

Modeling Non-point Source Solute Nitrogen Transport in Drainage Ditches under Coupled Water-Sediment-Plant Interactions: A Postprint

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Abstract

“Process interception” by ditch systems represents an important approach for the control and management of agricultural non-point source pollution at the present stage. In response to the current lack of clarity regarding migration and transformation mechanisms of non-point source solutes among different media within the water-sediment-plant system of farmland drainage ditches, this study—based on a brief analysis of the water-sediment-plant composite ecological structure of farmland drainage ditches and the functional characteristics of its components, and focusing on water in the ditch as the central medium—analyzed the migration and transformation processes of non-point source nitrogen solutes within single media and between different media in the ditch system. Considering that both water and pollutants in drainage ditches primarily migrate linearly along the ditch direction, a one-dimensional migration model for farmland drainage and its non-point source pollutants in drainage ditches was developed using the water flow continuity equation and non-conservative pollutant transport-diffusion equation, and appropriately simplified according to the specific conditions of the experimental ditch. Through controlled laboratory monitoring experiments, a quantitative calculation method was also proposed for assessing the influence of different media (water, sediment, plant) in the ditch system on the attenuation degree of non-point source pollutants in water. The established model and calculation parameters were applied to a field experimental ditch, with ammonium nitrogen and nitrate nitrogen serving as examples for simulation. The results demonstrated good agreement between simulated and measured values, with simulation efficiency coefficients of 0.87 for ammonium nitrogen and 0.93 for nitrate nitrogen, indicating that the established model and simulation approach conform to the actual conditions of the experimental ditch and exhibit good applicability.

Full Text

Simulation of Non-Point Source Solute Nitrogen Transport in Drainage Ditches Under Water-Sediment-Aquatic Plant Synergy

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Abstract: As water environmental issues become increasingly prominent and point source pollution from industrial wastewater is gradually controlled, agricultural non-point source pollution resulting from unscientific application of fertilizers and pesticides has attracted growing attention. Building upon strengthened “source control” in fields, maximizing the “process interception” function of drainage ditch/pond systems represents a critical approach for agricultural non-point source pollution control and management. Drainage ditches possess compound ecosystem structures consisting of water-sediment-plant synergy, and understanding the migration and transformation of non-point source solutes is essential for studying the interception, control, and management of agricultural non-point source pollution in these systems. However, the transformation processes of non-point source solutes within each medium of the water-sediment-plant system in farmland drainage ditches remain largely unclear. Based on analysis of the ecological structure and functional characteristics of farmland drainage ditches, this paper describes the transformation of non-point source soluble nitrogen within the water-sediment-plant system. Combining the characteristics of water pollution migration in farmland drainage ditches, a model for agricultural water pollution migration was constructed based on the continuity equation for flow and pollutant migration equations. The model was used to develop a quantitative calculation method for assessing the influence of different water-sediment-plant media on the attenuation degree of non-point source pollutants in water bodies. Using the Qingshui drainage ditch in the People’s Victory Canal Irrigation District in Henan Province as a case study, the model was applied to calculate parameters for ammonia nitrogen and nitrate nitrogen pollutants in a field experimental ditch. The results showed good agreement between simulated and measured values, with simulation efficiency coefficients of 0.87 for ammonia nitrogen and 0.93 for nitrate nitrogen, demonstrating that the model is available and effective for simulating drain pollution in agricultural farmlands.

Keywords: Non-point source pollution; Farmland drainage ditch; Water-sediment-plant system; Nitrogen; Transportation simulation

1.1 Ecological Characteristics of Drainage Ditches

Farmland drainage ditch systems originate from field-scale ditches, connecting through branch and main channels before discharging into larger water bodies. The ecological structure of drainage ditches can generally be divided into three layers [Figure 1: see original paper]. The upper layer consists of emergent stems and leaves of aquatic plants extending above the water surface, which vary with water level fluctuations. Aquatic plants are typically emergent species adapted to drainage ditch environments, such as reed (*Phragmites communis*) and cattail (*Typha orientalis*), which grow vigorously in summer and autumn and wither in winter and spring. The middle layer comprises farmland drainage water, plankton, and the submerged portions of plant stems and leaves, with water levels exhibiting periodic changes in response to irrigation and rainfall-runoff events. The bottom layer contains plant root systems, nutrient-rich sediment substrates (deposits), and various microorganisms thriving within them.

This unique water-sediment-plant ecological structure determines the distinctive characteristics of drainage ditches. Aquatic plants adapted to this environment undergo periodic annual growth cycles. Sediment substrates at the ditch bottom are periodically exposed and submerged with water level fluctuations. The abundant “nutrient source” in sediment deposits supports plant growth and microbial proliferation. When farmland drainage flows through the system, complex physical, chemical, and biological transformations occur for nutrients including organic matter, nitrogen, and phosphorus [11].

1.2 Migration and Transformation Processes of Nitrogen Nutrients in Drainage Ditches

Nitrogen loss from excessive fertilization and irrigation constitutes the primary nutrient salt in farmland drainage ditches. Nitrogen exists in two main forms in drainage water: organic nitrogen and inorganic nitrogen. The migration and transformation of nitrogen within the water-sediment-plant system of farmland drainage ditches includes mineralization (ammonification), nitrification and denitrification, plant uptake, and sediment adsorption. These processes can be specifically decomposed as follows: First, nitrogen exchange between sediment substrate and water body, including sediment deposition and adsorption of organic nitrogen from water, ammonium nitrogen regeneration from sediment, microbial uptake and decomposition of ammonium and nitrate nitrogen from water, and nitrification-denitrification between water and sediment. Second, nitrogen transformation between aquatic plants and water. On one hand, plants can directly absorb NH_4^+ and NO_3^- ions from ditch water and assimilate them into required substances (proteins and nucleic acids), thereby purifying the water. On the other hand, due to the periodic nature of plant growth, withering in autumn and winter increases organic nitrogen in water, which subsequently mineralizes into ammonium nitrogen, causing secondary water pollution. The migration and transformation of nitrogen in the water-sediment-plant system of farmland drainage ditches is summarized in [Figure 2: see original paper].

2.1 Water and Pollutant Transport Model for Ditch Systems

Considering that water and pollutants in drainage ditches primarily migrate longitudinally along the ditch direction, based on the water flow continuity equation and non-conservative pollutant transport-diffusion equation, previous research [12] described water and pollutant transport in drainage ditches using Equations (1) and (2). In these equations: Q represents the flow rate through ditch cross-section A at time t ($\text{m}^3 \cdot \text{s}^{-1}$); q is the lateral flow intensity (considering ditch seepage) ($\text{m}^3 \cdot \text{s}^{-1} \cdot \text{m}^{-1}$); A is the cross-sectional area of flow (m^2); x is the distance from the control section to the simulated section (m); C is the concentration of typical pollutants in drainage ditches ($\text{mg} \cdot \text{L}^{-1}$); C_q is the concentration of typical pollutants at various ditch “point source” inlets ($\text{mg} \cdot \text{L}^{-1}$); C is the concentration of a certain pollutant in deep drainage water accepted by branch-level drainage ditches ($\text{mg} \cdot \text{L}^{-1}$); u is the cross-sectional average flow velocity ($\text{m} \cdot \text{s}^{-1}$); K_s is the pollutant shear flow dispersion coefficient ($\text{m}^2 \cdot \text{s}^{-1}$), also known as the longitudinal dispersion coefficient, which characterizes the rate of pollutant dispersion along the flow direction (longitudinal) in flowing water, with the physical meaning being the area of pollutant longitudinal dispersion per unit time; K is the comprehensive attenuation coefficient (d^{-1}), representing the rate of pollutant attenuation in water, with the physical meaning being the percentage of pollutant attenuation in water per unit time. The comprehensive attenuation coefficient depends not only on the degradability of the pollutant itself but also on hydrodynamic conditions and external water temperature.

The field experimental ditch for this study was the Qingshui Ditch in the People’s Victory Canal Irrigation District of Henan Province. The test section was 600 m long with no other ditch branches between the upstream and downstream control sections, featuring relatively closed flow with minimal variation between upstream and downstream. Additionally, due to the shallow depth of the drainage ditch, deep secondary drainage and lateral seepage were minimal. Therefore, Equations (1) and (2) could be simplified for this study by removing the lateral flow intensity q , branch ditch inflow term C_q , and deep drainage term C , resulting in the simplified Equations (3) and (4). In these equations, parameters including flow rate Q , cross-sectional area A , flow velocity u , and typical pollutant concentration C can be measured experimentally. The pollutant shear flow dispersion coefficient K_s can be derived using measured flow velocity or concentration data from simulated cross-sections [13].

2.2 Determination of Comprehensive Attenuation Coefficient K

The comprehensive attenuation coefficient K not only reflects the degradation rate of pollutants under water body action but also serves as an important parameter for studying regional water quality changes and calculating water environmental capacity and pollution assimilation capacity. Although research on determining K values has a long history both domestically and internationally, existing methods and results primarily focus on large water bodies such as rivers and lakes, with limited research on comprehensive attenuation coef-

ficients in farmland drainage ditches. Combining the nitrogen migration and transformation processes in ditch systems shown in [Figure 2: see original paper], the attenuation coefficient K for farmland drainage ditches can be further decomposed as:

$$K = K_w + K_m + K_p$$

where K_w refers to pollutant attenuation caused by water body movement (d^{-1}), such as surface evaporation and nitrification-denitrification occurring within the water body; K_m is pollutant attenuation in water caused by ditch bottom sediments (d^{-1}), including organic nitrogen sedimentation, ammonium nitrogen adsorption and regeneration, and microbial decomposition of pollutants in water; and K_p is pollutant attenuation in water caused by aquatic plants in ditches (d^{-1}), including plant adsorption and purification of various nitrogen-containing ions from water (positive value) and increased nitrogen pollutants in water due to plant withering (negative value). K_w , K_m , and K_p can be determined through either monitoring experiments or relevant kinetic models. This study employed the monitoring experiment method.

2.3 Parameter Determination

To determine the temporal variation of K_w , K_m , and K_p in farmland drainage ditches, static monitoring experiments were conducted at the Water Saving and Agricultural Ecology Experimental Base of the Yellow River Institute of Hydraulic Research in Xinxiang City, Henan Province, due to the uncontrollable nature of various influencing factors in field experiments.

The experiments involved three identical PVC simulation tanks (length \times width \times height: 2 m \times 0.65 m \times 0.65 m). Tank 1 contained only water samples to determine K_w . Tank 2 had 20 cm of sediment uniformly laid at the bottom, with sediment sourced from the Qingshui Ditch in the People's Victory Canal Irrigation District, naturally air-dried, crushed, and impurities removed. Tank 3 contained both 20 cm of sediment and transplanted reeds at an initial planting density of 10 plants \cdot m². The experimental period was from April 2014 to March 2015. During the experiment, nitrogen and phosphorus concentrations in the water were regulated using mixed solutions of urea [CO(NH₂)₂] and diammonium hydrogen phosphate [(NH₄)₂HPO₄] based on measured pollutant concentrations from the field experimental ditch. Water samples were collected every 10 days, and sediment samples were collected monthly. All samples underwent necessary pretreatment according to relevant regulations before laboratory analysis, with the main analytical instrument being the Smartchem 140 automatic chemical analyzer. Specific experimental arrangements are described in reference [14].

Based on monitoring data from water and sediment samples, nitrogen content in water was calculated using the volumetric method (Equation 6), and nitrogen content in sediment was calculated using the gravimetric method (Equation 7). In these equations: W_w and W_m represent nitrogen content in water and sediment (g), respectively; V_w is the water volume in the tank (m³), measured

on-site during sampling; M_m is the sediment mass in the tank during monthly sampling (kg), obtained by subtracting the extracted sediment mass from the total initial sediment mass; C_w is the volumetric concentration of corresponding nitrogen in water ($\text{mg} \cdot \text{L}^{-1}$); and C_m is the mass concentration in sediment ($\text{g} \cdot \text{kg}^{-1}$).

Based on these calculations, K_w can be further calculated using Equation (8), while K_m and K_p can be similarly derived. $K_w = (W_{w,i,1} - W_{w,i+1,1}) / (W_{w,i} \times d)$, where i and $i+1$ refer to two adjacent sampling events and d is the number of days between samplings. It should be noted that the attenuation coefficients calculated using these equations were obtained from static tank simulations of drainage ditch monitoring experiments, which differ from field farmland drainage ditches in terms of hydraulic characteristics and aeration. However, their advantage lies in experimental controllability, enabling separate estimation of the influence degree of sediment and aquatic plants on solutes in water.

2.4 Model Input and Solution

During model calculation, water quantity used Type 1 (Dirichlet) boundary conditions, with the upper boundary condition being the water volume at the starting end of the simulated drainage ditch and the lower boundary condition being the water level at the ditch outlet, forming a closed calculation model. Pollutant calculation employed Type 2 (Neumann) boundary conditions, with pollutant concentration at the starting end of the simulated drainage ditch set as the initial condition. The solution process can be decoupled, meaning the flow continuity equation is solved first to obtain hydraulic parameters, followed by solving the pollutant transport-transformation equation to derive typical pollutant output results from the ditch.

After determining relevant parameters, the flow equation was discretized using a four-point implicit difference scheme, while the shear flow dispersion equation for pollutant advection and attenuation was discretized using an explicit difference scheme, with computer programming employed for solution.

3.1 Model Validation

The established model and related parameters were applied to and validated using the field experimental ditch—the Qingshui Ditch in the People's Victory Canal Irrigation District. Originating in Yuanyang County, Xinxiang City, Henan Province, the ditch flows through Huojia County before entering the main channel of the People's Victory Canal. Named for its historically clear water, the ditch has experienced water quality deterioration in recent years due to fertilizer and pesticide application in surrounding farmland and the inflow of agricultural drainage. During irrigation periods, large volumes of drainage enter from upstream while farmland on both sides extracts water from the ditch for irrigation, making it an integrated irrigation-drainage ditch. The experimental

ditch section was located approximately 500 m downstream of the intersection hub between Qingshui Ditch and the East Main Canal, with a total length of 600 m, relatively closed conditions, and no other inlets or outlets. Three cross-sections (Nos. 1, 2, and 3) were established along the flow direction, as shown in [Figure 3: see original paper].

During simulation, nitrogen concentrations measured at Cross-section 1 (D1) along the flow direction in [Figure 3: see original paper] were used as input concentrations, while Cross-section 3 (D3) served as the target concentration for simulation, with ammonia nitrogen and nitrate nitrogen as examples. Results are presented in [Figure 4: see original paper].

3.2 Analysis of Simulation Result Rationality

Measured volumetric concentrations of ammonia nitrogen and nitrate nitrogen at Cross-section 3 are plotted in [Figure 4: see original paper]. The simulation results for ammonia nitrogen show some differences from measured values at concentration peaks and troughs. On April 24, 2014, the measured peak concentration was $0.41 \text{ mg} \cdot \text{L}^{-1}$, while the simulated concentration was $0.46 \text{ mg} \cdot \text{L}^{-1}$. The lowest measured concentration was $0.10 \text{ mg} \cdot \text{L}^{-1}$ on August 12, 2014, with a corresponding simulated value of $0.07 \text{ mg} \cdot \text{L}^{-1}$, showing slight discrepancies at peak and valley points. The average measured ammonia nitrogen concentration during the simulation period was $0.23 \text{ mg} \cdot \text{L}^{-1}$, while the simulated average was $0.25 \text{ mg} \cdot \text{L}^{-1}$, indicating close agreement. Comparison of the measured and simulated ammonia nitrogen concentration curves in [Figure 4a: see original paper] reveals consistent trends. The nitrate nitrogen simulation in [Figure 4b: see original paper] shows better agreement than the ammonia nitrogen simulation in terms of both peak-valley fitting and concentration curve trends. The average measured nitrate nitrogen concentration was $0.98 \text{ mg} \cdot \text{L}^{-1}$, with a simulated average of $1.02 \text{ mg} \cdot \text{L}^{-1}$, showing even closer correspondence.

To further analyze simulation result rationality, the Nash-Sutcliffe simulation efficiency coefficient (NSC) was employed to evaluate model performance using Equation (9): $NSC = 1 - \frac{\sum(X_{\text{obs}} - X_{\text{calc}})^2}{\sum(X_{\text{obs}} - \bar{X}_{\text{obs}})^2}$, where X_{obs} represents measured values, X_{calc} represents model-calculated values, and \bar{X}_{obs} is the arithmetic mean of measured values. When calculated values equal monitored values, $NSC = 1$, indicating optimal simulation. Typically, NSC ranges between 0 and 1, with larger values indicating better matching between calculated and observed values. Applying Equation (9), the ammonia nitrogen simulation efficiency coefficient was 0.87 and the nitrate nitrogen NSC was 0.93, both indicating acceptable simulation results (greater than 0.75), with slightly higher accuracy for nitrate nitrogen than ammonia nitrogen.

4 Discussion and Conclusions

Currently, few mathematical models specifically describe water and pollutant transport in farmland drainage ditch systems, with most models developed

based on lake or river models. In the 1990s, based on the shallow lake eutrophication model PCLake, Wageningen University in the Netherlands developed the PCDitch model [15-16]. This model includes modules for nutrient cycling, algae, aquatic plants, benthic organisms, zooplankton, and fish, covering both water and sediment zones, and serves as a eutrophication model for simulating water environment changes in farmland ditches and their biological impacts. Subsequently, the Waterpas-models could predict impacts of ditch water level changes and farming practices on agricultural productivity, income, and the environment [17], finding that excessively high ditch water levels not only reduce crop yields and farmer income but also increase nutrient loads in surface water, thereby exacerbating regional agricultural non-point source pollution. In China, due to the “tree-type” hierarchical layout of irrigation district drainage ditch systems, similar to linear river networks, simulations of water and pollutants in ditch systems often borrow from river water quality and quantity models. Water quantity simulation employs the flow continuity equation, while pollutants are simulated using one-dimensional non-conservative pollutant transport-transformation equations [18-19], achieving good simulation results and becoming the primary models for ditch system water and pollutant transport simulation.

The greatest challenge in applying flow continuity equations and non-conservative pollutant transport-transformation equations is the requirement for substantial ditch experimental monitoring data for validation and calibration, with the key difficulty lying in determining the typical pollutant comprehensive attenuation coefficient K . As shown in [Figure 2: see original paper], nitrogen migration and transformation processes in the water-sediment-plant system of drainage ditches are highly complex. Reference [18] categorized factors influencing K into six types, suggesting they are difficult to describe individually, and employed inverse problem-solving techniques to determine K . Reference [19] also considered K as a comprehensive effect of multiple processes for agricultural non-point source solutes in drainage ditch systems, including pollutant settling in water, resuspension of settled pollutants, sediment interception and adsorption of water pollutants, and other processes, noting that these processes are difficult to monitor and distinguish in practice. Therefore, the comprehensive effect is generally adopted, with K determined through a two-point method using experimental data from typical ditch sections at different levels. These approaches essentially normalize various migration and transformation processes in the water-sediment-plant composite ecosystem, failing to distinguish the mechanism roles of individual ecological components. In this study, based on nitrogen migration and transformation characteristics in the water-sediment-plant system, the comprehensive attenuation coefficient K was further subdivided into three components: K_w (pollutant attenuation caused by water body movement), K_m (pollutant attenuation caused by bottom sediments), and K_p (pollutant attenuation caused by aquatic plants), with parameters determined through simulation experiments. This approach provides deeper insight compared to previous comprehensive effect studies and

better aligns with interaction mechanisms within and between individual media in the water-sediment-plant system.

Simulation results indicate that reference [19] applied Equations (1) and (2) to simulate water quantity and pollutants in the first drainage ditch of the Qingtongxia Irrigation District, achieving a Nash-Sutcliffe efficiency coefficient of 0.82. This study simplified Equations (1) and (2) to Equations (3) and (4) based on actual ditch conditions, simulating ammonia nitrogen and nitrate nitrogen transport in the research ditch, with efficiency coefficients of 0.87 and 0.93, respectively, demonstrating good simulation performance.

In summary, the conclusions of this study are: (1) Based on analysis of the unique ecological structure of farmland drainage ditches, the migration and transformation processes of non-point source nitrogen solutes within the water-sediment-plant system were proposed. (2) Combining characteristics of water-pollutant transport in farmland drainage ditches, a water-pollutant transport model for farmland drainage ditches was preliminarily constructed based on flow continuity and pollutant migration-transformation equations, with a quantitative calculation method proposed for assessing the influence of different water-sediment-plant media on pollutant attenuation through monitoring experiments. (3) The model and calculated parameters were applied to a field experimental ditch, with ammonia nitrogen and nitrate nitrogen as examples. Results showed good agreement between simulated and measured values, with ammonia nitrogen simulation efficiency coefficient of 0.87 and nitrate nitrogen coefficient of 0.93, demonstrating the applicability of the established model and parameter calculation results.

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