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Examination of nutrient and sediment loss mitigation for farm roadway runoff on an Irish dairy farm

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ABSTRACT

In Ireland, farm roadway runoff is a potential farm-scale pollution contributor of nutrients and sediments to connecting open drainage channels that pose a challenge to meeting European Union Water Framework Directive goals. To date, recommended mitigation measures such as swales, sediment ponds, bunded drains, willow beds, among others, have not been widely tested for efficiency and therefore are limiting farmers' willingness to implement them. This study quantifies the efficiency of bespoke sediment ponds at three locations (in a treatment train with diversion bars and riparian buffers) as farm roadway runoff mitigation measures by identifying runoff connectivity to open drainage channels, co-developing and co-implementing with farmers and monitoring for efficiency in nutrient and sediment removal. The study results suggest sediment ponds are efficient for removing sediment, total suspended solids and particulate nutrients, but vary in their effectiveness in removing dissolved nutrients due to biogeochemical and hydrological processes. The study concludes that sediment ponds are efficient for reducing roadway runoff pollution to open drainage channels but need to be designed to incorporate segmentation, consider all site conditions and encourage vegetation growth for enhanced nutrient and sediment removal, which may facilitate uptake among farmers. Long-term monitoring would be required to inform maintenance procedures and scheduling.

1. Introduction

Agricultural pollution from nutrient and sediment losses remains a concern for water quality degradation globally (McDowell et al., 2020; Shortle and Horan, 2017). In the European Union (EU), pollution from agriculture contributes to 22 % of surface water and 28 % of groundwater pollution (European Environment Agency (EEA) (EEA), 2021). To alleviate this environmental concern, multiple international, regional and local policies and regulations for managing agricultural pollution have been developed and continue to be adapted for practical implementation. In 2000, the EU developed the Water Framework Directive (WFD) (2000/60/EC; Official Journal of the European Community OJEC, 2000) for member states to adopt an integrated approach for managing waterbodies to reduce pollution and improve water quality to a "good status" by set deadlines. As part of the WFD integrated approach on water quality management, the Nitrates Directive (91/676/EEC) targets reducing agricultural pollution to waterbodies through good agricultural practices (Official Journal of the European Community OJEC, 1991) and requires EU member states to develop a Nitrates Action Programme (NAP) in reaching this goal.

In Ireland, programmes of measures to fulfil the goals of the WFD are set out and revised within the NAP (Department of Housing, Local Government and Heritage DHLGH, 2021a) to minimise both diffuse and point agricultural pollution potential. The NAP measures include, but are not limited to, limits on farm stocking rates and nutrient application rates, prohibitions on organic and chemical fertiliser application at environmentally-sensitive periods, minimum storage capacity for livestock manures and minimum set-back distances from waters (Department of Housing, Local Government and Heritage DHLGH, 2021b).

List of abbreviations: CSAs, Critical Source Areas; DON, Dissolved Organic Nitrogen; DRP, Dissolved Reactive Phosphorus; DUP, Dissolved Unreactive Phosphorus; N, Nitrogen; NAP, Nitrates Action Programme; NH_4-N, Ammonium as Nitrogen; NO_3-N, Nitrate as Nitrogen; P, Phosphorus; PON, Particulate Organic Nitrogen; PP, Particulate Phosphorus; TN, Total Nitrogen; TON, Total Oxidized Nitrogen; TP, Total phosphorus; WSP, Water-soluble P.

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Recent iterations of the Nitrates Directive (S.I. No. 605 of 2017) acknowledge the risk of pollution from farm roadway runoff into connected open drainage channels and stipulate that "there shall be no direct runoff of soiled water from farm roadways to waters from 1 January 2021", alongside mitigation guidance options under the NAP to manage farm-scale agricultural pollution. Recent research on roadway runoff shows nutrient losses occurring both on open and closed periods on grassland farms (Fenton et al., 2024b; Sifundza et al., 2024). It has been found that 8.4 % (Rice et al., 2022) to 11.6 % (Maher et al., 2023) of roadway sections are connected to open drainage channels, while farm roadway and open drainage channel densities are highest on heavy textured soils. During rainfall events, nutrients within and on farm roadway sections connected to open drainage channels form critical source areas (CSAs) (Opoku et al., 2024a, 2024b) and are a sub-component of the nutrient transfer continuum (Fenton et al., 2022). After identification of CSAs, breaking the pathway before delivery of nutrient-rich roadway runoff to open channels is advised on farms (Fenton et al., 2021; Lucci et al., 2010; Opoku et al., 2024b).

Mitigation measures exist in broad terms, but bespoke solutions are needed for specific runoff problems. Primarily, approaches for preventing roadway runoff connectivity focus on breaking the pathway with on-roadway flow diversion structures and retention mitigation systems to reduce the transfer of nutrient and sediment losses to open drainage channels (Fenton et al., 2021; Tanner et al., 2023). The NAP recommends multiple mitigation measures and highlights a "right measure, right place" approach in their use to address diffuse pollutant sources, including farm roadways (Department of Housing, Local Government and Heritage DHLGH, 2024). However, the implementation of these recommended mitigation measures has generally only occurred on European Innovation Partnership (EIP) participant farms with no efficiency testing to guide future iterations and improvements of the mitigation measures. This limits knowledge of the efficiency of these mitigation measures, especially as they have tailored designs.

The efficiency of mitigation measures likely varies depending on the geo-positioning and design of the measure (Tanner et al., 2020; Thomas et al., 2016) and on the CSA characteristics (Tanner et al., 2020) such as farm management (e.g. grazing, stocking rate), rainfall, landscape characteristics (e.g. slope, soil) and contributing roadway area (e.g. size, composition, length and slope). These factors influence the impact on the hydrological and biogeochemical processes that determine the efficiency of mitigation measures (Persson and Wittgren, 2003). Furthermore, their efficiency may be influenced by available farmland sizes, which is often a constraint due to farmers' inability to release farm areas (Lastra-Bravo et al., 2015; Wilcock et al., 2012) for environmental measures. Ryan et al. (2025) observed that farmers are inclined to undertake evidence-based measures and those that require high-level knowledge or understanding for effective implementation. Assessing the efficiency of NAP-recommended mitigation measures in breaking the pathway and slowing farm roadway runoff to reduce agricultural nutrient and sediment transfer to connected open drainage channels will provide a thorough understanding of the context under which these mitigation measures may be effective. Such an understanding of their efficiency will improve knowledge of the mitigation measures in managing farm roadway runoff on Irish farms.

This study selects an Irish dairy farm with a high density of farm roadways and open drainage channels. The study aims to (1) use existing tools to examine and identify farm roadway CSAs where connectivity runoff enters open drainage channels, (2) co-develop and implement bespoke mitigation measures for these identified locations with the farmer, and (3) monitor the efficiency of the implemented mitigation measures at these locations under practical conditions.

2. Materials and methods

2.1. Study site

A dairy farm (45 ha) situated in the south-west of Ireland was selected (Fig. 1) following a previous semi-quantitative risk assessment on open drainage channels where farm roadway runoff connectivity was a prevalent issue and locations with a high risk of roadway runoff were identified (Opoku et al., 2024b). The location has a 10-year average annual rainfall of 1541 mm. The annual agronomic soil testing for phosphorus (P) using Morgan's reagent (Peech and English, 1944) carried out on the fields showed that 10.3 % had high soil P index 4 ($>$ 8.0 mg L^{-1} P). The site has undulating topography with steep slopes (4 – 8 °) and soils classified as "heavy-textured". The soils vary from mineral to humic (Fealy et al., 2009), and are mainly moderately drained (55 %) or poorly drained (45 %), with 13.6 % of the fields having in-field drains installed. The nature of the soils and the topography enable overland flow and potential runoff from CSAs of sediment, nitrogen (N) and P into open drainage channels. The fields in the central parts of the farm have mostly moderately draining soils and therefore have a potential for infiltration (leaching) of nutrients, which complicates the task of isolating pollutant loss pathways on the farm.

2.2. Identifying runoff connectivity and critical source areas

Using the semi-quantitative risk assessment of Opoku et al. (2024b), three locations on the farm (A, B and C) with "high risk" open drainage channels with roadway runoff connectivity as a major contributor were identified (Fig. 1). These locations were cross-checked with the national EPA nutrient loss pathway map (https://gis.epa.ie/EPAMaps/Water) for risky pathways and delivery points to identify roadway runoff CSAs with a high likelihood of nutrient and sediment loss. The identified points of surface runoff delivery to open drainage channels from the national EPA nutrient loss pathway map were noted for further assessment. Following this, a ground survey assessment and visual assessment (Fenton et al., 2024a) was conducted during hydrologically-active periods to fine-tune these farm roadway CSAs and to identify the optimum locations for mitigation measures. At all locations, specific criteria of direct connectivity points where roadway runoff enters open drainage channels, the impact of the connectivity and the associated visual indicators, were used in the ground survey assessment. Direct roadway runoff entry into open drainage channels, with impacts of potential nutrient and sediment delivery, were observed at all locations. Open drainage channels routinely run through small culverts beneath farm roadways. Due to topography, these culverts lay in valley sections of the roadway, and they influenced roadway runoff behaviour. The delivery points were determined by local road configuration at these points. Other visual indicators including the disturbed roadway surface created by farm machinery tracks and livestock movements indicating freshly deposited and legacy P bound sediments, and the formation of runoff rills on the sections of farm roadways indicating overland flow, were also identified at all locations.

2.3. Co-developing and co-implementation of mitigation measures

Several farm visits were undertaken to determine possible mitigation solutions for the three identified farm roadway CSAs in consultation with the farmer. For all three locations, a treatment train mitigation measure of diversion-sediment pond-vegetated riparian buffer was proposed because it combines multiple measures with diverse functions to compensate for the limitations of each measure (Nicholson et al., 2012; Quinn et al., 2007). The diversion bar/cambered roadway diverts runoff to the sediment pond for primary treatment (sedimentation) and subsequently to the riparian buffer for secondary treatment (removal of dissolved pollutants). At Locations A and B, on-roadway concrete-based diversion bars extending 0.3 m beyond the edge of the roadway were

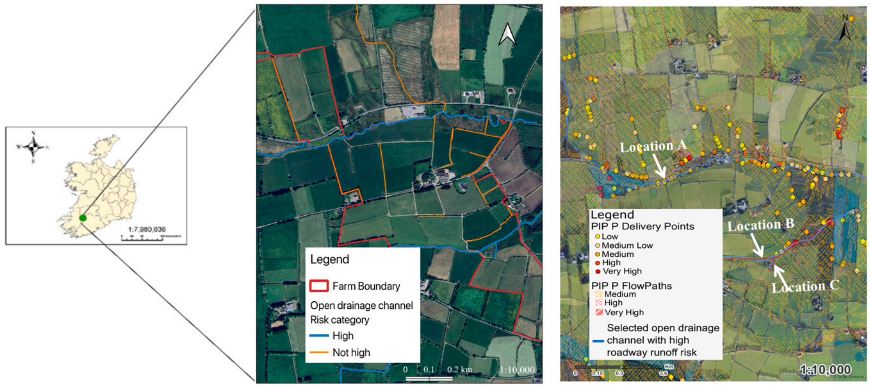

Fig. 1. Map of Ireland indicating farm location, "high risk" open drainage channels with roadway runoff connectivity (Opoku et al. 2024b), and the farm roadway CSA locations on the "high risk" open drainage channels (https://gis.epa.ie/EPAMaps/Water).

installed to direct runoff to the sediment ponds. For Location C, the farm roadway was resurfaced using gravel and cambered to divert roadway runoff towards the sediment ponds. A constant groundwater spring flow from an adjacent field through the cambered section into the sediment pond was observed at this location. The process of co-designing mitigation measures with landowners or advisors typically involves compromises associated with many factors that affect the final mitigation designs. These may for example, limit the size of the installed sediment pond from optimal e.g., this decision could be based on land availability at the delivery point or an unwillingness of the landowner to use that land.

The optimal sediment pond volume was calculated based on the hydraulic loading rate of the site to ensure optimal pollutant removal through sediment retention (Smith and Muirhead, 2023; Robotham et al., 2021). For each location, the sediment pond volume, V (m^3), was calculated using:

$$V = R \times T \tag{1}$$

where R is the peak discharge rate (m^3s^{-1}), and T is the residence time (s). The peak discharge rate, R, in Eq. 1 was calculated using (Barber, 2013):

$$R = C \times A \times I \tag{2}$$

where C is a dimensionless runoff coefficient dependent on hydrological factors (the soil type, land use, degree of imperviousness, slope, surface roughness, antecedent moisture condition, duration and intensity of rainfall, recurrence interval of rainfall, interception and surface storage variables); A is the contributing farm roadway area (m^2), and I is the average intensity of rainfall ($m\ s^{-1}$). A value of 0.5 was assigned to C, which was estimated for forest roadways (Jordán and Martínez-Zavala, 2008) of similar gravel and unpaved characteristics. Using local meteorological records, rainfall intensity, I, for a 6-hour duration, 1-in-5-year return period, storm event was used – 5.67 mm hr^{-1} (1.57×10^{-6} m s^{-1}). Contributing farm roadway areas of 429.3 m^2 over an 8.4 ° slope (Location A), 106.8 m^2 over a 6.7 ° slope (Location B) and 249.5 m^2 over a 7.3 ° slope (Location C) were used.

The residence time, T, in Eq. 1 was calculated using:

$$T = s/Vs \tag{3}$$

where s (m) is the travel distance set for sediments to fully settle in the sediment pond (using s at 1 m) and Vs (ms^{-1}) is the velocity of sediment settling for clay sediment, calculated using Stokes' Law:

$$Vs = \frac{d^2 g(\mathrm{Dp} - \mathrm{Df})}{18\mu} \tag{4}$$

where, d is the diameter of the particle (3.9×10^{-6} m for clay; Barber, 2013), g is gravity (9.8 m s^{-2}), Dp is the density of clay particles (2860 kg m^{-3}; Schjønning et al., 2017), Df is the density of the fluid (1000 kg m^{-3}), and μ is the dynamic viscosity of the fluid (0.001 kg (m $s)^{-1}$).

Based on these hydrological flow estimations, the volumes (V) at depth (s) = 1 m required for the sediment ponds were calculated as 28.6 m^3, 7.1 m^3 and 16.6 m^3 for Locations A, B and C, respectively. While these estimated sediment pond sizes may allow optimum effectiveness, site constraints including high water table of the adjacent open drainage channels and limited land area, especially at Location A, necessitated resizing of the sediment pond sizes to ~ 4 m^3, ~ 7 m^3 and ~17 m^3 at sediment settling depths (s) of 0.5 m, 0.5 m and 1 m for Locations A, B and C respectively (Fig. 2). These constraints led to an undersized sediment pond volume at Location A, while Locations B and C remained with their optimum sediment pond volumes. Following Barber (2013), the sediment ponds were configured into pond cells to enhance removal efficiency while adapting to the site conditions. For the sediment pond configurations, two sediment pond cells, each measuring 2.5 × 1.5 × 0.5 m (L x W x D) at Location A, one sediment pond cell measuring 4.0 × 3.5 × 0.5 m at Location B and two sediment pond cells, each measuring 4.25 × 2.0 × 1.0 m at Location C, were excavated (Fig. 2). The sediment ponds were manually levelled after digger excavation and crosschecked with a spirit level. This allowed accurate measurement of the accumulated sediment volume.

To prevent pond bank erosion, sediment ponds were excavated to create banked sides for stability (Barber, 2013). The sediment ponds were lined with heavy-duty woven weed mats up to the banks, which enabled estimation of accumulated sediment volume. The heavy-duty weed mats used had 0.63 porosity, calculated as 1 - (bulk density of the woven weed mat material/bulk density of a polypropylene solid) (Kalazić et al., 2023), using manufacturer's product specifications of mass per area of 100 g/m^2 and thickness of 0.3 mm which yielded a bulk density of 333 kg/m^3. The bulk density of polypropylene solid was taken as 900 kg/m^3 (Jones et al., 2025). The weed mat was double-lined to reduce porosity. It was assumed that the base weed mat layer in contact with the wet, fine (clayey) soil particles of these heavy textured soils under the pond water pressure clog over time, further reducing porosity and infiltration potential. The weed mats were firmly attached to the sides and base of the ponds with wires and ground cover pegs. Edges (excluding exit and entry) along the sediment pond cell(s) were bunded and grassed to prevent overland flow from adjacent areas during rainfall events. At the exit of every sediment pond, a 1 m-long, 0.10 m-diameter plastic pipe was connected to the next sediment pond or discharged into the 3 m-wide riparian buffer.

The riparian buffer was installed at the end of sediment pond configuration at each location before the adjacent open drainage channel to meet the current recommendation of at least a 3 m-wide vegetated riparian buffer to prevent direct soiled runoff into waterbodies under the 2022 NAP 5 in the EPA Research Report No. 485 (Ó

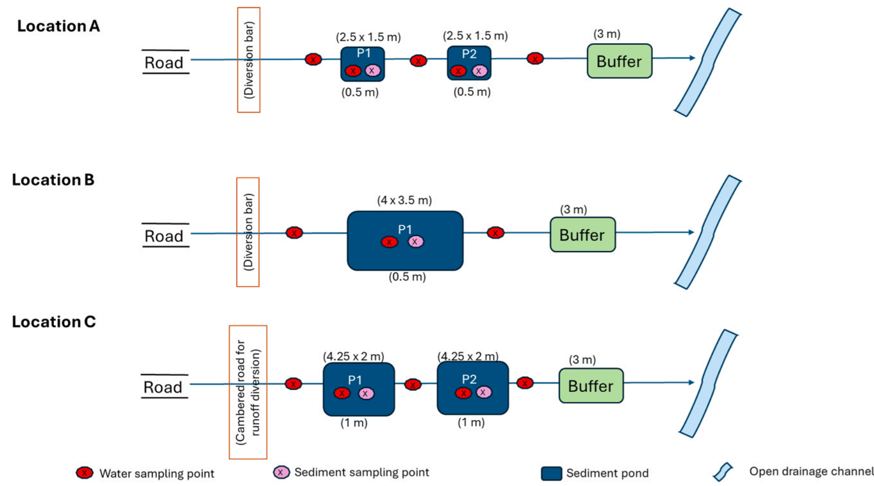

Fig. 2. Schematic representation of mitigation measures and dimensions.

Huallacháin et al., 2023). While such an additional measure is expected to further reduce the sediment and pollutant concentration in the runoff from the sediment ponds, the nature of vegetated riparian buffers does not allow direct measurement of downstream water quality at all locations, and therefore measurement at these locations was not undertaken. This study, therefore, only evaluates the efficiency of the sediment ponds on the farm.

2.4. Water and sediment sampling for testing mitigation measures

2.4.1. Water and sediment sampling

Water and sediment samples were taken during the hydrologically-active periods between the week of 22nd October 2024–19th March 2025, except for 3 weeks from late December 2024 to early January 2025 when the site was not accessible due to heavy snowfall. Using sampling points in Fig. 2, two 50 ml paired (filtered and unfiltered) water samples were taken weekly from all water sampling points in all locations for N and P fractions analysis. In addition, 500 ml water samples were taken weekly at these water sampling points in all locations for total suspended solids (TSS) concentration measurement. Inlet water samples were collected from diverted roadway runoff flows at the entry points for each location. All the 50 ml (filtered and unfiltered) and the 500 ml water samples were stored and transported in cool boxes to the laboratory for water analysis and TSS within 24 h of sample collection.

To measure accumulated sediment volume in each pond, 1 m graduated staffs were used to measure the depth of accumulated sediment over the manually levelled pond area over the study duration (Cooper et al., 2019). In each pond, two 1 m graduated staffs were firmly fixed to the pond base along a transect, one near the inlet and the other near the outlet, to measure accumulated sediment depth. The average of the two measured depths was multiplied with the pond's cross-sectional area to estimate accumulated sediment volume. Reference markings on the pond banks were made during installation to assess and correct any positional shifts. The average depth readings of accumulated sediment from both graduated staffs within each pond were calculated every 4 weeks. For a particular pond cell, the calculated average depth and pond area were multiplied to estimate the accumulated volume for that 4-week period. After each 4-week measurement of accumulated sediment, ~0.5 kg of fresh (wet) sediment samples were collected from the base of each pond cell. The sediment samples were transported in cool ice boxes to the laboratory and then analysed for water-soluble P (WSP) to ascertain the sediment P composition.

2.5. Laboratory analysis

The unfiltered 50 ml grab water samples were analysed calorimetrically for nitrite (NO_2-N), ammonium (NH_4-N), total oxidized nitrogen (TON), and total reactive phosphorus (TRP) using a Thermo Fisher Scientific Gallery ™ Discrete Analyzer. The unfiltered samples were analysed for total phosphorus (TP) and total nitrogen (TN) was analysed using the Hach Ganimede P analyser and the Hach Ganimede N analyser, respectively. The filtered 50 ml grab water samples were analysed for dissolved reactive phosphorus (DRP) and total dissolved phosphorus (TDP) using a Thermo Fisher Scientific Gallery ™ Discrete Analyzer and a Hach Ganimede P analyser, respectively. All water samples, reagent blanks and check standards were analysed following the Standard Methods (American Public Health Association (APHA) (APHA), 2005). All quality control (QC) samples/check standards were prepared from certified stock standards from a different source than calibration standards. Quality control samples were analysed at the beginning and end of every sample batch, for every 10 samples within a batch, and if the QC fell outside limits, samples were repeated to the last correct QC. Blanks were included in every sample batch for analysis, and approximately 10 % of samples were repeated. Tolerances ranged up to a maximum of ±7.5 % of the nominal value. All instruments used were calibrated in line with the manufacturers' recommendations. Nitrate-N was calculated by subtracting NO_2-N from TON, particulate phosphorus (PP) was calculated by the difference between TP and TDP, and dissolved unreactive phosphorus (DUP) was calculated by the difference between TDP and DRP. Total suspended sediment concentrations were measured using the standard gravimetric method (American Public Health Association (APHA) (APHA), 2005).

For WSP analysis, portions of the sediment samples for each pond cell were prepared by air-drying and sieving through 2 mm, and 1 g of the prepared sediments were moistened with 2 ml of deionised water and allowed to stand for 22 h. These were further moistened with 70 ml of deionised water, equilibrated for 1 h on a reciprocating shaker (van der Paauw, 1971) and filtered using Whatman No. 4 filter paper before the filtrate was quantified calorimetrically for P. Using the sediment mass (g) and total volume of deionised water (ml; converted to L) used for moistening the sediment, P concentration (mg L^{-1}) in the filtrate was converted to mg/g.

2.6. Data analysis

Microsoft Excel software version 16.0 (2016) was used for data computing and preparation prior to statistical analysis, and R Studio version 4.3.2 (2023) was used for statistical procedures. To assess the efficiency of the sediment ponds deployed at the various locations, the water sampling results for the N and P fractions, TSS, and physical and chemical sediment characteristics were compared. The removal efficiency was defined as the percentage removal calculated as the difference in water quality parameter concentrations at the inlet sampling point of the sediment pond and the outlet sampling point of the sediment pond (Eq. 5):

$$\text{Removal efficiency }(\%) = \frac{\text{Inlet water concentration} - \text{Outlet water concentration}}{\text{Inlet water concentration}} \text{ x } 100\,\% \tag{5}$$

All inlet and outlet water quality data were assessed for normality with the Shapiro-Wilk test and were not transformed. To test efficiency of sediment ponds statistically at individual locations, the inlet and outlet water quality data for each location were tested for statistically significant differences using the paired T-test for normally distributed water quality parameters (Barber, 2013; Robotham et al., 2021) and the Wilcoxon Signed-Rank (pairwise test) for non-normally distributed water quality parameters. All significant differences were observed at an alpha level of 0.05 (95 %) confidence level, and where alpha level was much lower, a 0.01 (99 %) confidence level was used. All water quality parameter values "<LOD" (below the Limit of Detection) or "not detectable" were treated as zero for analysis. Mean comparisons were undertaken for WSP, accumulated sediment volume and weather data (rainfall (precipitation) and temperature). Rainfall refers to the total precipitation, and as these heavy textured, poorly drained soils remained wet throughout the study period, precipitation/rainfall may be considered as very crucial for runoff. Concentrations of P and N fractions of the inlet and outlet of pond configuration systems for a location were examined as proportions of total P and N.

3. Results and discussion

3.1. Assessment for critical source area identification and co-designing of sediment ponds

Visual or ground assessment was guided by surface flow pathways and delivery points available via the EPA nutrient loss pathway maps, which provide details such as actual direction of runoff flow to open drain channels. Although the EPA nutrient loss pathway map use hydrological factors such as soil drainage and topological properties to show delivery points for where roadway runoff into open drainage channels may occur, the ground assessment that indicated that anthropogenic influence including design and characteristics of installed culvert also influenced roadway runoff behaviour. These influences determined flow direction towards to open drainage channels and contributed to temporary localised ponding of roadway runoff prior to discharge across all surveyed locations.

Aside from animal waste (urine and faeces) deposition being key pollutant sources in farm roadway runoff (Fenton et al., 2024b; Sifundza et al., 2024), the eroded rills on farm roadways and the localised ponding observed informed potential risk of nutrient-rich sediment from disturbed roadway surfaces noted as CSAs. This agrees with Maher et al. (2023), who reported that farm roadway characteristics influence farm animal movement, and this makes some sections of the farm roadway more vulnerable to surface disturbance and high waste depositions (Sifundza et al., 2024). The movement of farm machinery and animals is expected to disturb the roadway surfaces back into the localised ponding, potentially increasing pollutant discharge into open drainage channels at certain times. The identification of temporary localised ponding during the ground assessment influenced the decision to use diversion bars and in some cases road cambering (as in Location C) to divert flow and limit the temporal occurrence of roadway ponding.

Furthermore, the farmer co-development aided identification of unused land spaces on the farm for implementation of the sediment ponds. Utilising unused spaces has also been reported as a potential option for successful implementation of environmental management measures in Batáry et al. (2015) and Burland and von Cossel (2023). This increases farmers' commitment as they do not lose their productive farmlands, thus having minimal to no practical impacts. Lastly, the involvement of farmers in co-designing promoted cooperation which ensured on-farm safety as farm animals were prohibited from accessing sediment ponds and ensuring smooth operation of the installed sediment ponds.

3.2. Sediment trapping in sediment pond configurations

During the study, accumulated sediment volumes (m^3) within pond cells increased by 0.169 m^3 in pond cell 1 and 0.128 m^3 in pond cell 2 at Location A, 0.088 m^3 in pond cell 1 at Location B, and 0.077 m^3 in pond cell 1 and 0.038 m^3 in pond cell 2 at Location C. This indicates a 53–567 % sediment accumulation (relative to the initial sampling volumes) across locations during the monitoring period (Table 1). These findings show that sediment accumulation in ponds is related to contributing area, with larger areas yielding more accumulated sediment in ponds. Although precautions were taken to limit porosity of the weed mat and ensure accurate accumulated sediment volume estimations (Section 2.3) with the method adopted, very fine sediment (and associated P concentration) may be expected to travel through weed mat, leading biased underestimation in all sediment ponds. However, this methodology as opposed to previous sediment measurements in Barber and Robotham where circular plates (saucers) were placed at the base of pond to trap sediment was best suited for these sediment ponds (of full capacity for nearly all the sampling period) as drawing out these plates may lead to sediments washing off and causing random error as opposed to systematic error.

The sedimentation process is influenced by factors including pond size and flow reduction capacity, runoff flow velocity, sediment size characteristics. Sediment size influences sedimentation, allowing coarse sediment to settle more quickly and fine particles to remain suspended until flow slows (Clarke, 2013; Levine, 2020; Ockenden et al., 2012). Higher mean sediment accumulation in the first pond cells at Locations A and B (Table 1) suggests coarse sediment trapping, which occupies more volume. Conversely, lower mean sediment accumulation in their respective second pond cells (Table 1) suggests fine sediment trapping which, due to their smaller size and lower weight, travel far and occupy less volume. Visual observations, especially during and immediately after rainfall, revealed cloudier water in second pond cells, suggesting resuspension of lightweight fine sediments. This aligns with findings of multi-pond studies, which also observed that first pond cells trapped heavier and less mobile sediments than subsequent cells (Barber, 2013; Robotham et al., 2021). Fine sediments are major P carriers that contribute to P losses (Ballantine et al., 2006; Shore et al., 2015). This may have contributed to the higher WSP concentrations in sediments of the second pond cells relative to the first pond cells at Locations A and C (Fig. 3).

Rainfall, the primary driver for sediment mobilisation from farm roadways (Fenton et al., 2021; Rice et al., 2022), had positive

Table 1
Accumulated sediment volumes (m^3) and percentage increase from start to the end of monitoring.

Location (contributing area (m^2))	Pond	Volume of sediment accumulated at initial measurement (m^3)	Volume of sediment accumulated at final measurement (m^3)	Mean volume of sediment accumulated at each measurement (m^3)	Total sediment volume increase relative to initial volume (%)
A (429.3 m^2)	Cell 1	0.038	0.206	0.13	450.0
	Cell 2	0.023	0.150	0.10	566.7
B (106.8 m^2)	Cell 1	0.049	0.137	0.09	178.6
C (249.5 m^2)	Cell 1	0.068	0.145	0.11	112.5
	Cell 2	0.072	0.111	0.09	52.9

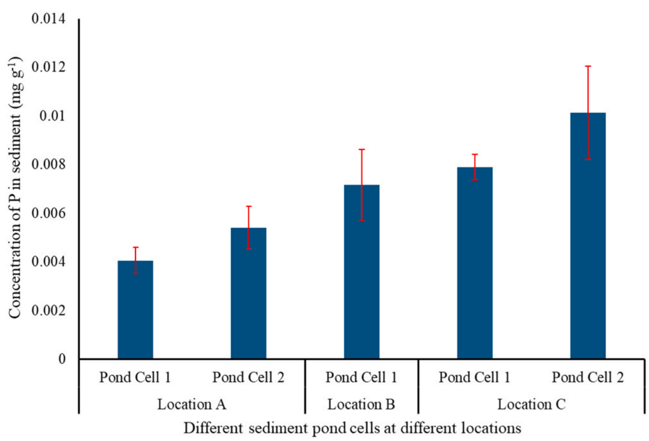

Fig. 3. Water-soluble P (WSP) of sediment samples at different locations and pond cells.

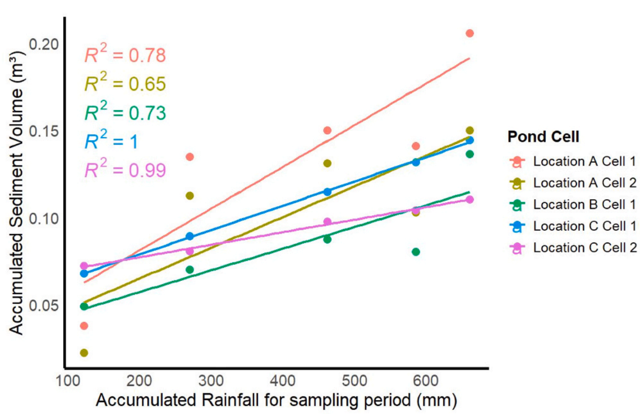

Fig. 4. Correlation between accumulated sediment volume (m^3) and accumulated rainfall (mm) for every sampling period.

correlations with sediment accumulation ($R^2 = 0.65 - 1$) (Fig. 4), suggesting that transport of accumulated sediments in ponds was dependent on rainfall. This correlation was even more pronounced in the first pond cells.

3.3. Nutrient and TSS removal in sediment ponds

3.3.1. Nitrogen removal efficiency in sediment ponds configurations

Over the sampling period, TN removal efficiencies in Location A and C (both two-cell configurations) were similar, at 30.9 ± 39.0 % and 27.4 ± 42.6 % removal respectively, while the one-cell pond configuration system at Location B recorded only 0.46 ± 13.8 % removal

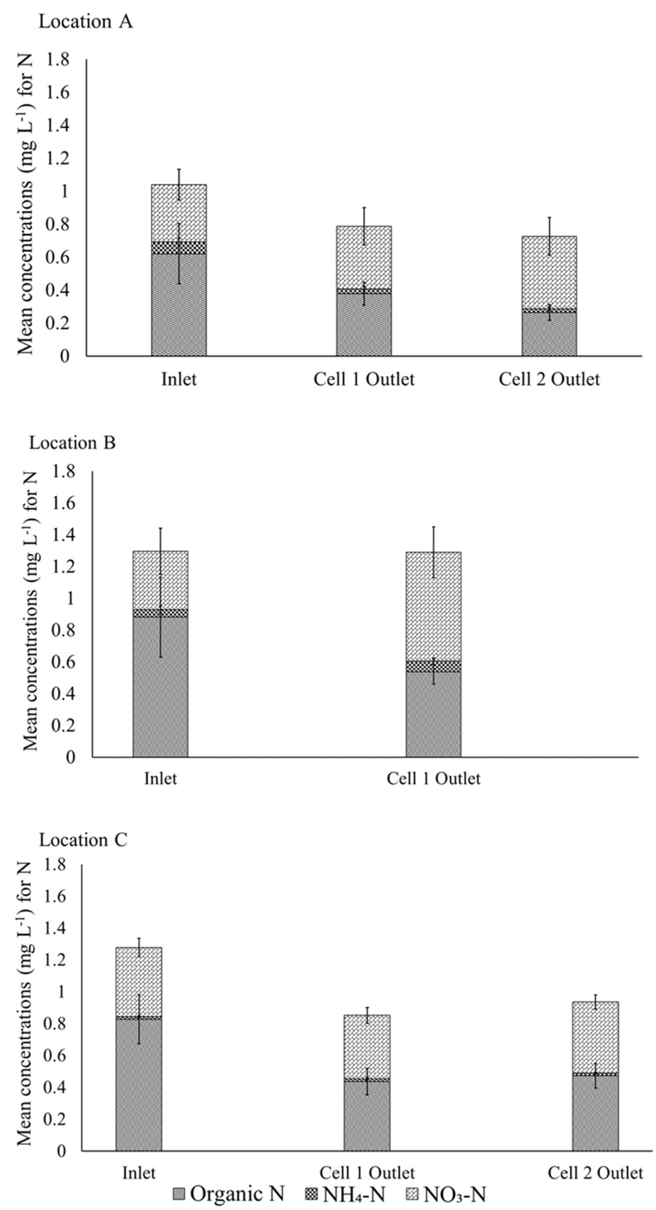

Fig. 5. Nitrogen (N) mean ± standard error concentrations from sediment pond cells at Locations A, B and C on study farm.

(Fig. 5). Nevertheless, the mean outlet TN concentrations for the sediment ponds at all locations (Table S1) were lower than the current N discharge limit of 10 mg L^{-1} under EU Urban Waste Water Treatment

Directive (UWWTD 91/271) (European Commission, 2024). The TN removal efficiencies in Locations A and C are consistent with the average TN removal efficiency of 31 % for wet ponds in Koch et al., 2014, but relatively lower than Mallin et al. (2012) reported results of 66 – 96 % TN reduction in a 4.7 ha multi-segmented constructed wetland designed for a 24-hour duration, 1-in-100-year return period storm event. Within runoff treatment systems, sedimentation and microbial transformations (mineralisation, nitrification and denitrification) and plant uptake are the primary N removal mechanisms (Kill et al., 2018; Vymazal et al., 1998), and these factors considerably influence variation in N removal efficiencies. The relatively low TN removal in this study could be due to low temperatures measured during the study period (6.73 ± 0.26 ℃) which reduce the microbial transformations (Kill et al., 2018; Robotham et al., 2021), regular wet season runoff which limited hydraulic retention (Braskerud, 2002), and lack of vegetation in the lined study pond cells. The study site, characterised by a wet, heavy textured soils and a high (10-year average) annual rainfall of 1541 mm, is prone to frequent runoff due to prolonged soil saturation during the wet season and this limits TN removal efficiency. Kim et al. (2011) showed relatively lower N removal during the wet season due to influence from seasonal factors like temperature (Persson and Wittgren, 2003). Based on this assumption, the low temperatures in the wet season likely inhibited microbial activities, limiting N removal efficiency in this study. The wet season is a critical period for roadway runoff mobilisation under the Irish temperate conditions, and with the observed low N removal efficiency in this study, it is imperative to enhance sediment pond performance. Vegetation in sediment ponds enhanced TN removal efficiencies in previous studies (Beutel et al., 2009; Wang and Sample, 2014). While the sediment ponds assessed in this study had no vegetation growth as they were firmly lined with heavy-duty weed mats (which blocks vegetation growth) to allow for accurate measurement of accumulated sediment volumes, the introduction and assessment of sediment ponds with vegetation, under the cold and wet temperate Irish will inform options for optimal N removal.

Organic N concentrations decreased at all three locations, albeit only significantly at Locations A and C (Table S1). These positive organic N removal efficiencies agree with Mallin et al. (2012), who reported an average 70 % organic N removal efficiency through a treatment system. In segmented pond systems, the first pond cell slows flow velocity and retains particulate nutrient forms. In contrast, flow in the one-cell pond configuration at Location B lacks segmentation, potentially leading to short-circuiting (with potential direct flow out of ponds) and limiting organic N removal via sedimentation.

Organic N exists in dissolved (DON) and particulate (PON) forms, and removal mechanisms may vary depending on its forms. Removal mechanisms include sedimentation for PON and microbial mineralisation for DON, depending on the labile or refractory composition of organic N for microbial breakdown (Bronk et al., 2007; Mallin et al., 2012). Ponds are generally static systems, where nearly all nutrient transformations occur through exchange processes (Boyd, 1995). Higher NO_3-N concentrations recorded at exit pond cell outlets (Fig. 5) suggest that mineralisation of organic N to NH_4-N, followed by rapid nitrification to NO_3-N, may have occurred within the ponds. The statistically significant positive organic N removal efficiencies (Table S1) in the two-cell pond configuration may stem from enhanced PON sedimentation due to pond segmentation at Locations A and C.

Inorganic nitrogen removal efficiencies varied considerably across locations. At Locations B and C, NH_4-N concentrations increased, whereas it reduced in Location A (Table S1). The average influent NH_4-N concentrations were, however, very low, ranging from 0.02 to 0.072 mg L^{-1}. Similar NH_4-N removal inefficiencies of −61 ± 118 % were reported by Robotham et al. (2021) in a three small online-pond study. The positive mean NH_4-N removal efficiency in Location A (68.11 ± 66.00 %) may be due to shorter hydraulic residence time in these undersized pond cells which may have limited the ammonification of retained organic N in the first pond cell, leaving lower NH_4-N concentrations to travel to the second pond cell and then the outlet. On this assumption, where NH_4-N removal inefficiencies recorded in the optimal size ponds maybe due to ammonification of retained organic N.

Through nitrification, NH_4-N concentrations convert into NO_3-N concentrations (Vymazal et al., 1998), adding to the initial NO_3-N concentrations and increasing NO_3-N leaving the ponds. Although not statistically significant, all three locations had negative mean NO_3-N removal efficiencies (Table S1). Studies by Kim et al. (2011), Mallin et al. (2012) and Robotham et al. (2021) report contrasting results of positive mean reductions. Their results may have varied from this study primarily due to the low mean air temperature over the monitoring period of 6.7 ± 0.3 ℃ in which this study was conducted. The temperature may have inhibited microbial transformations (e.g. denitrification) for NO_3-N removal, compared to the reported mean removal efficiencies for all seasons. Incorporating vegetation within these pond cells to function as constructed wetlands (Tang et al., 2021) would improve NO_3-N removal via plant uptake and provide carbon for

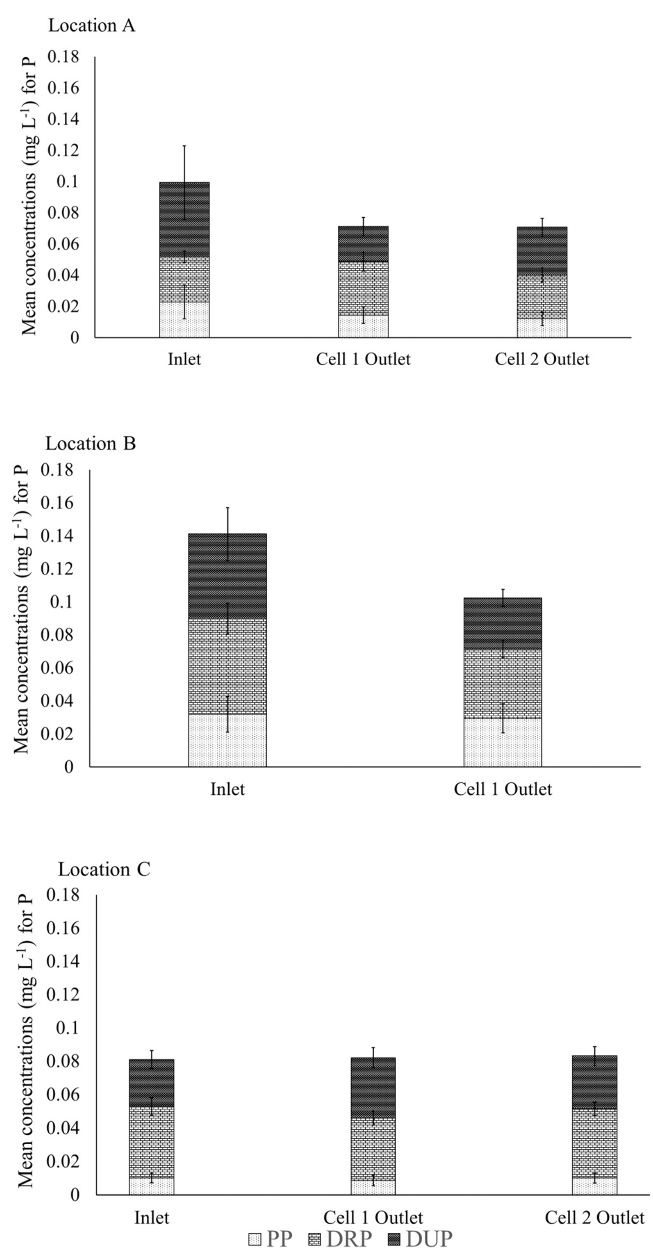

Fig. 6. Phosphorus (P) mean ± standard error concentrations from sediment pond cells at Locations A, B and C.

denitrification under anaerobic conditions.

3.3.2. Phosphorus and TSS removal in sediment ponds

The average influent TP concentration to the sediment ponds ranged from 0.08 mg L^{-1} (Location A) to 0.75 mg L^{-1} (Location C). Locations A and B had TP removal efficiencies of 17.0 ± 38.1 % and 11.7 ± 7.1 %, respectively (Fig. 6), whereas Location C had a TP removal efficiency of −10.4 ± 9.2 % (Table S1). Removal efficiencies of total phosphorus in previous studies show considerable variations, ranging from −34 % and 89 % in Robotham et al. (2021) and Mallin et al. (2012), respectively. This is primarily due to varying physical and biochemical processes. However, Yazdi et al. (2021) reported a 10 % TP removal efficiency during colder weather in a year-long sediment pond study which is comparable to the observed results at Locations A and B in this study which was also conducted under cold and wet seasons. The TP removal inefficiency at Location C aligns with Robotham et al. (2021) who reported TP removal inefficiency in a multi-pond study due to hydrological fluxes. In this study, the TP removal inefficiency at Location C is similarly attributed to hydrological flux from the continuous spring flow entering the sediment ponds at Location C. Notwithstanding the negative TP removal at Location C, mean outlet TP for all locations remained lower than the current P discharge limit of 0.7 mg L^{-1} under the EU Urban Waste Water Treatment Directive (UWWTD 91/271) (European Commission, 2024).

All locations had positive mean PP removal efficiencies: 47.0 ± 60.3 % at Location A, 7.1 ± 17.4 % at Location B and 1.1 ± 4.4 % at Location C (Table S1). These reductions indicate effective PP removal by sedimentation, consistent with the observations of Shan et al. (2002). There was a similar trend for TSS (Fig. 7), with mean removal efficiencies of 63.0 ± 79.2 %, 81.5 ± 90.9 % and 57.9 ± 84.7 % at Locations A, B and C, respectively. This demonstrates suspended sediments' contributions to P concentrations (Cooper et al., 2015; Evans et al., 2004), and highlights sediment pond systems' role in trapping particulate pollutants (Gu et al., 2017; Mekonnen et al., 2017). Total dissolved P, comprising DUP and DRP, dominated P in the inlet, ranging from 77.5 % (Location B) to 94.1 % (Location C) of TP (Fig. 6). Locations A, B and C had positive mean DRP removal efficiencies of 3.9 ± 19.2 %, 27.9 ± 44.6 %, and 3.0 ± 21.9 %, respectively. The DRP reductions are consistent with the 14.9 % and 29 ± 37 % mean removal efficiencies in the pond treatment studies of Barber (2013) and Robotham et al. (2021), respectively. Adsorption is the principal removal mechanism for dissolved P (Lai and Che, 2008), and this is influenced by the availability of the adsorbing sites. The WSP, which indicates readily available P, of pond sediments at Location C was relatively higher than at Locations A and B (Fig. 3), indicating P-concentrated sediment. Concentrated P sediments have limited adsorption sites, and this may have potentially lowered adsorption, leading to low P removal at location C. Further research on equilibrium P concentration and sorption analysis on the sediment, however, may be required improve understanding on the adsorption.

The reduction trend of DRP was also observed for DUP at Locations A and B, but increased significantly at Location C ($p < 0.05$). The continuous hydraulic loading and base flow, driven by the connecting groundwater spring emerging through the cambered section of the reconstructed farm roadway into Location C's pond cells, may have impacted the P removal. Kill et al. (2018) attributed low nutrient removal in a runoff treatment system to constant groundwater flow seeping from adjacent fields. Such conditions create consistent flow currents that reduce residence time (Brown et al., 1981), cause sediment resuspension (Saeed et al., 2019) to release P into the water column (Sinke et al., 1990; Søndergaard et al., 2003) as organic P (DUP), and increase aeration for microbial desorption (Stahlberg et al., 2006; Yu et al., 2022). This finding reinforces the importance of matching pond design to actual local hydrological context and provides a novel idea, by including other characteristics associated with flow such as permanence and seasonal dynamics, where present, into the pond volume estimations. Through this research, farmers are aware of external factors that can influence sediment pond performance and are encouraged to provide relevant local hydrological knowledge to engineers or scientists to ensure optimal sediment pond designs for effective nutrient and sediment removal. This approach, where farmers provide vital local knowledge in co-designing, is expected to ensure successful implementation of sediment ponds on Irish farms. Previous studies of Amblard et al. (2023), Campling et al. (2021) and Richard et al. (2020) support this as an effective tool for integrating local farm-specific insights to improve design constraints and ensure successful implementation of agri-environmental measures for improving water quality.

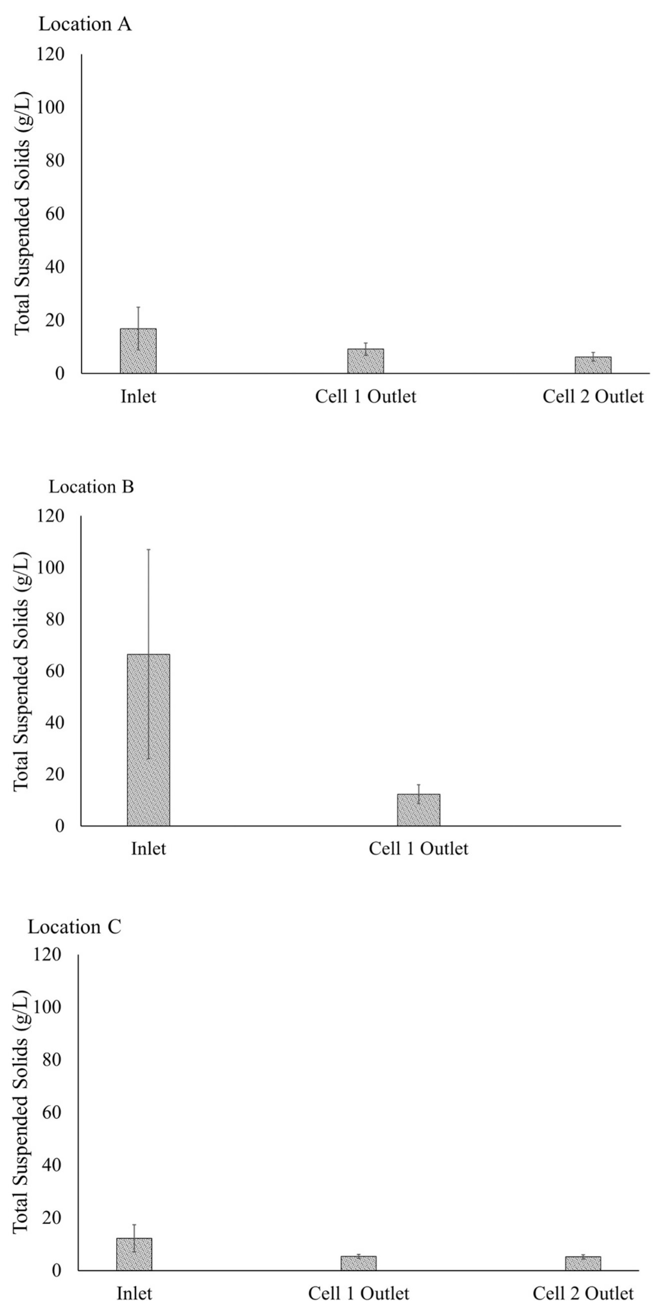

Fig. 7. Mean total suspended solids (TSS) ± standard error concentrations from sediment pond cells at Locations A, B and C.

4. Conclusion

This study showed that sediment ponds, implemented at appropriate locations for managing farm roadway runoff loss to open drainage channels, are effective in removing sediment, TSS and particulate nutrients, but vary in the removal of dissolved nutrients. The trapping of sediment in ponds is dependent on the contributing area as a sediment source and rainfall as a mobiliser, while nutrient removal is dependent on the pond design and site conditions. Policy recommendations delivered through farm advisory services to farmers on future iterations of sediment ponds design should promote pond segmentation, accounting for site-specific hydrological conditions such as constant hydrological loadings from groundwater springs (if present), and inclusion of vegetation to improve their hydrological and biogeochemical functioning for enhanced nutrient removal. With the provision of this high and practical knowledge on sediment pond effectiveness (including those with constrained pond design sizes), farmers will be more likely to use sediment ponds for managing farm roadway runoff entering open drainage channels. Long-term monitoring of at least one year to capture all seasonal runoff variations and further research on equilibrium P concentration of sediment are required to make estimations for maintenance measures such as pond dredging.

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CRediT authorship contribution statement

D.G. Opoku: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Validation; Visualization; Roles/Writing – original draft; Writing – review & editing. **M.G. Healy:** Conceptualization; Funding acquisition; Investigation; Methodology; Supervision; Validation; Visualization; Writing – review & editing. **O. Fenton:** Conceptualization; Funding acquisition; Investigation; Methodology; Supervision; Validation; Visualization; Writing – review & editing. **P. Tuohy:** Conceptualization; Funding acquisition; Investigation; Methodology; Project administration; Resources; Supervision; Software; Validation; Visualization; Writing – review & editing.

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. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agwat.2025.110007.

Data availability

Data will be made available on request.

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