.00	0.00	0.00	51.23	2.55	0.00
SO2	10.83	0.00	0.00	0.00	0.00	0.00	3.23	1.20	0.00
SO3	12.56	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SZ3	52.11	0.12	6.93	0.02	0.00	14.20	9.54	13.09	0.00
SZ4	66.54	0.00	0.42	0.05	0.00	3.19	0.42	0.12	0.18
J1	46.80	0.00	5.62	1.82	0.00	0.00	54.49	10.24	0.00
J2	55.26	0.00	19.00	0.31	0.00	0.00	30.96	15.36	0.00

Drainage system	Soil-vulnerable category III.				Soil-vulnerable category IV.			
	Arable land	Grassland	Forest	Built-Up	Arable land	Grassland	Forest	Built-Up
	AR3	GR3	FR3	BU3	AR4	GR4	FR4	BU4
	% of the whole drainage subcatchment							
ST1	10.85	9.32	0.00	0.00	0.00	0.44	0.00	0.00
LI1	0.00	0.00	0.00	0.00	20.66	0.00	0.00	0.00
LI2	0.00	0.00	0.00	0.00	0.00	23.40	0.00	0.00
SL3	1.92	13.85	0.00	0.00	0.00	0.00	0.00	0.00
PR4	0.50	25.30	0.00	0.00	0.46	1.99	0.00	0.00

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CHY1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
P6	0.00	0.00	0.00	0.00	27.74	3.48	0.00	0.00
P33	0.00	0.00	0.00	0.00	19.92	26.21	0.00	0.00
KL	0.00	0.00	0.00	0.00	5.08	24.53	0.00	0.81
KP	1.90	0.14	0.00	0.00	24.05	5.23	0.00	0.00
S1	0.00	0.00	0.00	0.00	1.20	98.80	0.00	0.00
S2	48.40	0.00	0.00	0.00	0.00	0.00	0.00	0.00
SM1	17.71	0.00	23.61	0.00	0.00	0.00	0.00	0.00
SM2	5.95	13.35	52.93	0.00	0.00	0.19	0.00	0.00
SM4	0.00	0.00	0.00	0.00	30.42	0.00	0.00	0.00
SO1	0.00	0.00	6.89	0.00	0.00	29.15	0.00	0.00
SO2	0.00	0.00	44.23	0.00	0.00	47.92	3.42	0.00
SO3	0.00	0.00	58.20	0.00	0.00	39.17	2.63	0.00
SZ3	0.15	13.20	0.15	1.57	2.96	35.69	0.52	1.86
SZ4	8.58	0.48	66.10	0.23	0.38	17.52	2.27	0.08
J1	0.00	0.00	0.47	0.00	0.00	26.99	0.38	0.00
J2	0.00	0.00	0.00	0.00	0.00	28.23	6.03	0.11

Table 2. An overview of the evaluated tile drainage systems: Drainage subcatchment area and land use types in four soil infiltration-vulnerable categories

Variable	Coefficient	Standard deviation of residuals	P	R^2	R^2 adj.	F stat	MSE	Testing the regression triplet
Constant	3.2160		0.0378					
AR1	2.1655		0.0022					
AR2	0.2404		0.0000					
AR3	0.1157		0.1883					
GR3	-0.2684		0.0601					
Regression		4.0943	0.0000	85.9592	82.6554	26.0189	16.7630	OK
Constant	4.9037		0.0191					
AR1	1.9620		0.0053					
AR2	0.2295		0.0000					
GR3	-0.3120		0.0358					
FR4	-0.7534		0.3043					
Regression		4.1789	0.0000	85.3727	81.9309	24.8100	17.4633	autocorrel. of residuals
Constant	4.7588		0.0193					

Variable	Coefficient	Standard deviation of residuals	P	R^2	R^2 adj.	F stat	MSE	Testing the regression triplet
AR1	2.0825		0.0034					
AR2	0.2321		0.0000					
GR2	-0.0610		0.3254					
GR3	-0.2802		0.0546					
Regression		4.1900	0.0000	85.2946	81.8345	24.6509	17.5565	OK
Constant	3.5890		0.0223					
AR1	2.0512		0.0035					
AR2	0.2471		0.0000					
GR3	-0.2892		0.0469					
Regression		4.1930	0.0000	84.4073	81.8086	32.4800	17.5815	OK
Constant	2.9001		0.0723					
AR1	2.2738		0.0026					
AR2	0.2363		0.0000					
Regression		4.5681	0.0000	80.4646	78.4082	39.1296	20.8679	OK

Table 3. A list of statistically significant and correct multiple regression models (the grey band indicates the best regression model).

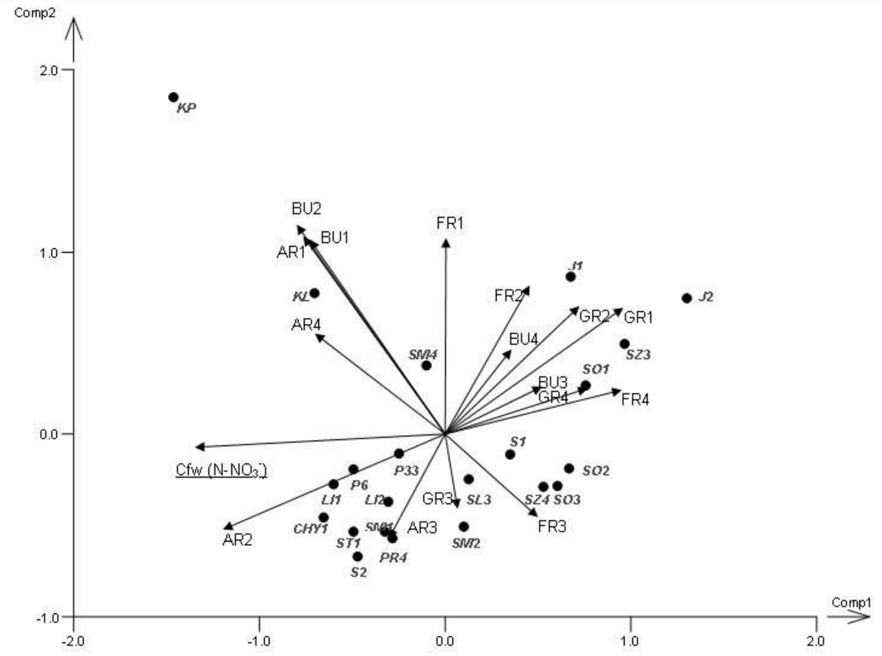

Figure 3. PCA Biplot for all variables – land use types in soil-infiltration vulnerable categories (I-IV)

Based on the acquired results, it can be said that the most important influence on values of nitrate nitrogen in drainage water was, in the case of the drainage systems analysed, the ratio (%) of ploughed land within the most infiltration vulnerable catchment areas.

3. Case study *B*

3.1. Materials and methods

3.1.1. Study area

The experimental catchment Dehtáře (Figure 1 and Figure 4) is situated in the Bohemian-Moravian Highlands, Czech Republic. It is a locally typical small agricultural catchment, where the tile drainage acts as the only permanent runoff from the catchment and the drainage system was built in the slope. The area is 57.9 ha, with tile-drained areas occupying 19 ha (32%). It has been used mainly as agricultural land, with low forest representation. The agricultural land is mostly exploited as arable, with permanent grassland in the lower part of the catchment. The altitude varies between 549.8 and 497 m a.sl. Total precipitation throughout the vegetation period ranges between 350 and 450 mm, and in the winter months between 250 and 300 mm, with a total annual average of 666 mm. The substrate is formed by partially migmatized paragneiss in various degrees of degradation. Quaternary sediments are represented by slope sands and loams reaching 1-2 m thickness. The representation of soils is variable, with Gleyed Cambisols, Gleysols, and sporadically Histosols. In the recharge area, the soil cover is more homogenous, with Haplic and, Shallow Haplic Cambisols and Cambic Hyperskeletic Leptosol prevailing. The drainage system was built in 1977, with a slope of 5%. The spacing of collection drains is 13 or 20 m apart; the depth of collection drains is 1.0 m, of conduit drains 1.1 m, and the interception drains are deposited at a 1.1-1.8 m depth. Detailed information about the catchment including a geophysical survey was published e.g. by [49].

3.1.2. Design of the pilot plant experiment

The water quality has been monitored since 2003. Samples have been taken at one or two week intervals. Five sites in the drainage system, with different land use in recharge and discharge areas, were chosen to be monitored for this analysis (Table 4). Part of the recharge area (Figure 4), with an area of 4.6 ha, has been grassed since the hydrologic year 2007. To evaluate the effects of this grassing, the whole time monitored was divided into two periods, period 1-before grassing (2003-2006) and period 2-after grassing (2007-2013). All grassland in the catchment was fertilised by approximately 100 kg N/ha per season (mostly by urea and liquid manure). The arable land in the catchment was fertilised according to crop rotation (grains, potatos and oilseed rape) in the amount of cca. 120 kg N/ha per season.

The most important sites for the evaluation were K1, K2 and K5. The site K1 represents an outlet of an eponymous systematic drainage subsystem with 1 ha area and drains spaced at 13 m. This subsystem is connected to an intercepting drain K2 which collects hypodermic water from upslope areas. The discharge area of these sites (area of their placement) was permanently

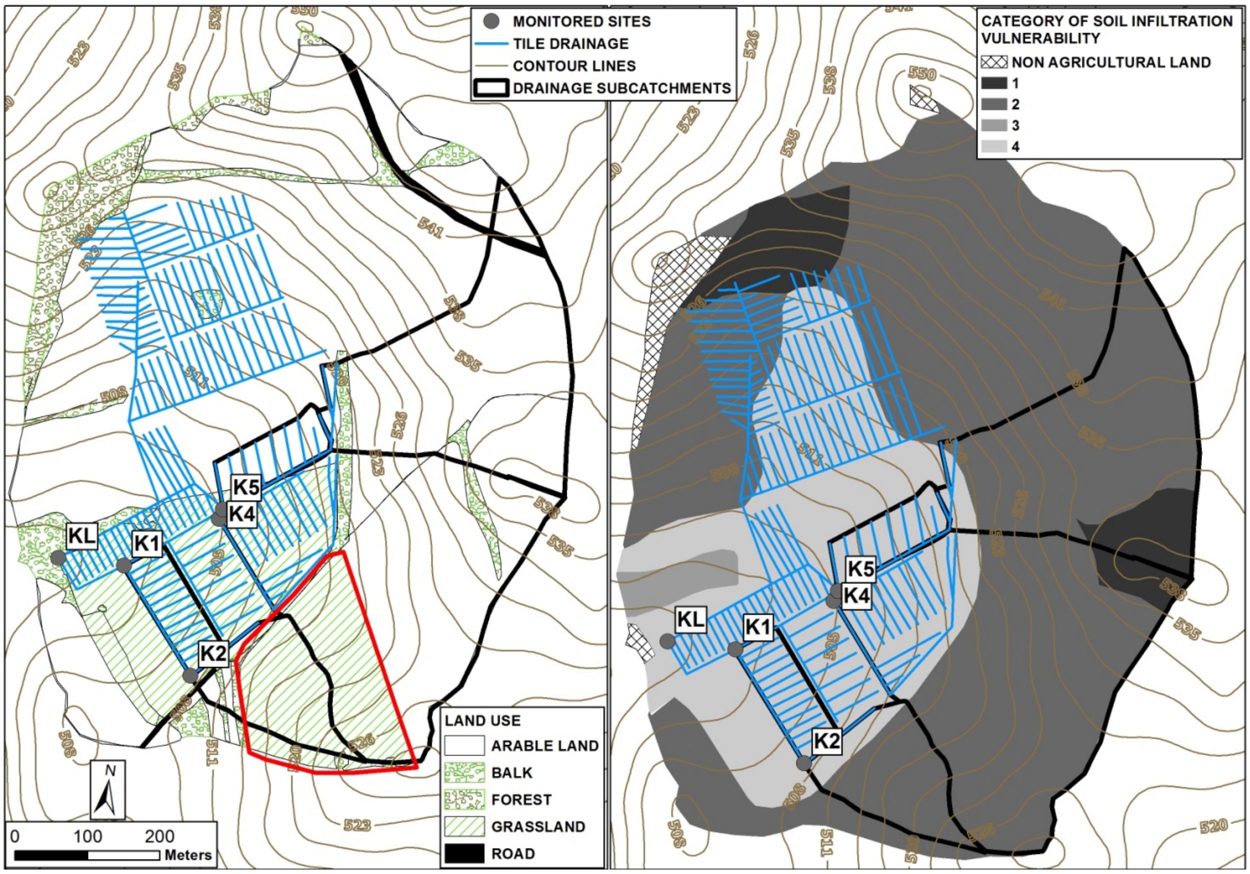

Figure 4. An overview map of the experimental catchment Dehtáře. Red encircled is the area experimentally grassed in 2007.

grassed, their recharge area had been used as arable land until the hydrologic year 2007, when it was experimentally grassed. The site K5 represents an outlet of an eponymous systematic drainage subsystem with 1 ha area and drains spaced at 20 m; both recharge and discharge areas were used as arable land. The site KL is the closing outlet of all these subsystems.

Site	Drainage type	Period 1 (2003-2006) land use		Period 2 (2007-2013) land use	
		Discharge area	Recharge area	Discharge area	Recharge area
K1	systematic	grassland	arable land	grassland	grassland
K2	intercepting	intercepting drain	arable land	intercepting drain	grassland
K4	systematic	grassland	arable land	grassland	arable land
K5	systematic	arable land	arable land	arable land	arable land
KL	closing profile	arable land + grassland	arable land	arable land + grassland	arable land + grassland

Table 4. Land use in the recharge and discharge areas of the drainage systems used for the experiment

3.1.3. Data evaluation

Similarly to case study *A*, the solute concentration value data were adjusted to facilitate hydrologic modeling. Flow-weighed nitrate concentration values, computed according to formula (1), were further analysed using the statistical software Statgraphics. In addition to calculating common summary statistics, the analysis of variance and Kruskall-Wallis test were conducted to test whether there were significant changes in nitrate concentrations and nitrogen load caused by grassing.

3.2. Results

The nitrate concentrations in drainage waters were strongly variable during the entire period monitored and also between the particular periods. The observed concentrations varied from 18 to 253 mg/l throughout the whole period monitored. The main reasons for this variability were the variable soil nitrogen stocks and the strong concentration dependence on the discharge levels. That is why the highest concentrations were measured in late summer or early autumn – during the period of low drainage discharges and prevailing base flow. On the contrary, during spring snowmelt and summer high-flow events with increased discharge (especially its direct-event component), nitrate concentrations decreased due to a high degree of dilution. The exceptions were some high-flow events measured on sites with arable land in the recharge zone just after the application of fertilizer. The inter-seasonal variability was caused mainly by different precipitation courses in particular seasons and by crop rotation.

The results of the statistical analysis for both periods of the experiment (2004-2006 and 2007-2011) are depicted in Figure 5 and Table 5. The evaluation showed that the flow-weighted nitrate concentrations in period 1 (before grassing the recharge zone) were surprisingly higher in drainage subsystems K1 and K4 with the permanent grassland in drained area (discharge zone) than in the subsystem under arable land. Moreover, the concentrations mostly exceeded the level of 100 mg/l. After grassing the K1 subsystem recharge area, some changes occurred. At first, the nitrate concentration decreased during high–flow events. After that, approximately one year after grassing, the long-term course of NO_3 concentrations changed direction and became decreasing (Figure 6).

The significance of the changes in nitrate concentration values was tested using the Kruskall-Wallis test comparing the medians of the concentrations from periods 1 and 2 (before and after grassing). The results of this test are presented in Table 6. It is obvious that the statistically significant decrease in nitrate concentrations happened in the grassed recharge zone. Decreases of 32.1% and 25.7% were detected in systematic drainage subsystem K1 and intercepting drain K2 respectively. In the same period, an increase in nitrate concentration was detected in sites without land use change in their recharge zone. There was an increase of 10.8% in the drainage subsystem K5 with arable land in both (recharge and discharge) zones and of 8.6% in the subsystem K4 with grassland in the discharge zone, but arable land in the recharge zone. Evaluating the whole drainage system, the fall in nitrate concentrations by 10.5% was detected after grassing about 20% of this systems recharge zone. Nevertheless, from the statistical point of view, this fall appears to be insignificant.

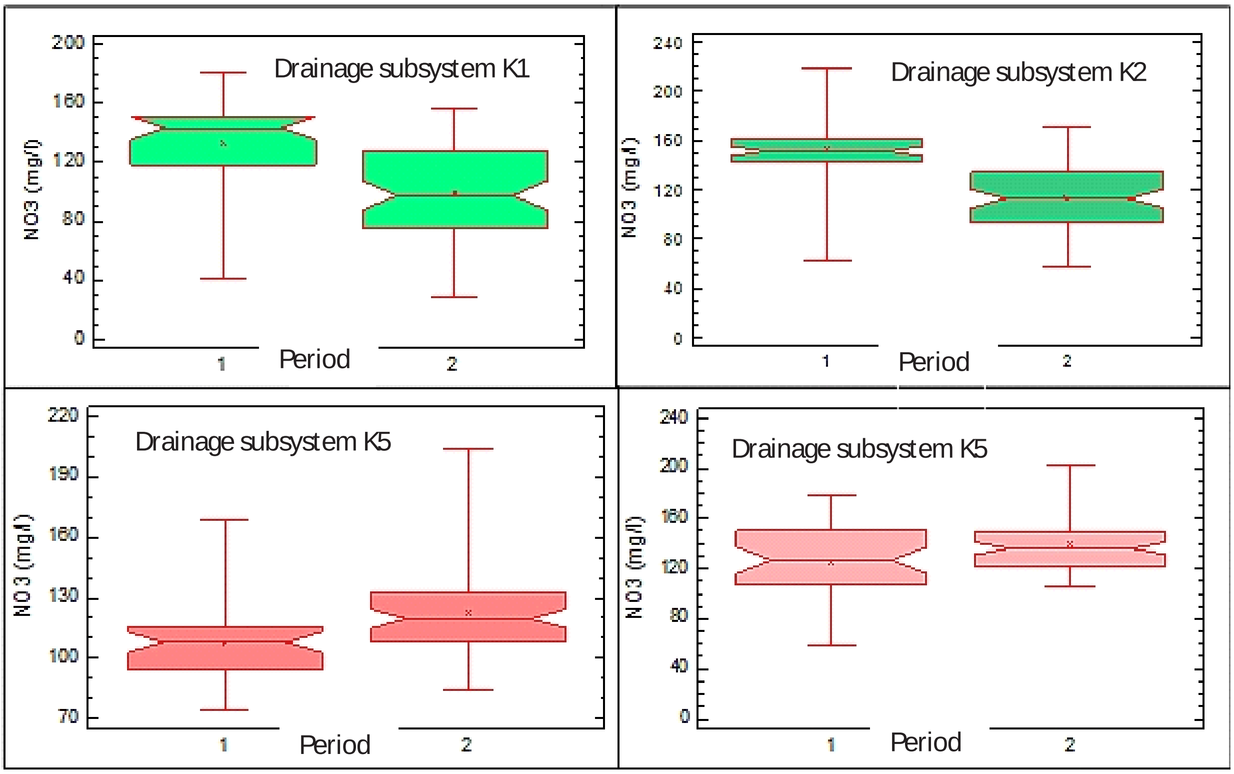

Figure 5. Nitrate concentrations in waters of drainage subsystems with different landuse before (period 1) and after (period 2) grassing of a part of a catchment recharge zone.

Site	Period 1 NO_3 (mg/l)				Period 2 NO_3 (mg/l)			
	Mean	**Median**	**Minimum**	**Maximum**	**Mean**	**Median**	**Minimum**	**Maximum**
K1	132.7	143.3	40.6	181.5	99.5	97.4	29.2	155.9
K2	152.5	151.5	62.9	219.0	113.0	112.9	56.7	170.5
K5	107.9	108.8	74.0	168.6	122.7	119.7	84.7	203.8
K4	125.3	126.6	58.6	178.2	139.4	136.2	105.6	202.9
KL	109.4	107.3	39.7	168.5	98.6	96.0	36.1	147.9

Table 5. The basic statistical evaluation of nitrate concentrations in water of drainage subsystems with different landuse before (period 1) and after (period 2) grassing in part of a recharge zone.

Despite the decrease proved above, the nitrate concentration values in the drainage water have still remained too high. The reason is that most of the samples were taken during prevailing base flow. The concentrations in this runoff component would remain high for a longer time because of the long residence time of ground and hypodermic water in the catchment. However, what is much more important than the instantaneous level of nitrate concentration, is that the trend in nitrate concentration became permanently decreasing in all measured sites with grassed recharge zone (Figure 6). While the linear trend was detected increasing in all sites during period 1, it reversed approximately one year after grassing.

Site	Landuse in recharge zone during experiment	Change in median of NO_3 concentrations (%)	Results of Kruskall-Wallis test	p value
K1	grassland	-32.1	significant	0.001
K2	grassland	-25.7	significant	0.001
K5	arable land	10.8	significant	0.002
K4	arable land	8.6	significant	0.028
KL	arable land + grassland	-10.5	insignificant	0.065

Table 6. The rate of change in nitrate concentrations after partially grassing a recharge zone and the results of the Kruskall-Wallis test for determining the significance of the change

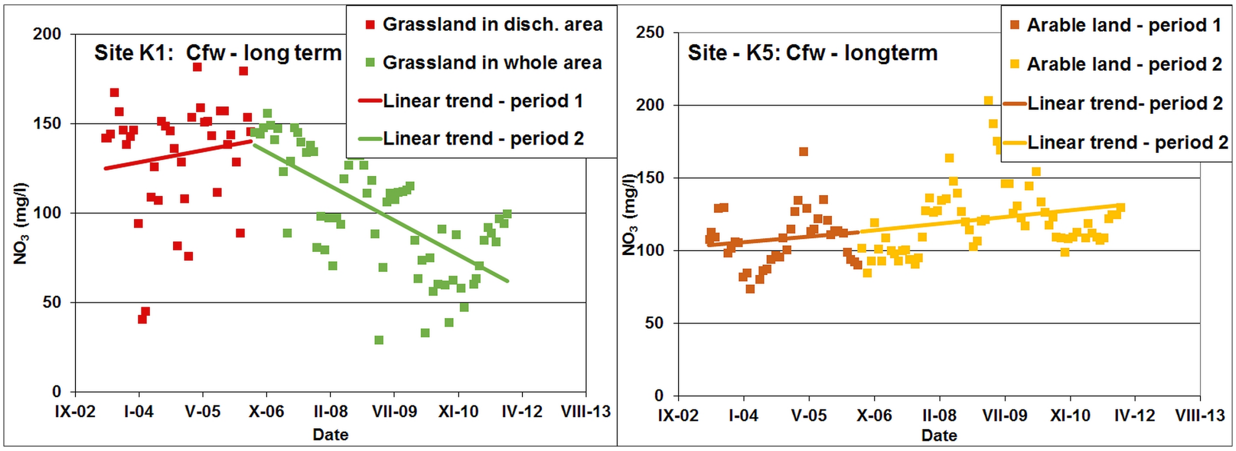

Figure 6. The long term trend of nitrate concentrations in the drainage subsystem K2 with grassed recharge zone and drainage subsystem K5 with remaining arable land in its recharge zone

In association with the change in nitrate concentrations, the nitrate-nitrogen leaching decreased after grassing in recharge area of the drainage system. Basic statistical evaluation of nitrogen leaching from drained subcatchments with different land use is depicted in Figure 7. Again, in all sites with grassed recharge zone, the decrease in all statistical characteristics of nitrogen load happened since part of the recharge zone was grassed (period 2). In the scale of whole drainage system, the monthly average load decreased by 23% from 3.2 kg N/month/ha to 2.6 kg N/month/ha. In the drainage subsystem K1, where the recharge zone was grassed completely, the decrease of the monthly average nitrogen load was even by 47% from 4.75 kg N/month/ha in period 1 to 2.52 kg N/month/ha in period 2. Evaluating drainage subsystems without land use change in the recharge zone, N load stagnation was registered in subsystem K4 (grassland in the discharge zone and arable land in the recharge zone), where the average monthly N load was 4.0 kg N/month/ha in period 1 and 3.9 N/month/ha in period 2. In the subsystem K5 (arable land in both zones), the increase of N load by 17% (from 4.1 to 4.8 kg N/month/ha) was recorded consequently with N load decrease in subsystems with grassed recharge zones. This decrease manifested as much more significant during high–flow events, which had the biggest share of the total annual nitrogen loads.

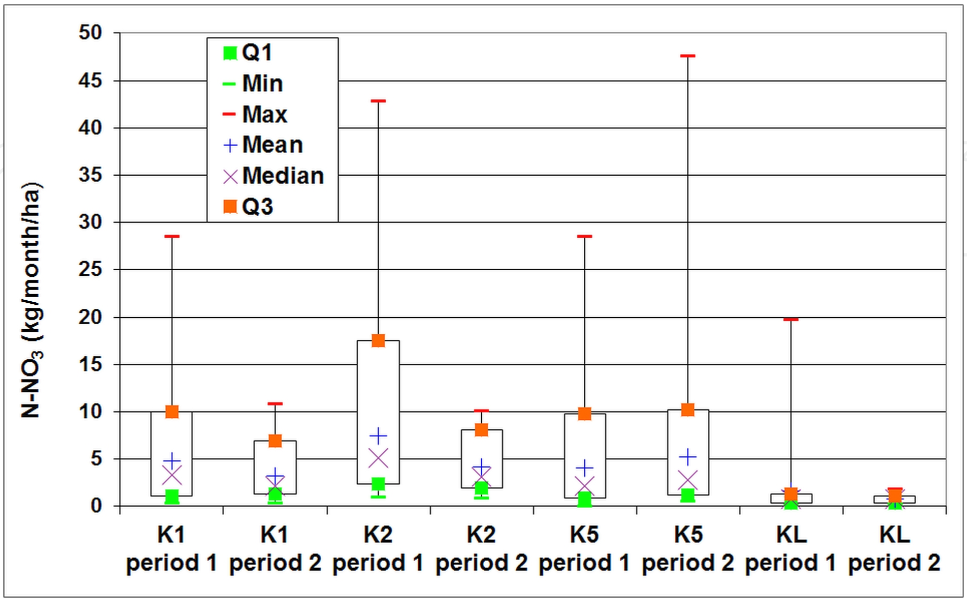

Figure 7. The average monthly N-NO_3 load per 1 ha from drainage subsystems with different landuse (period 1 – before grassing, period 2-after partially grassing the recharge zone)

4. Discussion

The results of both studies reported in this chapter corresponds with findings about grassland ability to reduce nitrogen leaching, as reported by many author from the Czech Republic and other countries. As it has been often demonstrated [30,31,45,50,51], nitrate concentrations in waters generally increase in connection with land use gradient from forests, meadows, pastures to arable lands; the last being the most worsening one. According to [44], the nitrate concentrations in small water courses of the Czech Republic are more affected by the area of arable land within their catchments than by the amount – to a certain level-of applied nitrogenous fertilizers. Other authors [52,53,54] have also reported the positive effect of grassing on water quality while some authors mention a certain lag effect of grassing on water quality improvement [55]. The degree of NO_3 concentration and N leaching changes evaluated by the case study *B* is within the range of findings reported by [44] in the Czech Republic. That work documented by multifactorial regression within the Švihov drinking water reservoir basin the share of arable lands as the most important factor affecting the nitrate concentration in water of small water courses. Similarly, the study [23] proved for catchments of three different scales significant linkages between landuse and water quality, reporting that every decrease in arable land area by 10% would cause an average decrease in C90 NO_3 values of 6.4 mg/l. The work [44] further proved that for the small streams in the Šumava region (Southern CR), improvement of water quality in this region was caused by grassing of arable and tile drained soils. The delay of reaction of nitrate concentration on grassing, detected in the case study *B*, was approximately one year. This period corresponds to the average residence time of the prevailing drainage runoff component (hypodermic water) in drainage discharge in sloped catchments, as reported e.g. by [56].

The ability of grasslands to reduce nitrate pollution is explained by the fact that grassland can absorb and use bigger amount of nitrogen in comparison to field crops [57]. Permanent grasslands cover the soil year round and have a big stock of active subsurface biomass in the root system, which can immobilize a significant amount of soil nitrogen. Nitrogen in mineral form thus exists in the soil in small concentration because nitrates produced by nitrification are readily utilized by the grass. Besides this, after grassland is fertilised, nitrogen is quickly immobilised in the soil organic matter and protected againts leaching. Another important factor for grassland efficiency in nitrogen load is the bigger amount and increased activity of soil microbes, which is much higher under grassland than under field crops [58-61].

The results of the experiment *B* in the Dehtáře catchment also implied that grassing, being considered as an efficient measure for nitrogen leaching mitigation, should include a proper targeting in certain landscape zones, especially in tile drained catchments [8,44,62]. Drainage systems in the foothill areas of highland are characterized by their location in slopes and a considerable portion of drainage runoff can originate outside of the drained area [18-19,63]. To understand the mechanism of drainage runoff generation in sloped conditions, the theory of catchment slope zones (areas) must be applied. Perceived hydrogeologically, a catchment splits into recharge zones, where rainwater infiltrates and gradually joins groundwater, and discharge zones, where groundwater approaches surface water body or soil surface [22]. The recharge zones are usually situated in the morphologically uppermost areas of a catchment, close to the catchment watershed divide. The soils of recharge zones are typically shallow and stony, with high sand content and high infiltration capacity. The coarse-textured soils of the recharge zones are, with a respect to groundwater resources, well suited to grassing, which, beside water quality benefits, increases their field water holding capacity and enables infiltration of a bigger precipitation amounts – compared to ploughlands, including rainstorms [18,23,64-65]. The discharge zones are usually situated in the foothills and along surface water courses and lakes and are prone to surface waterlogging. Typical soils in the discharge zones are generally deeper and heavier, with higher clay content and a lower capacity for infiltration. A connection between the recharge zones and the discharge zones is carried out by transient zones, where water from precipitation is either transformed to surface runoff or to groundwater which flows downslope in a quasi-steady way [18,24,27]. The transient zones are situated mainly in the middle sections of slopes. Groundwater in natural catchments flows from the recharge zones to the discharge zones. The discussed drainage systems in slopes are mostly placed in at the interface of the transient and discharge zones or in the discharge zones [18]. The joint influence of different land use, soil properties and tile drainage within various landscape zones on the hydrological regime of a small catchment was demonstrated by [2]. Being built in slope, tile drainage systems represent a shortcut between recharge and discharge zones which significantly curtails the water residence time in catchments, hastens the runoff reaction to precipitation, shortens the time to reach the peak discharge during events and increase the nitrate concentrations and loads in related waters [4,11,17,66-68].

5. Conclusions

The results presented show that nitrate concentration values in drainage water were influenced the most by the land use of the recharge zones within the drainage subcatchment. These findings can be generalised for slopy agricultural catchments with common land use in soil environments formed on crystalline rocks. The land use and soil analyses together with monitoring the drainage water quality and quantity, and also an experiment with land use change in the recharge zone proves that grassing focused on the proper catchment area can be employed as a useful tool for reducing nitrates in drainage water. While permanent grassland placed directly in the drained area (corresponding to the catchment discharge zone) did not show any influence, the grassing focused on the catchment recharge area demonstrated a significant decrease in both, NO_3 concentrations and N loads. The acquired findings are of crucial importance for improving the water quality of small streams as well as groundwater in agriculturally exploited areas, for planning protective zones within large catchments of potable water reservoirs, and also for protecting small local surface or groundwater sources of potable water.

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