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Performance, stability, and cost-effectiveness of a bioreactor-biochar (B^2) system for nutrient removal from agricultural drainage

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ABSTRACT

Extensive tile drainage systems in the Midwestern United States are a major source of nutrient pollution, contributing to water quality impairment in downstream watersheds. This study presents an integrated evaluation of an innovative two-stage woodchip bioreactor-biochar (B^2) treatment system for reducing nitrogen (N) and phosphorus (P) losses from tile-drained croplands by combining laboratory studies, field trials, and a techno-economic assessment (TEA). Laboratory experiments showed that designer biochar pellets produced from sawdust pretreated with lime sludge significantly enhanced the adsorption capacity for dissolved reactive phosphorus (DRP, water-soluble orthophosphate) compared with that of lime sludge alone. In a one-year field trial, the B^2 system demonstrated sustained nutrient removal when treating 3, 018 m^3 of drainage water. The woodchip bioreactor reduced nitrate-nitrogen (NO_3-N) concentrations by 58 % with a cumulative load reduction of 1.8 kg. Ammonium-nitrogen (NH_4-N) loads were reduced from 2.83 to 0.73 kg, with removal efficiency increasing from 64 % to 72 % under the subsequent biochar treatment. Biochar sorption channels reduced DRP by 3–92 % (median 69 %) and total P by 20–94 % (median 55 %), effectively mitigating DRP and TP leaching observed in the woodchip bioreactor effluent. The TEA indicated that the pilot-scale B^2 system achieved unit removal costs of \$90.3/kg NO_3-N/year and \$63.9/kg DRP/year. When the system was scaled to treat drainage water from a 10-ha drainage area, the system yielded average removal costs of \$4.7 $\pm$ 1.9/kg NO_3-N/year and \$103.7 $\pm$ 153.5/kg DRP/year, with an annualized system cost of \$1020.2 $\pm$ 80.4 per year. The TEA analysis also suggested that the cost-effectiveness of the B^2 systems can be further improved through strategic site selection, material sourcing, and flow management. Overall, these results highlight the B^2 system as a practical, scalable, and cost-effective strategy for improving water quality in tile-drained agricultural landscapes.

1. Introduction

Intensified agricultural activities are placing unprecedented pressure on ecosystems, resulting in the degradation of water quality [1,2]. In the Midwestern United States, subsurface tile drainage systems are widely used to improve field trafficability by removing excess water from the soil profile [3,4]. However, previous studies have shown that these systems are the primary contributors to nutrient losses, primarily nitrogen (N) and phosphorus (P), from agricultural fields into nearby water bodies [5,6]. According to the Measured Annual Nutrient loads from AGricultural Environments (MANAGE) database, the average NO_3-N concentration in tile drainage is approximately 12.9 mg/L [7]. In contrast, total P has been reported at ~0.1 mg/L and dissolved reactive P (DRP) at ~0.01 mg/L in monitoring studies across the region, although higher values (e.g., 1.17 mg/L) have been observed in fields when receiving manure amendments [8]. Reports indicate that tile-drained systems can increase P losses by up to 80 % and N losses by up to 43 % compared to undrained croplands. [9–11].

Over the past decades, efforts have been devoted to developing low-cost, sustainable, and low-maintenance edge-of-field technologies, such as woodchip bioreactors, P removal structures, saturated buffers, and constructed wetlands, to reduce nutrient losses from agricultural drainage into surrounding watersheds [12,13]. Among these, denitrifying woodchip bioreactors have proven to be practical and cost-effective solutions for N removal from the drainage water and agricultural runoff [14,15]. According to the US Department of Agriculture's

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(USDA) Conservation Practice (Standard No. 605) [63], these bioreactors use woodchips or other biomass as carbon sources to facilitate denitrification, a microbial process that converts nitrate-nitrogen (NO_3-N) to inert nitrogen gas (N_2) [16]. However, in addition to denitrification, ammonium-nitrogen (NH_4-N) can also be generated via dissimilatory nitrate reduction to ammonium (DNRA), although this process is generally minor in woodchip reactors except during the start-up phase when readily leachable carbon is abundant [17–19]. This outcome is undesirable in the bioreactor systems since NH_4-N more readily promotes algal growth than NO_3-N. Furthermore, while effective for NO_3-N removal, woodchip bioreactors have limited capacity for P removal [20]. In many cases, elevated concentrations of dissolved reactive P (DRP, water-soluble orthophosphate) have been observed following treatment with biomass-based bioreactors during the start-up phase [21,22], suggesting that woodchip bioreactors may act as a source of P release during certain operational periods.

To effectively and comprehensively reduce excess nutrient losses, two-stage or multi-stage systems that integrate woodchip bioreactors with complementary treatment technologies, such as P removal structures, have been developed [23–28]. For instance, Abdi et al. [23] developed a two-stage system that combined woodchip bioreactors with secondary expanded shale aggregate filters, achieving over 99 % nitrate removal and 80–87 % DRP reduction. Biochar, a carbon-rich material produced from biomass by pyrolysis, has demonstrated strong sorption capacity for nutrients and emerging contaminants, making it a promising sorbent for water quality management [29,30]. Recent studies have shown the synergistic benefits of combining woodchip bioreactors with biochar-based treatment systems. For example, researchers demonstrated sequential removal of nitrate and sulfate in a bioreactor combining woodchips and hematite-coated biochar [31]. Our previous study demonstrated that combining woodchip bioreactors with biochar treatment systems can simultaneously remove nutrients and some emerging contaminants [28]. Compared to the woodchip bioreactor alone, the two-stage woodchip bioreactors and biochar treatment systems achieved removal efficiencies exceeding 75 % for nitrate, 99.03 % for DRP, 69.51 % for ibuprofen, 73.65 % for naproxen, 91.09 % for sitagliptin, and 96.96 % for estrone [28]. Despite encouraging laboratory-scale results [28,31], the practical feasibility and long-term performance of these two-stage woodchip bioreactor–biochar systems at the field scale remain poorly understood. To the best of our knowledge, although pilot-scale two-stage treatment systems have been evaluated [21]; a field-scale evaluation specifically integrating woodchip bioreactors with biochar treatment units has not yet been reported. More importantly, comprehensive techno-economic analyses are lacking, limiting the ability of stakeholders and decision-makers to evaluate the scalability and economic feasibility of these systems at watershed or regional levels. Addressing these knowledge gaps is crucial for developing engineering-scale system designs and promoting the broader adoption of two-stage treatment systems in the integrated water quality management strategies.

In light of these challenges, the overarching objective of this study was to develop and field-test an innovative woodchip bioreactor and biochar (B^2) treatment system to remove nutrients from tile drainage water in an agricultural field. The specific objectives were to (1) manufacture field-applicable designer biochar pellets that can efficiently capture DRP; (2) design, construct, and evaluate a field-scale B^2 nutrient treatment system and evaluate its performance in mitigating N and P losses under real-world conditions; (3) assess the cost-effectiveness of the B^2 system for mitigating excess nutrient export from tile-drained agricultural fields.

2. Materials and methods

2.1.1. Manufacturer of designer biochar pellets and their laboratory evaluation

In this study, pine-derived sawdust (*Pinus* spp.) was thoroughly mixed with lime sludge at a 1:4 (w/w, additive to biomass) ratio for producing designer biochar. Based on our previous work, lime sludge, which is predominantly composed of precipitated calcium (Ca) and magnesium (Mg), served as an effective modifying agent that enhanced the physicochemical properties of the resulting designer biochar [32]. The lime sludge was collected from a local drinking water treatment facility in Champaign, Illinois, and its physicochemical properties were characterized (as shown in Table S1). In addition to its role as a Ca source, the lime sludge also functioned as a binder, which enabled the agglomeration of biochar powder into stable pellets. The sawdust with a particle size of less than 1.0 mm was pre-mixed with lime sludge and then pyrolyzed to produce designer biochar powder. Prior laboratory studies demonstrated that pretreating sawdust with calcium-rich materials significantly enhanced the P removal capacity of the powdered designer biochar, with the formation of Ca—P precipitates on the biochar surface identified as the predominant mechanism [32,33]. To improve the mechanical strength and engineering value of the designer biochar for field-scale application, a pelleting process was performed. The powdered biochar was pelletized using a MILL-10 Pellet Mill (Colorado Mill Equipment, USA), yielding uniform pellets approximately 0.6 cm in diameter and 1 cm in length. The main physicochemical properties of the designer biochar pellets have been detailed in Supplementary Table S1.

To assess the DRP sorption capacity of the designer biochar pellets, a series of laboratory batch experiments were performed. Briefly, 0.10 $\pm$ 0.01 g of air-dried biochar pellets was added to 100 mL of KH_2PO_4 solutions prepared with deionized water in 150 mL acid-washed glass containers, and the mixtures were reciprocally shaken at 180 rpm for 24 h at room temperature (22 °C). The initial pH of the solutions was adjusted to ~7.0 and monitored throughout the experiments. A series of KH_2PO_4 concentrations ranging from 0.1 to 500 mg P/L was tested to evaluate phosphorus sorption isotherms, and all experiments were conducted in triplicate. The lime sludge as a phosphorus sorption material (PSM) was tested in parallel for comparison using the same conditions. The DRP concentrations were determined following the analytical method described in our previous study [33]. The suspensions were filtered through 0.45 μm membrane filters, and the DRP concentrations in the filtrates were determined using the molybdenum blue colorimetric method. Quality control included calibration with KH_2PO_4 standards and duplicate analyses every ten samples to ensure analytical precision.

2.2. Field demonstration of woodchip bioreactor - biochar (B^2) system

A field-scale study of a two-stage woodchip B^2 treatment system was conducted on an active one-hectare farmland at the University of Illinois Urbana-Champaign, Urbana, USA (40.093944° N, −88.223469° W) from December 2021 through August 2022. Due to the budgetary constraints and manageable workload limitations, the study was implemented as a single field-scale trial. The study site is located in a humid continental climate zone, with an average annual precipitation of approximately 1000 mm, most of which occurs during the growing season. The tile water temperature ranged from 5 °C to 24 °C with an average of 16 °C. In addition, a substantial fraction of early spring flows through the bioreactor is associated with snowmelt, which typically precedes crop growth and contributes to elevated drainage volumes. These climatic conditions are typical of the Midwestern United States and can influence both agricultural runoff patterns and the performance of field-based treatment systems. The soil at the site is classified as Drummer silty clay loam, which is characterized by relatively poor drainage properties (USDA Soil Taxonomy). The B^2 system was designed to collect and treat drainage water from 1 ha tile-drained cropland (Fig. 1). The woodchip bioreactor system was constructed with a trench 2 m deep with a 4.0 × 1.0 m footprint, filled with a 1.5 m layer of woodchips, and covered with 0.5 m of topsoil. The fresh woodchips were Illinois-grown hardwood species - specifically white oak (*Quercus alba*), which were approved by the Natural Resources Conservation Service (NRCS) for use in woodchip bioreactors in Illinois. Following the woodchip bioreactor, two sequential biochar channels (impermeable polyethylene pipes, 1.8 m in length and 0.15 m in diameter for each) were installed (Fig. 1). The biochar channels 1 and 2 contained 12 kg and 14 kg of designer biochar pellets, respectively.

To evaluate nutrient removal performance, water samples were collected 1–2 times per week and within 24 h after rainfall events exceeding 0.5 in. Samples were obtained from four locations along the treatment system: the woodchip bioreactor inlet (D1), the bioreactor outlet (D2), and the outlets of the two biochar channels (D3 and D4), respectively (Figs. 1, S1c). This sampling strategy allowed for the monitoring of nutrient concentrations at each treatment stage and facilitated the assessment of nutrient re- moval by each individual treatment component. For effective denitrification, it is essential that the water level at the bioreactor inlet (D1) remains lower than at the outlet (D2) to create an anaerobic zone that promotes nitrate reduction. All collected samples were stored in pre-cleaned 50 mL HDPE bottles, kept at 4 °C during transport, and shipped to the laboratory within 24 h for analysis. Samples were analyzed for NO_3-N, NH_4-N, DRP, and total phosphorus (TP) following standard EPA-approved methods [34,35]. Additional details on the water sample collection methods, water flow measurement, and nutrient load calculations can be found in a previous study [21]. Briefly, flow-weighted mean concentrations were used to estimate daily and cumulative nutrient loads, which enabled quantification of the treatment system's overall performance under varying hydrological conditions. Daily nutrient loads were calculated by linearly interpolating concentrations between consecutive sampling events and integrating these interpolated values with measured flow data. Removal efficiency for each sampling event was determined by comparing influent and effluent nutrient concentrations collected on the same day, providing event-specific measures of treatment performance. This approach, widely adopted in bioreactor and agricultural drainage studies [36–38], provides a reasonable approximation of cumulative nutrient exports across the monitoring period. Drainage flow at the inlet and outlet of each system was continuously recorded at 15-min intervals with HOBO pressure transducers (HOBO U20–001-01, Onset Computer Corporation, Bourne, USA) connected to dataloggers, in combination with V-notch weirs (90°) placed in the control structures. Water stage measurements were converted to flow rates using standard hydraulic relationships for V-notch weirs, as described in Oladeji et al. [21].

$$Q = 1.7406 \cdot H^{1.9531} \tag{1}$$

where Q is the discharge (L $\min^{-1}$), and H is the hydraulic head (cm) above the V-notch weir crest. All flow and concentration data were synchronized by timestamp to ensure consistent load calculations.

2.3. Cost analysis

The cost-effectiveness of the B^2 system was evaluated using Present Value (PV) cost and the unit removal cost of nutrients, expressed in $/kg DRP and $/kg NO_3-N. A cash flow discounting approach was applied to calculate the total PV cost and annualized cost of the B^2 system, using a 2.5 % discount rate and a 12-year service life, based on previous studies on bioreactor-based conservation practices [39,40]. Furthermore, Zhou et al. [30] indicated that, for biochar-based phosphorus removal structures, replacing the biochar once every two years represents a reasonable balance between maintaining removal efficiency and minimizing replacement costs. Therefore, in our current study, the biochar system was designed to replace the material every two years to maintain treatment efficiency. Although woodchips typically have a longer lifespan and do not require complete replacement, we incorporated a 25 % woodchip replenishment every 6 years in the TEA. This assumption was informed by field-scale observations, reporting gradual settling and decomposition of woodchips over time. The details are in the Supplementary materials (Supplementary Text 1). Capital costs include the cost of the structures, excavation, labor, materials, and accessories including pipes and fittings. Operation and maintenance (O&M) costs accounted for estimated expenses related to material replacement, transportation, and labor. Nutrient removal costs were calculated by dividing the annualized system cost (USD/year) by the annual nutrient removal (kg/year). The annual implementation cost was computed by converting the PV cost using the annuity factor.

$$\text{Present Value (PV)} = \sum\nolimits_{t=0}^{n} \frac{C_t}{(1+r)^t} \tag{2}$$

$$\text{Annual cost} = PV^{*}r\frac{(1+r)^n}{(1+r)^n - 1} \tag{3}$$

where C_t, n, and r denote the cost occurring at t years in the future, the service life of the system, and the discount rate respectively.

The cost analysis was conducted for the pilot-scale B^2 system to assess the actual expenditures. In addition, a full-scale system was envisioned to estimate costs for broader environmental applications. The costs of the pilot-scale B^2 system included structures and materials, which were derived from direct invoices and information obtained through personal communication with the farm owner. These costs were treated as PV costs.

The full-scale system was evaluated to reflect realistic field conditions, accounting for uncertainties that may influence both nutrient removal performance and economic feasibility of the B^2 system. For the full-scale B^2 system, the pilot-scale design was scaled up by a factor of 10 based on treatment capacity (woodchip bioreactor and biochar pellets) to serve a tile-drained field of approximately 10 ha. The scaling included

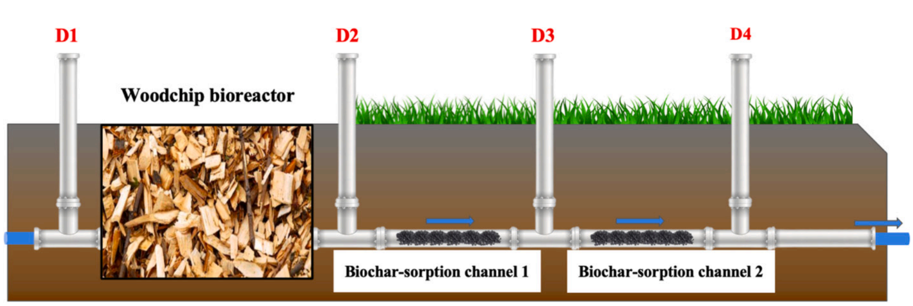

Fig. 1. Schematic diagram of woodchip bioreactor and biochar (B2) treatment system. Sampling locations include: D1 – influent; D2 – effluent from the woodchip bioreactor; D3 – effluent from the first biochar-sorption channel; and D4 – final effluent from the B^2 treatment system.

adjustments to the quantity of the filter materials (woodchips and biochar pellets) and the corresponding dimensions of each treatment unit. In the full-scale design, the woodchip bioreactor had a volume of 60 m^3, while the biochar unit had a volume of 8 m^3 and contained 225 kg of biochar pellets. The scaling reflects the typical size of woodchip bioreactors in the USA, which are generally designed to treat drainage water from 10 to 20 ha fields [40]. The details and justification of the design assumptions are provided in Supplementary Text SX1.

To assess how variability in key parameters affects the system's economic feasibility and treatment performance, a Monte Carlo simulation (4000 iterations were performed) was conducted to evaluate the PV cost and unit nutrient removal cost under uncertainty. Eight input variables were randomly sampled using a triangular probability distribution, defined by three parameters: a minimum value, a maximum value, and the most likely value. These variables included the influent concentrations of NO_3-N, DRP, daily treatment flow, number of flow days in a year, removal efficiencies of NO_3-N and DRP, and materials cost (woodchips and biochar pellets). The probability density function of the triangular distribution increases linearly from the minimum to the most likely value and then decreases linearly to the maximum. The details of the baseline assumptions and range of parameters are provided in Supplementary Table S2, with detailed justifications in Supplementary Text S1. We acknowledge that flow rate, influent NO_3^- concentration, water temperature, and removal efficiency are interdependent in woodchip bioreactor systems [41,42]. However, the Monte Carlo analysis was applied to the TEA to estimate potential variability in system performance and associated economic outcomes under plausible field conditions, rather than a mechanistic prediction of system behavior. Input distributions were based on field measurements and literature, providing a first-order approximation of uncertainty and highlighting key parameters for future process-based studies.

2.4. Data analysis

The statistical analysis was conducted using R, an open-source software for statistical computing. To test the significant differences in mean values and nutrient removal efficiencies, a paired *t*-test was used. If the *p*-value was less than 0.05 ($\alpha = 0.05$), the means of the two datasets were significantly different. In addition, a Spearman correlation analysis was conducted to examine the relationships between eight input parameters and three output variables: the PV of the system cost, the unit cost of N removal, and the unit cost of P removal.

3. Results and discussion

3.1. Performance of designer biochar pellets for DRP removal capacity and adsorption mechanism

Calcium-enriched materials such as lime sludge, steel slag, and gypsum, have been extensively used for DRP removal from aqueous solutions [43]. The DRP adsorption capacities of lime sludge and designer biochar pellets were compared through controlled laboratory-scale batch experiments. The results revealed that the lime sludge-enriched designer biochar exhibited significantly higher DRP adsorption capacity than lime sludge alone (Fig. 2). Under optimized conditions, the designer biochar pellets achieved a maximum adsorption capacity exceeding 400 mg/g, which was approximately 20 times greater than that of lime sludge under the same experimental conditions.

Lime sludge is primarily composed of calcium carbonate ($CaCO_3$), with trace amounts of magnesium (Mg) and other minerals, and typically exhibits a pH of around 9. In natural waters (pH 5–7), the dominant phosphate species is dihydrogen phosphate ($H_2PO_4^-$). As the pH increases to around 9, phosphate speciation shifts toward hydrogen phosphate (HPO_4^{2-}) (Fig. 3). At this pH, Ca^{2+} and Mg^{2+} ions released from lime sludge can readily react with HPO_4^{2-} to form insoluble precipitates such as $Ca_3(PO_4)_2$ and $Mg_3(PO_4)_2$ [32].

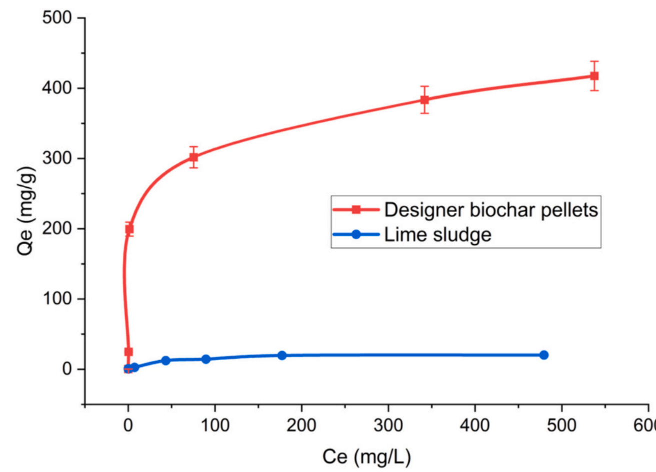

Fig. 2. Adsorption isotherms for DRP on the designer biochar and lime sludge.

In contrast, designer biochar pellets pyrolyzed from lime sludge pretreated biomass exhibit a substantially higher surface pH, reaching up to 12 under our experiment. This strong alkalinity originates from the thermal decomposition of $CaCO_3$ during pyrolysis, which produces reactive alkaline oxides that persist on the biochar surface. At this elevated pH, phosphate species further shift toward HPO_4^{2-} and phosphate (PO_4^{3-}), thereby enhancing additional precipitation reactions. The designer biochar retains a high surface concentration of Ca^{2+} and Mg^{2+} ions, which not only form $CaHPO_4$ and $MgHPO_4$ with HPO_4^{2-} but also react with PO_4^{3-} to produce more stable and less soluble minerals such as $Ca_3(PO_4)_2$ and $Mg_3(PO_4)_2$. Similar mechanisms have been observed in engineered Ca/Mg-rich biochars, where precipitation of dissolved P with metal ions significantly enhanced P removal [32]. Given the extremely low solubility product (*Ksp*) of $Ca_3(PO_4)_2$ (2.0×10^{-29} at 25 °C) compared to $CaHPO_4$, these additional precipitation-reactions significantly enhance DRP removal compared to lime sludge. In addition to the chemical precipitation mechanisms, the highly porous structure of the biochar pellets may help facilitate DRP removal (Fig. S1b). As a result, the designer biochar pellets demonstrate a substantially higher DRP sorption capacity than lime sludge, indicating their enhanced performance as a PSM for capturing DRP from drainage waters.

3.2. Field-scale performance of the B^2 treatment systems

3.2.1. Nitrogen removal performance of the B^2 treatment system

One-year monitoring of the field-scale B^2 treatment system demonstrated effective nitrogen removal from agricultural drainage. A total of 3018 m^3 of drainage water was treated over the monitoring period, equivalent to 302 mm of drainage depth. The average drainage flow through the system was 4.47 m^3/day, with a maximum flow capped at 7.2 m^3/day. Flow occasionally dropped to zero during periods of no inflow. The average hydraulic retention time (HRT) in the woodchip bioreactor was 9.8 ± 5.7 h, while the retention time in the biochar system was approximately 4.6 ± 2.9 min. As water flowed through the treatment system, total NO_3-N loads decreased substantially (Fig. 4). Specifically, the total NO_3-N load decreased from 2.69 kg at the influent (D1) to 0.90 kg after the woodchip bioreactor (D2), 0.78 kg after the first biochar-sorption channel (D3), and 0.50 kg after the second biochar-sorption channel (D4) (Fig. 4a). The woodchip bioreactor played a dominant role in nitrate reduction, achieving an event-based mean removal efficiency of 58 % and a cumulative NO_3–N load reduction of 1.8 kg (Fig. 4a, b). However, NO_3-N removal efficiency varied, ranging from 9 % to 98 % (Fig. 4b). Such high variability in field-scale denitrifying bioreactors has been observed across multiple sites, typically attributed to fluctuations in ambient influent nutrient concentrations,

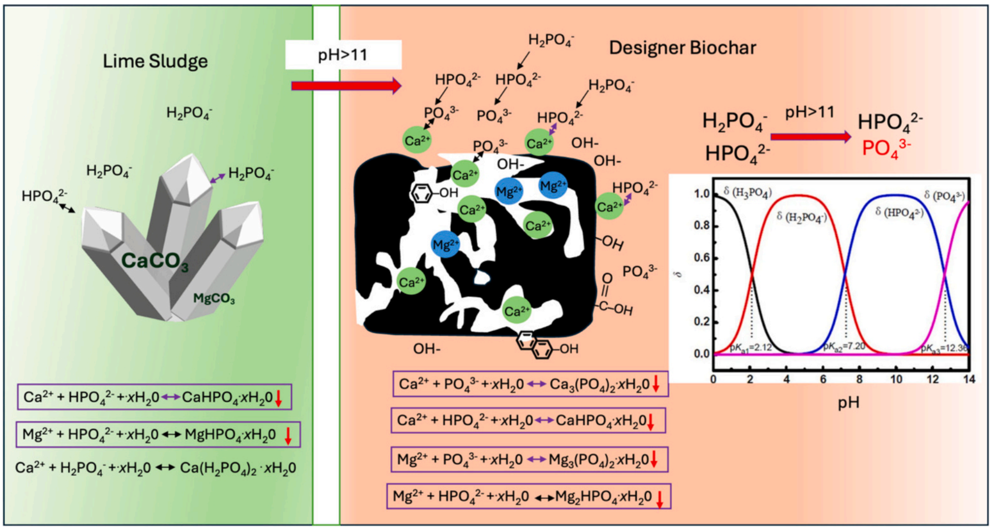

Fig. 3. Adsorption mechanisms for phosphate species on the lime sludge and designer biochar pellets.

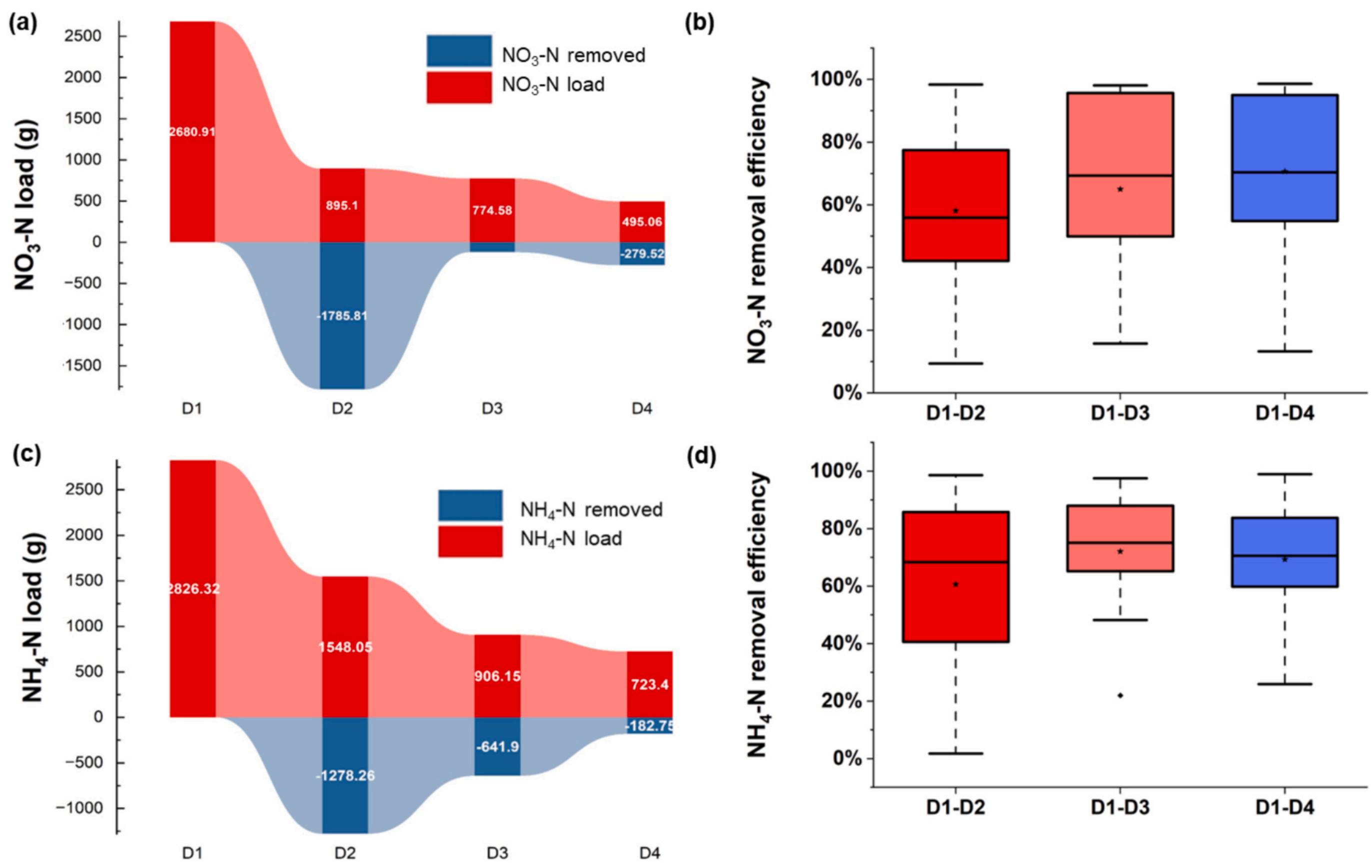

Fig. 4. N removal performance of the field-scale B^2 system from tile drainage, illustrating (a) NO_3-N load removal, (b) NO_3-N removal efficiency, (c) NH_4-N load removal and (d) NH_4-N removal efficiency.

seasonal temperature variation, and internal hydraulic conditions [44,45].

In terms of the nutrient concentration (Fig. S2), the average NO_3-N concentrations at locations D1, D2, D3, and D4 were 3.48 ± 5.03 mg/L, 1.39 ± 1.47 mg/L, 1.18 ± 1.64 mg/L, and 0.91 ± 1.17 mg/L, respectively. However, we also observed an exceptionally high concentration of 140 mg/L, which was primarily attributed to seasonal manure and fertilizer applications, as well as enhanced nitrate mobilization during storm events. Despite this variability, the B^2 treatment system consistently achieved high nitrate removal, with low removal efficiencies (~10 %) observed in only two of 23 sampling events. These low efficiencies were likely associated with short-term hydraulic condition changes, such as preferential flow or high drainage rates that reduced water residence time. Similarly, NH_4-N loads decreased as drainage water passed through the treatment system (Fig. S3). The cumulative NH_4-N load was reduced from 2.83 kg at the influent (D1) to 1.55 kg, 0.91 kg, and 0.73 kg at D2, D3 and D4, respectively (Fig. 4c), primarily due to nitrification, whereby NH_4^+ is oxidized to NO_3^- under aerobic conditions favorable for microbial activity. The average NH_4^+-N concentrations at the same sampling points D1, D2, D3, and D4 were 2.65 ± 2.78 mg/L, 1.87 ± 2.20 mg/L, 0.86 ± 0.61 mg/L, and 0.82 ± 0.59 mg/L, respectively. Mechanistically, nitrate reduction in the woodchip bioreactor primarily occurs through heterotrophic denitrification, in which nitrate is reduced to gaseous N_2 by anaerobic microorganisms using organic carbon from woodchips as the electron donor. The efficiency of this process is highly dependent on the HRT, dissolved oxygen (DO), temperature, and available carbon [19,44,45]. During periods of high flow, short HRT and elevated DO levels likely limited complete denitrification, resulting in partial NO_3^- removal or accumulation of intermediates such as N_2O. Conversely, higher removal rates (up to 98 %) were observed during warmer months, consistent with enhanced microbial activity and increased carbon availability.

The biochar channels contributed less to the total N load reduction in comparison with woodchip bioreactors, but their application further improved overall system performance. The average NO_3-N removal efficiency increased from 57 ± 20 % after the bioreactor to 63 ± 22 % and 69 ± 21 % following the first and second biochar channels, respectively (Fig. 4b), indicating that the biochar treatment enhanced NO_3-N removal. This improvement may be attributed to the continued denitrification within the biochar sorption channel, since the capacity of biochar for sorbing NO_3-N is rather limited. While biochar may also facilitate microbial nitrate reduction by providing a favorable habitat and enhancing electron transfer within the carbon matrix, such microbial activity is unlikely to be significant in the current system due to the short hydraulic retention time. A similar trend was observed for NH_4-N removal, with an efficiency increasing from 61 ± 32 % after the bioreactor to 72 ± 20 % and 69 ± 22 % following the first and second biochar channels, respectively (Fig. 4d).

3.2.2. P Removal performance of the B^2 treatment system

Despite the high N removal performance of the woodchip bioreactor system, substantial P leaching was observed. Field monitoring revealed that the cumulative DRP and TP significantly increased after the drainage water passed through the woodchip bioreactor during our experimental period (Fig. 5a–b). Specifically, DRP loads increased from 0.9 kg at the influent to 4.0 kg at the bioreactor effluent, while TP loads rose from 6.25 kg to 7.72 kg (Fig. 5a–b). The median influent concentrations were 0.055 mg/L for DRP and 0.22 mg/L for TP. In contrast, effluent concentrations from the bioreactor were considerably higher, with median values of 0.24 mg/L for DRP and 0.52 mg/L for TP. In several events, peak TP and DRP concentrations reached 40 mg/L and 4.77 mg/L, respectively, coinciding with periods of elevated influent nutrient loading (TP = 6.55 mg/L, DRP = 0.35 mg/L) and high hydraulic flushing (Figs. S4–S5). P release from woodchip bioreactors has been widely documented, particularly during the startup period, which typically lasts from several months to a year [28,46]. During the startup period, freshly installed woodchips generally release more carbon and P, whereas older woodchips tend to release less and may have acquired additional sorption capacity through interactions with suspended minerals in drainage water. This early-stage leaching is mainly attributed to (i) the desorption of inorganic P previously adsorbed on wood surfaces, (ii) the mineralization of organic P compounds contained in the wood biomass, and (iii) reductive dissolution of Fe- or Mn-bound P under anoxic conditions within the bioreactor [22,47,48]. Therefore, when fresh woodchips are used, it is necessary to implement management practices - such as secondary treatment system, pre-soaking, or controlled flushing - to minimize the risk of elevated P leaching during the early operational stage.

The incorporation of biochar channels significantly reduced both

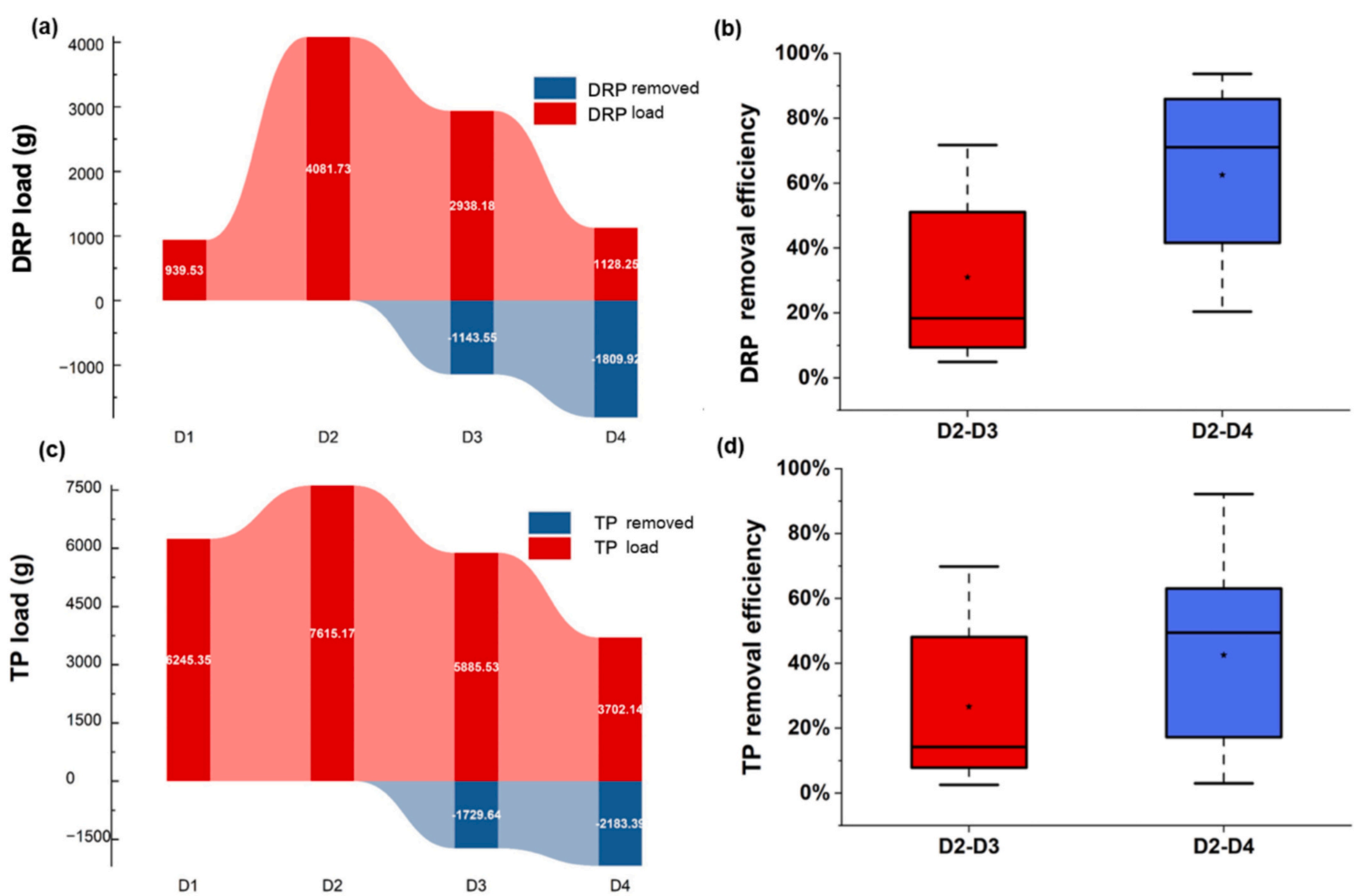

Fig. 5. P removal performance of the field-scale B^2 system treating tile drainage, illustrating (a) changes in DRP load, (b) DRP removal efficiency, (c) changes in TP load, and (d) TP removal efficiency.

DRP and TP concentrations in drainage water. After drainage water passed through the two sequential biochar sorption channels, DRP removal efficiencies in the first channel ranged from 3 % to 70 %, with a mean efficiency of 31 ± 23 % (Fig. 5b). The second channel achieved higher DRP removal, ranging from 3 % to 92 %, with a mean efficiency of 63 ± 26 % (Fig. 5b). The occasional low removal rates (e.g., ~3 %) were likely associated with preferential flow and limited water-biochar contact, which restricted sorption equilibrium within the packed pellet matrix. Similarly, TP loads decreased to 5.89 kg and 3.70 kg, respectively (Fig. 5d). The first-stage biochar system exhibited a TP removal efficiency between 5 % to 72 %, with a mean of 25 ± 24 %. TP removal was further enhanced in the second-stage system, ranging from 20 % to 94 % and achieving a mean efficiency of 42 ± 28 % (Fig. 5d). This performance improvement can be attributed to both physicochemical and hydraulic factors. The first-stage biochar primarily acts as a rapid-contact filter, capturing DRP through surface complexation and partial precipitation with Ca^{2+}/Mg^{2+} species. The second-stage biochar, receiving pre-treated effluent with reduced suspended solids, operates under a more stable flow regime and longer effective contact time, thereby favoring continued precipitation within internal pores.

Overall, the field-scale demonstration highlights that the B^2 treatment system - consisting of a woodchip bioreactor followed by biochar sorption channels - not only sustains substantial N and P removal performance, but also effectively mitigates P leaching, a common challenge associated with conventional woodchip bioreactors during the start-up stage.

3.3. Cost analysis

Estimating costs of proposed strategies can help guide investment decisions and inform incentive payments to landowners. The total PV cost of the pilot-scale B^2 system, designed to treat drainage water from a nearly 1-ha field, was estimated at $4086.0 (Table S3). Woodchip bioreactor components accounted for $2037.7, while biochar components constituted $2049.3. The annual cost of the treatment was $399.3/yr, based on an expected effective service life of 12 years. The unit removal costs of NO_3-N and DRP were $90.3/kg NO_3-N and $63.9/kg DRP per year, corresponding to observed nutrient removal rates of 2.2 kg NO_3-N/yr and 3.1 kg DRP/yr. At the pilot scale, the unit cost of N removal was substantially higher than values reported for full-scale bioreactors deployed at the tile-drained field [40] and spring bioreactors [39]. On the contrary, the cost for DRP removal in this pilot-scale B^2 system was considerably lower compared to existing P removal structures [43]. This discrepancy likely reflects the relatively low N inflow and elevated P inflow during this pilot-scale field experiment, possibly due to P fertilizer application before planting soybean crops.

To ensure broader applicability, a cost analysis was further conducted for the scaled-up B^2 system based on nutrient concentration ranges typical of tile-drained systems in the Midwestern states (Table S2). This analysis provides a more realistic assessment of the economic feasibility and scalability of B^2 system for nutrient management in agricultural landscapes. The total PV cost of the scaled-up B^2 system was $10,464.8 ± 824.5, with an annualized cost of $1020.2 ± 80.4/year (Table S4), less than three times the implementation cost of the pilot-scale system despite being ten times larger in size. These results reflect the economic advantage of scaling up for larger drainage areas, as fixed costs, such as control structures, are distributed over a greater treatment capacity, reducing the per-unit treatment cost. The costs of designer biochar and woodchip biomass are key factors influencing the overall system cost (Fig. 6). The cost of woodchips shows a strong positive correlation with system PV cost ($r = 0.99$), highlighting its economic impact, while biochar cost has minimal effect ($r = 0.26$). Despite the higher unit cost of designer biochar pellets ($300 ~ $700/ton, Supplementary Text 1) is higher than that of woodchip biomass (10/m^3-$70/$m^3$, Supplementary Text 1), and requires regular replacement, their impact on the system cost is small due to the relatively small biochar quantity needed compared to woodchip biomass. Conversely, the large volume of woodchip media required explains its strong correlation with system cost, consistent with observations across multiple bioreactor-based systems [40,49]. Therefore, strategic sourcing of woodchips, such as the selection of woodchip types that do not compete directly with high-demand market wood resources, could play a crucial role in reducing overall system costs, enhancing the economic feasibility, and promoting broader adoption of this treatment system.

The average cost of removing NO_3-N and DRP was $4.7 ± 1.9/kg

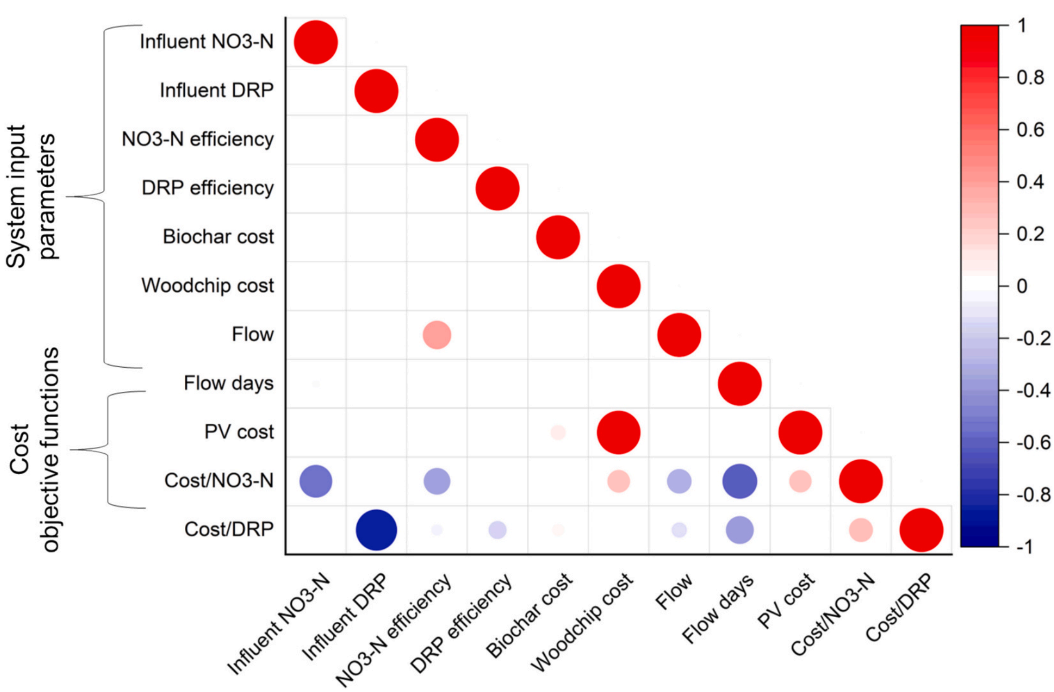

Fig. 6. Spearman correlation analysis of the cost estimates of the system with multiple input parameters.

NO_3-N/yr and \$103.7 ± 153.5/kg DRP/yr, respectively. These estimated costs are based on annual N and P removal rates of 174.9 ± 66.4 kg NO_3-N/yr and 4.9 ± 2.9 kg DRP/yr from a 10-ha drainage area, with an annualized system cost of \$1020.2 ± 80.4. The unit cost of nutrient removal is strongly influenced by the annual nutrient load reduction, which is determined by influent concentration, flow rate, and removal efficiency (Table S2). Although nutrient removal efficiency is expected to improve cost-effectiveness, the weak negative correlations between removal efficiency and unit removal cost ($r = -0.37$ for cost/NO_3-N, $r = -0.17$ for cost/DRP) suggest efficiency alone has a limited impact on reducing costs under typical conditions as discussed in Supplementary Text 1. Instead, nutrient removal costs are more strongly affected by influent concentrations, as indicated by stronger negative correlations ($r = -0.56$ for cost/NO_3-N, $r = -0.86$ for cost/DRP). Therefore, applying B^2 systems in nutrient hotspot zones, such as areas with concentrated tile drainage discharges or historically high nutrient loads, may offer greater cost-effectiveness and return on investment.

Similarly, the number of flow days per year shows a significant negative correlation with the unit cost of nutrient removal ($r = -0.63$ for NO_3-N and $r = -0.40$ for DRP), suggesting that systems that operate over more days annually can achieve lower unit removal costs. In other words, treatment systems installed at sites with more consistently distributed flow tend to be more cost-effective than those located in sites characterized by short-duration, high-intensity flow events. Our analysis assumes a uniform flow through the B^2 system throughout the year, which is an idealized condition that can typically be managed using control structures and bypass mechanisms during high-flow events. However, during low-flow seasons, the volume of drainage water and nutrients treated may decrease, potentially reducing cost efficiency. Although the current TEA does not fully account for flow variability under low flow, the results suggest that integrating flow variability into cost models could provide more realistic and informative economic assessments. Moreover, implementation strategies that enhance flow duration, including optimized control structure management and prioritizing installations in areas with sustained tile-drain flow, could further enhance the cost-effectiveness of B^2 systems.

Overall, the cost analysis demonstrates the economic viability and scalability of the B^2 treatment system for nutrient loss reduction in tile-drained croplands. The feasibility of a scale-up system depends not only on design considerations but also on site-specific conditions and operational strategies. Primary cost drivers include the woodchip media and flow distribution patterns. The removal cost of excess nutrients decreases in areas exhibiting higher nutrient loads and longer flow durations. These findings highlight the importance of strategic site selection, appropriate media sourcing, and effective flow management to maximize both economic and environmental benefits, thereby facilitating adoption of B^2 systems as a sustainable and cost-effective conservation practice for nutrient loss reduction.

4. Environmental implications

Effective nutrient management in agricultural landscapes relies on the implementation of efficient and well-designed conservation practices that can reduce nutrient losses while maintaining agricultural productivity and long-term sustainability. This study presents the first field-scale development and systematic evaluation of an innovative, cost-effective, and sustainable B^2 treatment system designed for tile-drained agroecosystems. Through a combination of laboratory experiments, field-scale demonstration, and techno-economic assessment, the B^2 system was shown to cost effectively and efficiently reduce N and P losses from agricultural drainage. Woodchip bioreactors have been widely adopted across tile-drained fields in the U.S. Midwest for nitrate removal. Building on this established edge-of-field practice - particularly through the addition of a secondary biochar-based treatment system - offers a practical, scalable, and low-barrier enhancement to existing technologies. Moreover, the adaptability of the B^2 system extends beyond agricultural drainage, with its design and treatment principles readily applicable to other settings such as stream water restoration and urban stormwater treatment [39,50]. In practice, the decision to adopt B^2 system at a specific site depends on several site-specific factors, including the water quality goals and the trade-off between expected treatment benefits and implementation costs.

Achieving ambitious nutrient loss reduction goals becomes more feasible when the costs of conservation practices are justified by the societal benefits of clean water [51]. According to previous estimates, the economic cost of N loading to coastal waters ranges from \$12 to \$56/kg of N [52–54]. In comparison, the economic damages associated with excess P runoff loss can result in total losses ranging from \$74.5 to over \$36,000/kg of P, depending on the extent of eutrophication, geographic context, and valuation methods [55,56]. Based on our evaluation (Section 3.3), the average cost of nutrient removal using the B^2 treatment system was \$4.7 ± 1.9/kg/yr for N and \$103.7 ± 153.5/kg/yr for P. These values suggest that the B^2 system is a cost-effective approach, especially considering the external damage costs associated with nutrient pollution. Compared with the unit costs of existing edge-of-field technologies, woodchip bioreactors typically achieve N removal at costs ranging from \$2 to \$88/kg N/yr [57,58]. Stormwater control practices may incur much higher costs, ranging from \$660 to \$1540/kg N/yr [59], while most P removal structures exhibit removal costs ranging from \$100 to \$1300/kg P/yr [43]. In comparison, constructed wetlands, while capable of treating both N and P simultaneously, generally demonstrate N removal costs ranging from \$2.85 to \$30/kg N/yr and P retention costs between \$50.42 to \$1116.58/kg P/yr [60,61]. In this context, the B^2 systems offer both competitive cost performance and treatment capacity. Moreover, the use of biochar as a secondary treatment, rather than incorporating it directly into the woodchip media as in biochar-amended woodchip bioreactors, provides additional economic and environmental benefits. A case study reported that the cost of producing P-laden biochar is approximately \$3.05 per kg, which is slightly higher than the cost of conventional triple-superphosphate fertilizer at \$2.88 per kg [62]. The two-stage configuration allows for easier replacement of spent biochar pellets. Spent biochar, once saturated with P, can be reused as P-laden biochar pellets and applied as a soil amendment or slow-release fertilizer in the nearby farms. In a previous study, the application of 2 % spent designer biochar significantly enhanced radish growth, increasing fresh biomass by 15.6–30.8 % compared with the control treatment [30]. Therefore, P-laden spent biochar can serve as a secondary phosphorus resource, providing slow-release nutrient availability to crops while reducing labor and costs associated with biochar disposal pathways further strengthen the environmental and economic sustainability of the B^2 system.

Given its low-cost structure, operational adaptability, and multiple environmental co-benefits, we recommend broader implementation of the B^2 system, particularly in nutrient-sensitive watersheds. To maximize its potential, we propose three key recommendations: (1) continue developing low-cost, effective biochar or other green materials for secondary treatment systems to enhance pollutant removal efficiency; (2) improve the operation of two-stage treatment systems, taking into account the different lifespans and maintenance requirements of woodchip bioreactors and secondary treatment systems; and (3) strategically deploy the B^2 system in nutrient hotspots identified through long-term water quality monitoring to maximize site-specific effectiveness and cost-efficiency.

Despite these encouraging results, several limitations remain. The current study covered a one-year monitoring period, which may not fully capture seasonal variations and long-term performance stability. Future research should therefore focus on multi-year field demonstrations across diverse soil and climate conditions, quantification of greenhouse gas emissions, and optimization of biochar replacement frequency and system hydraulics. Such efforts will help refine the life cycle and techno-economic frameworks for the B^2 system, enabling broader implementation in nutrient-sensitive watersheds and providing stronger scientific support for agricultural water-quality policies.

5. Conclusion

This study demonstrated the development and field-scale performance of an innovative two-stage woodchip bioreactor–biochar (B^2) treatment system for mitigating nitrogen and phosphorus losses from tile-drained agricultural fields. The B^2 system showed effective nitrogen removal performance. The NO_3-N and NH_4-N mean removal efficiency in the B^2 system could reach 58 % and 72 % respectively. Phosphorus removal was primarily achieved through the biochar sorption channels within the B^2 system, which reduced DRP by 3–92 % (median 69 %) and TP by 20–94 % (median 55 %). In addition, biochar sorption channels mitigate the P leaching from the first-stage woodchip bioreactor. A set of TEA based on the field-scale observation illustrates that the B^2 system is a cost-effective solution for nutrient management from agricultural drainage water. The pilot-scale B^2 system achieved unit removal costs of \$90.3 per kilogram of NO_3–N removed per year and \$63.9 per kilogram of DRP removed per year. Under the TEA scenario, the cost could be further reduced to \$4.7 $\pm$ 1.9/kg NO_3–N/yr and \$103.7 $\pm$ 153.5 kg/DRP/yr when the system is scaled up to treat drainage water from a 10-ha agricultural area. Further studies are needed to evaluate the long-term performance and large-scale applicability of the B^2 system in the nutrient-sensitive watersheds to maximize its effectiveness and sustainability.

CRediT authorship contribution statement

Hongxu Zhou: Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Haribansha Timalsina:** Writing – review & editing, Writing – original draft, Visualization, Formal analysis, Data curation. **Richard Cooke:** Writing – review & editing, Writing – original draft, Project administration, Funding acquisition, Conceptualization. **Rabin Bhattarai:** Writing – review & editing, Writing – original draft, Supervision. **Wei Zheng:** Writing – review & editing, Writing – original draft, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jwpe.2025.109258.

Data availability

Data will be made available on request.

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