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Management of critical source areas (CSAs) in pasture grazed by deer to reduce contaminant losses to water

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ABSTRACT

Agriculture is important for producing food but can have environmental impacts on water quality. There is a need to develop mitigation strategies for pastoral farming systems to find a better balance between food production and environmental footprint. This study used a paired catchment experimental design to test the effectiveness of three mitigation strategies targeting the critical source areas (CSAs) that were ephemeral streams in deer grazed pastures. The runoff from four small catchments on the same hill slope (2.2–3.6 ha) was monitored for 2 years, under typical farm management. The mitigation options of: full fencing, partial fencing and temporary fencing was then applied to the CSA areas in three of the catchments. The fourth catchment was left unchanged as a control to account for different weather conditions in the pre- and post-treatment phases. Post-treatment monitoring was conducted for a further 2 years. The effectiveness of the mitigation options was calculated for both reducing contaminant concentrations (nitrogen, phosphorus, sediment and *E. coli*) under low-flow conditions and contaminant loads during storm events. The effectiveness of the three mitigation options for reducing low-flow concentrations ranged from not effective for filterable reactive phosphorus (FRP), total phosphorus (TP), and total suspended solids (TSS) to 83 % effective for reducing *E. coli*. The effectiveness of the three mitigation options for reducing storm-flow event loads ranged from not effective for FRP to 93 % effective for reducing *E. coli*. The CSA managements all mitigate multiple contaminants and hence will have multiple water quality benefits downstream.

1. Introduction

The impact of livestock production systems on the environment is a concern worldwide (Leip et al., 2015). The largest area of productive land use in New Zealand (NZ) is pastoral agriculture which is known to have a significant impact on water quality (Larned et al., 2020). Agricultural headwaters have been identified as critical ecosystems as they represent the interface between terrestrial and aquatic ecosystems and yet they are poorly studied (Wohl, 2017; Bieroza et al., 2024). The four main water quality contaminants in diffuse losses from agriculture are nitrogen (N), phosphorus (P), sediment (Sed) and faecal-indicator bacteria (*E. coli*). Mitigation options have been developed and applied on farms in NZ over the past few decades (Monaghan et al., 2021). However, it was estimated that actual reductions of N losses to water have not occurred as increased farming intensity and land use change has more than offset any mitigation effects, resulting in a national scale 25 % increase in N losses (Monaghan et al., 2021). Losses of P and Sed have seen improvements, with national scale losses estimated to have been reduced by 23 and 29 %, respectively (Monaghan et al., 2021). Monaghan et al. (2021) were unable to provide any estimate of the effectiveness of *E. coli* mitigations due to a lack of suitable data to model losses from farms, indicating a need for more research on faecal contaminant losses from land to water. Estimates of the future potential reductions through increased implementation of established mitigations could reduce N and P losses by 16 and 23 %, respectively (McDowell et al., 2021). These potential reductions could be further increased by up to 34, 39 and 66 % for N, P and Sed, respectively, if all mitigation options currently under development were adopted on all farms – an unlikely outcome (McDowell et al., 2021). This illustrates the need for development and evaluation of new mitigation options for pastoral farming.

The critical source area (CSA) concept was developed in the 1990s and has been applied and tested around the world (McDowell et al., 2024). CSAs are small areas of a field or farm that generate most of the

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contaminant losses in surface pathways, due to a combination of high contaminant availability and high transport potential. Thus, mitigations focusing on CSAs often start by identifying the wettest parts of the landscape and then aim to keep contaminant sources and/or processes away from these wet areas. This problem is exacerbated in grazed pastures as animal hoof pressures on wet soils can decrease soil infiltration which further increases runoff volumes (Curran-Cournane et al., 2011; Donovan and Monaghan, 2021). Two recent studies have investigated the mitigation potential of CSA management in winter grazed forage crops (Monaghan et al., 2017; Ghimire et al., 2024). Forage crops grazed in winter represent a worst-case scenario of structurally disturbed soils from planting the forage crops and very high instantaneous stocking densities (Donovan and Monaghan, 2021). However, in these studies (Monaghan et al., 2017) observed contaminant reductions of 66, 67, 80 and 50 % for N, P, Sed and *E. coli*, respectively when applying strategic grazing by dairy cows to the CSA in these paddocks. Ghimire et al. (2024) investigated surface runoff from winter forage crops grazed by sheep and estimated reductions of 38, 48, 55 and 63 % for N, P, Sed and *E. coli,* respectively, when applying a strategic grazing strategy to the CSA area. These 2 studies demonstrate very high contaminant reduction potential of CSA management in forage crops which leads to the question of the mitigation potential for CSA management in pastures that are grazed year-round. McDowell et al. (2017) estimated that 77 % of contaminants lost to water come from first order streams that will be directly connected to most of the wet areas in a landscape that will be potential CSAs. Thus, CSAs in grazed pastures would appear to be a good target for mitigation of contaminant losses to water.

It is well recognised that the annual loads of contaminants in rivers are transported predominantly during the storm events (Defew et al., 2013; McDowell et al., 2025). This is particularly true for the contaminant *E. coli* where it was observed, in a small pastoral agricultural catchment, that 95 % of the annual load was transported during storms (Davies-Colley et al., 2008). In contrast, water quality monitoring protocols involve collecting samples on pre-determined days each year which results in most samples being collected during low-flow conditions (Keenan et al., 2024; McDowell et al., 2025). As a result, we do not have a comprehensive understanding of how the high concentrations and loads of contaminants in transient storm events impact on the overall state of water quality downstream in the predominantly low-flow stream conditions. This interaction between high runoff loads during rainfall events can further complicate our understanding of the impacts of on-farm mitigations on desired water quality outcomes. For example, placing fencing on the edge of a stream to prevent animal access to the water will reduce direct deposition into the stream (O'Callaghan et al., 2019; Muirhead, 2019) and would have a physical effect on any day of the year that animals are in that part of the catchment. Therefore, we would assume that stream fencing would reduce base-flow concentrations of contaminants (Muirhead et al., 2011). Moving the stream fence back further to create a riparian buffer should provide some additional mitigation effect but this additional effect will only occur when there is storm runoff from the land flowing through the riparian buffer (Bai et al., 2016). This raises the question of how effective a riparian buffer will be on the water quality data when the buffer is only "active" for a small number of days each year. A CSA is effectively the head of a riparian buffer system and therefore, in this study, we calculate the effectiveness of the CSA mitigations on the storm event loads and on the concentrations under low flow conditions which are most likely to impact on downstream water quality monitoring datasets.

Deer live in a variety of environments around the world but are predominantly a wild animal living in native forested areas. Deer were introduced into NZ in the mid-late 19th century and released into native forest areas for hunting. In the 1970's live captured deer were bred on farms and started the NZ deer farming industry (DINZ, 2023). A review of the environmental impacts of intensive deer farming identified that, due to specific behaviours of deer, soil and water quality were the main impacts of concern (de Klein et al., 2002). Deer impacts on water quality were confirmed by a review of contaminant losses from grazed pasture-based farming. which identified that deer farming had potentially higher contaminant losses than other dry-stock farming systems, particularly in terms of sediment losses (McDowell and Wilcock, 2008). Furthermore, there appeared to be greater variation in contaminant losses from deer farming which was assumed to be due to the potential losses from parts of the landscape impacted by deer behaviours such as wallowing and fence line pacing (de Klein et al., 2002). Studies have been conducted to investigate and mitigate contaminant losses from deer wallows, fencing line pacing and forage crops (McDowell et al., 2004; McDowell, 2007; McDowell, 2009; McDowell and Stevens, 2008). However, no studies have investigated the potential for CSA management of grazed pastures (by any grazing species) to mitigate losses to water.

Water quality standards and guidelines around the world are derived from large, perennially flowing rivers and are used to indicate the levels of contaminants in the water that will support the desired biodiversity of a healthy river and prevent the development of undesirable outcomes (e. g., Snelder et al., 2019). Ephemeral streams that will not have any of the habitat required to support desired stream biodiversity, regardless of the contaminant concentrations in the water. Furthermore, we would expect significant instream processing to attenuate the contaminants from ephemeral streams before they impact on the river network downstream (Alexander et al., 2007; Rafi et al., 2018). Therefore, the runoff data from ephemeral streams should not be directly compared to water quality standards with a pass/fail mindset. An alternative approach would be to compare the contaminant concentrations to local water quality standards (national bottom lines) in a qualitative approach with the assumption that: contaminant concentrations below the guidelines would be unlikely to have a downstream impact; contaminant concentrations at or slightly above the guidelines may have some impact downstream and; contaminant concentrations well above the guidelines have the potential to cause downstream impact. Furthermore, the contaminant concentrations during storm events are likely to impact further downstream due to rapid transit times and hence minimal attenuation.

To address these knowledge gaps relating to grazed pasture mitigations and impacts on runoff, this study investigated the management of CSAs within deer grazed pastures at a paddock (multiple hectare) scale in a hill country landscape. The methodology included remote sampling equipment to collect samples of runoff generated at any time of the day, or day of the week, for an extensive high-quality dataset. The measurements were conducted over four years using paired catchment design to account for weather difference between the pre- and post-treatment time periods (Kay et al., 2007). A total of four small catchments were monitored enabling one control catchment and three experimental catchments to test three different CSA management options.

2. Materials and methods

2.1. Site description

The study site was located on the AgResearch Invermay deer farm near Mosgiel, Otago, New Zealand (McDowell, 2022). Mean annual rainfall in the region has been reported as 687 mm and the predominant soil type is a Warepa silt loam (mottled fragic Pallic soil, NZ Soil Classification: equivalent to a typic Hapludalf in US Soil Taxonomy). The four small catchments were 3.6, 2.4, 3.4 and 2.2 ha for catchments 1 through 4, respectively (Fig. 1). The catchments were on a southwest facing slope at an altitude ranging from 300 to 380 m above sea level (Fig. 1). The maximum slopes of the catchments were 44, 37, 35 and 42 % and the average slopes were 15, 14, 12 and 13 % for catchments 1 through 4, respectively. Photographs are provided in the supplementary materials to illustrate the CSAs, treatments and general landscape of the study site. The pastures were predominantly ryegrass [*Lolium perenne*

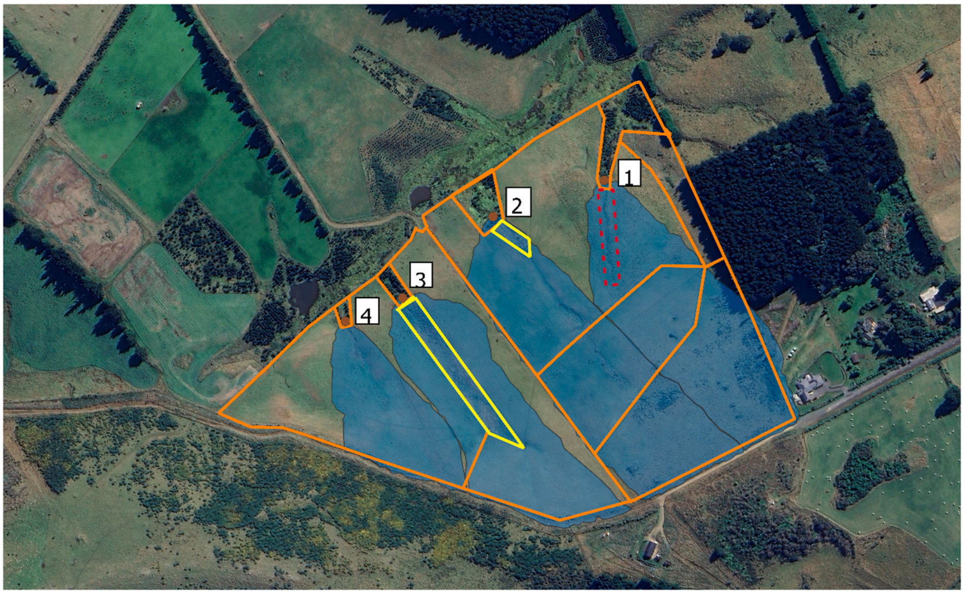

Fig. 1. Map of the four CSA experimental catchments at Invermay with the catchment numbers. The orange lines are the original paddock fencing at the start of the experiments. The yellow lines are the permanent fencing applied in the treatment phase of the experiment in catchments 2 and 3. The dotted red line shows where the temporary winter fencing is applied in catchment 1. The red dots are the location of the water quality sampling equipment in each of the catchments. The blue shaded areas are the respective catchment areas.

L.], white clover [*Trifolium ripens L.*] and brown top [*Agrostis capillaris*] and have been grazed predominantly by red deer since 1991. Throughout the 4 years of the experiment the pastures were grazed as a part of the whole 900 head deer farming operation on 194 ha of pasture area at Invermay, which involved fawning in these catchments during November each year. These catchments are ephemeral with runoff only occurring during periods of surplus rainfall, typically from late autumn until spring, referred to in this paper as the runoff period. A nutrient budget for the whole deer farm was conducted using the OverseerFM model (https://overseer.org.nz/overseerfm/#). Annual fertiliser application on the farm was 11 kg nitrogen ha^{-1} and 8 kg phosphorus ha^{-1}.

2.2. Experimental design

The experimental design is a paired catchment experiment (Kay et al., 2007) to test alternative management of the critical source areas (CSAs). In this steep hill country landscape, the CSAs were identified visually as small gullies where wet soils and deer hoof impacts had resulted in soil erosion and significant patches of bare soil likely to produce runoff (See Plate 1 in Supplementary Materials 1). In phase one, all 4 catchments were managed under a typical grazing rotation on the deer farm during 2021 and 2022 and water quality measurements conducted as below. During 2022 researchers worked with representatives of Deer Industry New Zealand (https://www.deernz.org/) environmental research portfolio and the Invermay Deer Farm Manager to select practical mitigation options for 3 of the 4 catchments as phase 2 of the paired catchment approach. The CSA management and water quality measurements were continued during 2023 and 2024. Catchment 4 remained as the “control” with no change in the farm management. This control is required to account for weather differences between years. Catchment 3 had the entire CSA area permanently fenced off and left to regrow as rank grass. This treatment represents the greatest environmental potential benefit and is referred to as the “fully fenced” treatment for the remainder of this paper. Catchment 2 had only the lower portion of the CSA area fenced off, leaving the upper portion of the CSA to continue with typical grazing as this was thought to be a compromise between production and environmental benefits and is referred to as the “partial fenced” treatment. The permanent fencing applied in Catchments 2 and 3 were installed in October 2022 after the runoff period and prior to fawning. Catchment 1 had temporary electric fencing during the runoff period when the soils were wet. The fence was removed in spring each year, prior to fawning to allow grazing for most of the growing season as an alternative compromise between production and environmental benefits and is hereafter referred to as the “temporary fenced” treatment. The decision on when to erect the electric fence in Catchment 1 was based on perceived soil moisture levels and weather forecasts of significant rainfall; these decision triggers resulted in the electric fences being erected on the 14th April 2023 and 10th May 2024. The temporary fencing was removed in September 2023 and no more runoff events were recorded that year. 2024 had higher spring rainfall and therefore, the fencing was only removed on the 5th November 2024 when the CSA area was still wet, as fawning started that week. This decision was guided by animal welfare reasons that required removal of electric fences in paddocks with young animals. In 2024 some runoff occurred after the temporary electric fence was removed. For clarity, when referring to the data from the different catchments we will use the labels CSA1 - CSA4, as technically the treatment is only applied in the post-treatment period. We use the name of the treatments when referring to the effectiveness of the treatments. Photographs of the control, temporary fenced and fully fenced CSAs are provided in the supplementary materials 1 (Plates 1 – 3).

2.3. Water sampling and laboratory analysis

At the outlet of each catchment a H-flume was installed with a stilling well attached. The flume walls were dug down into the poorly draining B horizon and were expected to capture and direct all the surface runoff and most of the shallow drainage water from the catchment through the flume for measurement. The water level through the flume was measured with a Unidata level sensor (O’Connor, Western Australia) connected to a Unidata Neon Metering module (O’Connor, Western Australia). Stage height was recorded every minute and data uploaded to the internet every 15 min. A rain gauge was installed at the Catchment 2 flume and used an electronic tipping bucket rain gauge calibrated to tip in 0.2 mm increments. Due to the compact location of the paired catchments (Fig. 1) the elevation difference between the rain gauge and highest point of the catchments is 80 m and all catchment areas were within a 500 m radius of the rain gauge. Water samples during low flow conditions were collected by grab sampling directly from water flowing through the flumes. The sample plan was that when water was flowing, and staff were available, a low flow sample was collected approximately weekly. Storm event samples were collected using a ISCO 3700 auto sampler with a 24-bottle base (Teledyne ISCO, Lincoln, USA), connected to the flume and was triggered to sample by the Neon logging system. At least 10 mm of water height in the flume was needed to draw in a water sample so the trigger level was initially set to 10 mm height and then to repeat sampling every hour thereafter. When the automatic samplers were triggered, the samples were collected within 24 h of the beginning of the event and returned to the laboratory at Invermay. If a stormflow event lasted more than 24 h, autosamplers were reset with clean bottles and sampling continued until the storm event slowed. The Neon logging system could be reprogrammed remotely enabling the autosampler trigger level to be raised when low flow conditions were above 10 mm flume height. The autosamplers were programmed to rinse the intake tubing once before collecting a 500 mL sample. The flumes are in remote locations and the Neon loggers and autosamplers were powered by solar panels and batteries. Therefore, the autosamplers were not refrigerated and samples were returned to the lab at ambient conditions in an insulated box. Harmel et al. (2016) found no difference in *E. coli* counts between refrigerated and ambient samples in Texas where ambient conditions were > 20 °C. Most of the samples collected in this study were during winter when the average daily air temperature is < 10 °C; we thus considered that refrigeration was not necessary. The bottles used in the autosampler bases were not autoclaved but were washed and rinsed in hot tap water and then dried overnight in an incubator at 45 °C before being reused. Furthermore, between stormflow events the autosampler bottles (which have no lids on them) can be in the autosampler base for weeks or even months. As a quality check of this cleaning process and field waiting time, every time storm event samples were collected, a deployed but unused bottle was selected from one of the samplers, 100 mL of sterile water added and then shaken for approximately 20 s before analysing the rinsing water for *E. coli*. *E. coli* was not detected in any of the quality control samples, providing confidence that the autosampler bottle washing system was appropriate and there was no contamination of the sample bottles in the field.

For each stormflow event, a minimum of 5 (or more for longer events) samples were selected to represent the hydrograph of the event i.e. increasing, peak and falling flow rates. In the lab, the selected samples were immediately analysed for *E. coli* using the Colilert and Quanti-Tray System (Muirhead et al., 2004) and then a sub-sample was frozen and stored for physical and chemical analysis. Physical and chemical analysis was for: total suspended solids (TSS; APHA 2540 D), total phosphorus (TP; APHA 4500-P H) and filtered reactive phosphorus (FRP; APHA 4500-P G), total nitrogen (TN; APHA 4500-N C), ammoniacal nitrogen (NH_4-N; APHA 4500-NH_3-N) and total oxidised nitrogen (NO_x-N = NO_3-N + NO_2-N; APHA 4500-NO_3^- I)). Due to changes in laboratory capacity, the nitrogen species analyses were conducted in different commercial laboratories for the pre- and post-treatment periods.

2.4. Data analysis

Exploratory analysis of the data was conducted by visual inspection of hydrographs, contaminant concentration data and relationships between flow and concentration. Statistical significance of the difference in contaminant concentrations between catchments or pre- and post-treatment were conducted using a student's t test with the *E. coli* data log transformed before analysis. Statistical significance was assumed when $p < 0.05$. Initial analysis of the concentration data was compared to New Zealand (NZ) national bottom lines, which are minimum water quality standards under NZ environmental legislation (MfE, 2024). These concentration comparisons were on a qualitative basis. As we had continuous flow data and discrete samples for contaminant concentration, we used method G from Defew et al. (2013) to interpolate and calculate loads of contaminants from each catchment. We used this method to calculate individual storm event loads and annual loads of contaminants for each catchment. The effectiveness of the mitigations at reducing loads were calculated from the individual storm event data. Individual storm event loads were assessed by the change in the ratio of individual storm loads from the treatment catchment (CSAs 1–3) to the control catchment (CSA4), pre and post the mitigations being applied. These loads were divided by the area of each catchment to present the results as yields per hectare. The effectiveness of the mitigations at reducing contaminant concentrations under low flow conditions was assessed by the change in the ratio of concentrations from the treatment catchment (CSAs 1–3) to the control catchment (CSA4) for low flow samples collected at the same time, pre and post implementation of the mitigations. The comparisons of loads and concentrations via the pair catchment experimental design were conducted using the methods of Bland and Altman (1999).

3. Results

3.1. Raw data

Annual rainfall over the 4 years of measurements was 1070, 1084, 1024, 1319 mm for 2021, 2022, 2023 and 2024, respectively (Fig. 2 and Table 1). These annual rainfall amounts appear to be higher than the published "long-term average" for this region (McDowell, 2022). The regional rainfall patterns were typical during the experimental period and therefore, the higher rainfall measured in the experimental catchments was due to naturally higher rainfall at the higher elevation of these catchments. The 2021 year started with significant atypical rainfall in early January and had already reached almost 200 mm of rain in the first week. This rainfall event would have generated runoff that was not captured as the measuring equipment was not installed in the catchments until April prior to the typical runoff period. The remainder of the year recorded just over 800 mm of rainfall making it a relatively dry runoff period compared to the other 3 years of measurements (Fig. 2). As a result of this relatively dry winter the reported runoff volumes from 2021 was less than the other 3 years (Table 1). Note – the paired catchment experimental design controls for different weather patterns across the time of the measurements and therefore, missing this atypical summer rainfall event will not bias the estimates of the long-term effectiveness calculation for these mitigation options.

The ephemeral streams of the catchments generally only generated runoff between April and October (Figures 3 and S1-S3). There were

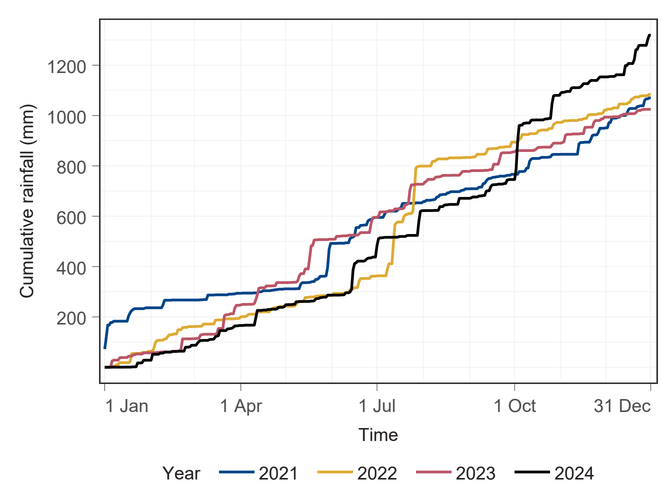

Fig. 2. Cumulative annual rainfall measured in the catchments each year. Note the runoff monitoring equipment was installed in April 2021 and would not have captured runoff from the heavy rainfall event in early January 2021.

Table 1
Annual rainfall and runoff volumes (mm) from the 4 catchments for each year of measurement.

	Rainfall	Runoff			
		CSA1	CSA2	CSA3	CSA4
2021*	1070	78	89	97	134
2022	1084	223	285	203	234
2023	1024	144	179	146	226
2024	1319	212	328	221	388

* Note the runoff measuring equipment was only installed in April 2021 and therefore the runoff volumes will not reflect the full years rainfall (see Fig. 2).

small differences in the annual patterns of runoff between the 4 catchments, but the hydrologic responses appeared very similar in timing and magnitude of runoff indicating that these catchments were suitable for a paired catchment experimental design. There were distinctly different patterns of runoff between years with 2023 showing large gaps between runoff events when flow stopped compared to 2022 where low flows continued between the first 6 events of the year (Figures S1 & S2). Runoff samples were captured from 9, 7, 10 and 9 events across the 4 years. The measured volumes of runoff over the 12 catchment years ranged from 78 to 388 mm with an average of 199 mm (Table 1). The Overseer model uses a 30-year average climate soil water balance model to estimate average drainage to 60 cm soil depth. For this deer farm model, the Overseer estimated average drainage was 194 mm which is very close to the 199 mm average measured in these experiments which gives us confidence that measured results in this study have captured the surface and shallow drainage from these small catchments.

Low flow responses were more variable, with the different catchments drying up at different times. A summary of the low flow samples collected is shown in Table 2. All low flow sample concentrations are displayed in Figs. 4 & 5 and S4 – S8 of the data. However, only the paired samples (collected at the same time) that corresponded with a sample collected from CSA 4 (the control catchment) could be used in the paired catchment data analysis shown in Figs. 6 & 7 and S9 – S12. In total 1162 (low-flow and event-flow) samples were analysed over the 4 years of measurements split relatively evenly between the 4 catchments.

Median concentrations of ammonia-N in any of the blocks of samples was less than the 0.2 mg L^{-1} national bottom line (MfE, 2024) and only outlier results exceeded the national bottom line of 0.4 mg L^{-1} for 95th

Table 2
Summary of the maximum number of low flow samples collected per year and maximum number of “paired samples” where samples were collected from both the control and a treatment catchment at the same time.

Year	Total samples	Paired samples
2021	12	10
2022	12	6
2023	7	6
2024	23	20

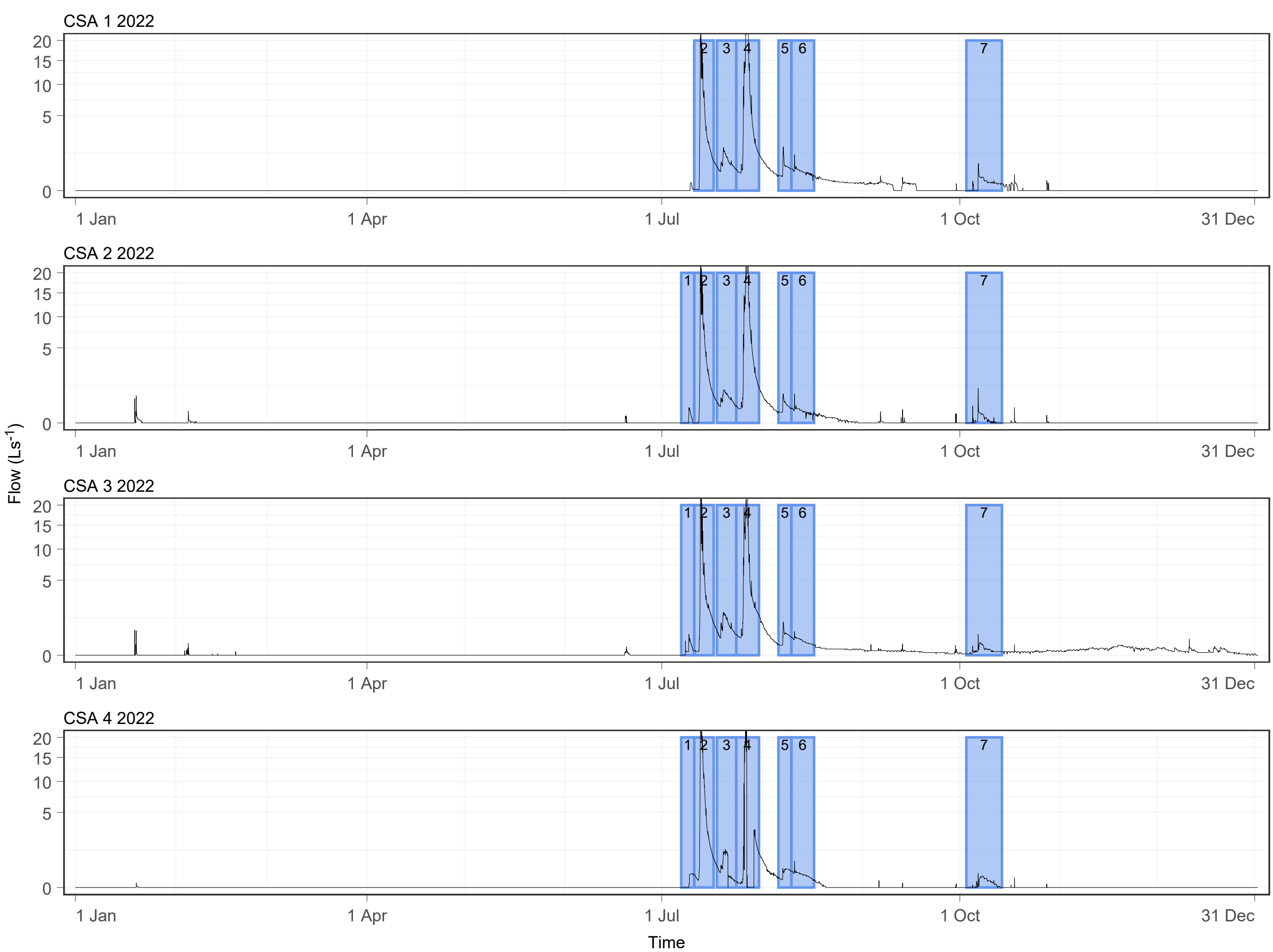

Fig. 3. Flow rates of runoff measured in the four catchments during 2022. The blue boxes and numbers identify the individual storm flow events captured that year. Note the non-linear scale used on the Y axis.

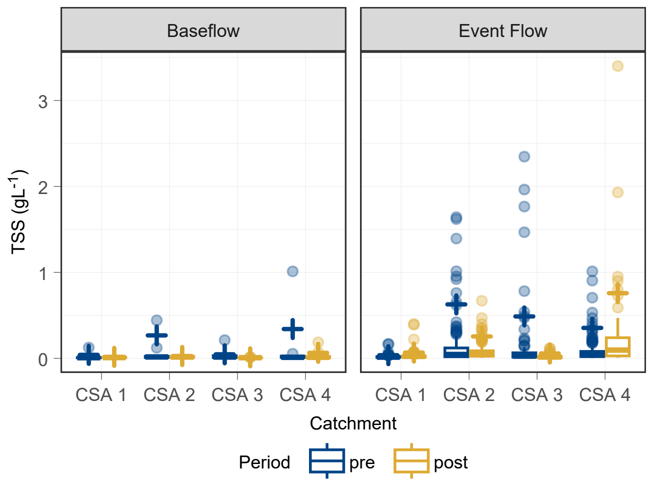

Fig. 4. Total suspended sediment concentrations in the samples collected in the four catchments, split into samples collected during low flow and event flow conditions and also split into the pre- and post-treatment periods. The horizontal line represents the median concentration, the boxes represent the interquartile range, the whiskers represent the quartile plus 1.5 times the interquartile range, the points represent the outliers and the crosses represent the 95th percentile.

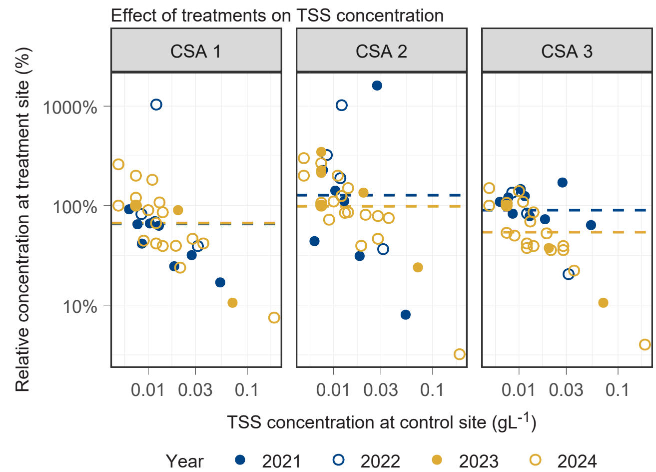

Fig. 6. Low flow total suspended sediment concentrations in the treatment catchments relative to the control catchment for each year. The blue dotted line represents the relationship pre-treatment, and the yellow dotted line represents the post-treatment relationship. The difference between the yellow and blue lines represents the effectiveness of the different treatments. Note the log scale on the Y axis.

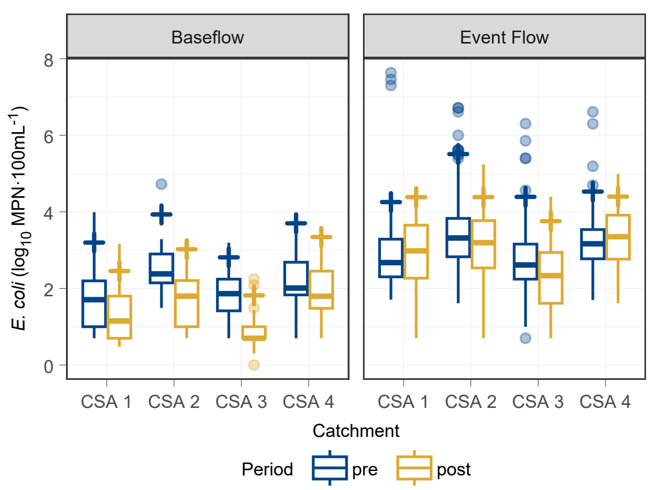

Fig. 5. Log_{10} *E. coli* counts in the samples collected in the four catchments, split into samples collected during low flow and event flow conditions and also split into the pre- and post-treatment periods. The horizontal line represents the median count, the boxes represent the interquartile range, the whiskers represent the quartile plus 1.5 times the interquartile range, the points represent the outliers and the crosses represent the 95th percentile.

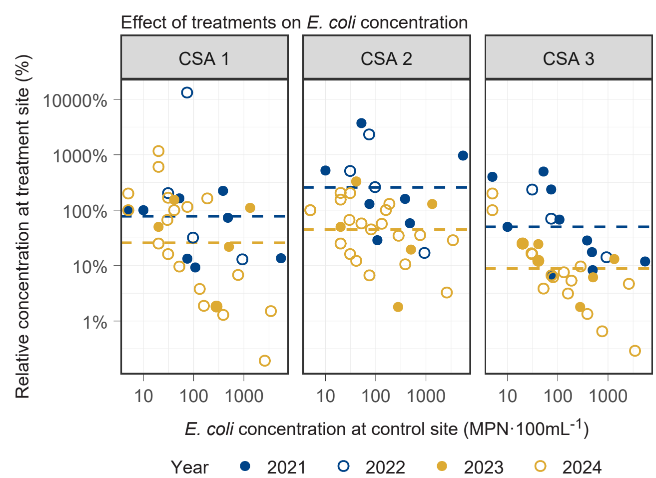

Fig. 7. Low flow *E. coli* counts in the treatment catchments relative to the control catchment for each year. The blue dotted line represents the relationship pre-treatment, and the yellow dotted line represents the post-treatment relationship. The difference between the yellow and blue lines represents the effectiveness of the different treatments. Note the log scale on the Y axis.

percentile (Figure S4). This indicates that ammonia concentrations in the runoff from deer grazed pastures are unlikely to have a down-steam impact on water quality. There was no significant difference between the low and event flow concentrations of ammonia ($p > 0.05$). The ammonia concentrations were significantly ($p < 0.05$) higher in the pre-treatment period compared to the post-treatment period under low flow conditions for all 4 catchments, but only significant for the three treatment catchments under event flows. This apparent reduction in effect under low flows was observed in CSA4 that had no change in farm management or treatment over the 4 years, indicating that it may have been caused by the change in labs used to analyse the nitrogen samples between 2022 and 2023. The analysis of the data from the paired catchment experimental design uses the data from the control catchment (CSA4) as the basis of the comparisons. As the results from individual years were all analysed in the same lab, we feel it is still appropriate to use this data in the mitigation effectiveness analysis. We did not notice any change in other nitrogen species associated with the change in analytical laboratory.

The dominant nitrogen species in the runoff was total oxidised nitrogen (Figure S5). The total oxidised nitrogen species were significantly higher in event flows than during low flow conditions ($p < 0.05$). For total oxidised nitrogen the national bottom lines are a median of 2.4 mg L^{-1} and a 95th percentile of 3.5 mg L^{-1} and the medians of all treatments were below the median bottom line, all low flow samples were below the 95th percentile standard, but some event-flow runoff samples exceeded the 95th percentile standard (Figure S5). This implies that the runoff from deer grazed pasture is unlikely to have a downstream impact on total oxidised nitrogen levels, particularly during low flow conditions.

There were large differences in the concentrations of FRP and TP between the low flow and event samples (Figures S7 & S8). The national bottom-line for FRP is a median of 18 μg L^{-1} and the low flow samples met this standard, but the event runoff samples exceeded this standard (Figure S7). The national bottom-line 95th percentile for FRP is 54 μg L^{-1} and only 2 of the low flow samples exceeded this level but many of the event flow samples were greater than this level (Figure S7). This indicates that low flow runoff is unlikely to impact on low flow water quality downstream, but the storm event runoff is likely to impact on downstream water quality. The FRP and TP concentrations were very similar between the pre and post treatment periods in the control catchment (CSA 4).

There are no national bottom lines for TSS as the water quality metrics are based on visual clarity (MfE, 2024). As expected, the TSS concentrations in low flow were much lower than in event flows but the fully and partially fenced treatments still significantly ($p < 0.05$) reduced TSS concentrations under low flow conditions (Fig. 4). The event flow concentrations of TSS in CSA1 was significantly ($p < 0.05$) less than the control catchment in the pre-treatment period (Fig. 4). At the start of the experiment CSA1 had the least amount of soil physical damage and bare soil – which was a reason that the temporary fencing mitigation option was selected for this catchment.

The *E. coli* counts in low flow were significantly lower than in event flow (Fig. 5). The national bottom line for median *E. coli* counts is 130 most probable number (MPN) 100 mL^{-1} (Log_{10} 2.1 MPN 100 mL^{-1}) and from the samples collected during low flow conditions, only the samples from the pre-treatment period exceeded this level (Fig. 5). The national bottom line for 95th percentile *E. coli* counts is 1200 MPN 100 mL^{-1} (Log_{10} 3.1 MPN 100 mL^{-1}). Some samples exceeded the 1200 MPN 100 mL^{-1} level under low flow conditions for all catchments pre- and post-treatment, with the exception of CSA3 where the fully fenced treatment was applied and only 1 sample exceeded the level of 1200 MPN 100 mL^{-1} (Fig. 5). Under event-flow conditions all catchments exceeded the national bottom-lines for *E. coli* (Fig. 5). This indicates that low flow runoff from the fully fenced treatment would be unlikely to have a downstream impact on water quality, but storm flow runoff may have an effect. Note that the *E. coli* counts are highly variable with approximately 5 orders of magnitude variation under low flow conditions and 7 orders of magnitude variation under storm flow conditions (Fig. 5).

3.2. Mitigation effectiveness – low flow concentrations

For this analysis, shown in Figs. 6, 7, S9 – S13, in a hypothetically perfect result, all of the pre-treatment data points would be plotted on a horizontal line at Y = 100 % and the post-treatment data points would be plotted on a horizontal line at Y < 100 %. The scatter of data points around these calculated lines is the variability of the data.

When comparing the low flow contaminant concentrations between the pre- and post-treatment periods for individual catchments there were some statistically significant differences. However, when we conducted the analysis using the paired catchment methodology (Kay et al., 2007), none of the analyses were statistically significant (all tests $p > 0.16$) which is likely due to the highly variable results from environmental data. With a paired catchment experimental design, results from two catchments are compared and, therefore, the variability is combined, thus making it difficult to generate statistically significant results. This can be seen in the data in Figs. 6, 7, S9 – S13 where the concentrations for chemicals and sediment can be 10-fold different and the *E. coli* can be 100-fold different between the paired catchments. However, the paired catchment design does account for differences in weather patterns between pre- and post-treatment periods which prevents incorrect conclusions being drawn where it could be assumed that the size of the changes in a single catchment has been caused by the treatment, when it may have in fact been caused by the different weather patterns. Therefore, we have presented the percentage reduction values as calculated by the paired catchment experimental design and used the pre- and post-treatment changes in concentrations assessed by the t test to support the statistical significance of the reductions (Table 3).

The analysis indicates reductions in low flow ammonia concentrations from all 3 treatments ranged from 46 % to 70 % and were statistically significant (Figure S9). The total nitrogen low flow concentrations were much higher than ammonia concentrations but only the partially fenced mitigation significantly reduced TN concentrations (Figure S11).

The phosphorus data implies that the temporary fencing and fully fenced mitigations resulted in an increase in FRP concentrations but none of these changes in FRP concentrations were statistically significant ($p > 0.05$). The fully and partially fenced mitigations significantly reduced the low flow concentrations of total P (Figure S8). The sediment data indicated that the temporary fencing mitigation did not reduce the low flow concentrations but the partially and fully fenced mitigations significantly reduced the concentrations (Table 3 and Fig. 6). However, sediment losses were minimal during low flow conditions. The *E. coli* low flow counts all showed statistically significant reductions (67–83 %) in response to the mitigations (Table 3 and Fig. 7).

3.3. Mitigation effectiveness – storm-event runoff

In this analysis we have calculated the effectiveness of mitigations on storm event runoff based on reductions in individual event loads. We have calculated the annual loads from each catchment for each year but have not used the annual data for the effectiveness calculations for two reasons. Firstly, as we only have 2 years of data pre- and post-treatment, we effectively have n = 2 for the annual data compared to using individual events where n > 14. Secondly, the calculation of the annual loads is dominated by the largest event recorded in each year and, therefore, the effectiveness calculation will be dominated by the effectiveness for that single largest event (data not shown). Whilst the information we are interested in is the potential effectiveness of these mitigation measures over the long-term (i.e. including weather patterns across multiple years) we feel that calculating the storm-flow mitigation effectiveness on an individual runoff event basis will provide the best estimate of the long-term effectiveness of these mitigation actions.

The temporary fencing mitigation had no effect on the storm-flow ammonia runoff although it should be noted that the ammonia concentrations in CSA1 were significantly lower compared to CSA4 in the pre-treatment period, potentially because of the high pasture cover in CSA1 at the start of the experiments (Table 4 and Figure S14). The partial and fully fencing mitigations significantly reduced the storm-flow event loads of ammonia by 61 and 88 %, respectively. Similarly, there was effectively no mitigation effect of the temporary fencing on total nitrogen loads in runoff (1 %) and significant reductions of the partial and full fencing mitigations of 17 and 32 %, respectively (Table 4 and Fig. 8).

The FRP results imply that the mitigations either had no effect or increased the amount of FRP lost from these catchments in storm-flow (Figure S16). The temporary fencing mitigation appeared to increase

Table 3
Summary of the percentage reductions for ammonia (NH_4-N), total oxidised nitrogen (NO_x-N), total nitrogen (TN), filtered reactive phosphorus (FRP), total phosphorus (TP), total suspended solids (TSS) and *E. coli* from the CSA treatments for reducing the low-flow concentrations calculated by the paired catchment methodology. A negative value indicates an increase in concentration. The reductions in bold are supported by a statistically significant ($p < 0.05$) reduction in pre- and post-treatment concentrations by the t test.

CSA Treatment	NH_4-N (%)	NO_x-N (%)	TN (%)	FRP (%)	TP (%)	TSS (%)	*E. coli* (%)
Temporary fencing	**46**	79	27	-19	-1	-2	**67**
Partial fencing	**70**	**71**	**18**	42	**34**	**23**	**83**
Fully fenced	**65**	37	31	-19	**50**	**40**	**82**

Table 4

Summary of the percentage reductions for ammonia (NH_4-N), total oxidised nitrogen (NO_x-N), total nitrogen (TN), filtered reactive phosphorus (FRP), total phosphorus (TP), total suspended solids (TSS) and *E. coli* from the CSA treatments for reducing the individual event loads (proxy for long-term annual load) calculated using the paired catchment methodology. A negative number indicates an increase in loads. The reductions in bold are supported by a statistically significant ($p < 0.05$) reduction in pre- and post-treatment concentrations by the t test.

CSA Treatment	NH_4-N (%)	NO_x-N (%)	TN (%)	FRP (%)	TP (%)	TSS (%)	*E. coli* (%)
Temporary fencing	**2**	42	1	-59	-21	**28**	34
Partial fencing	**61**	**9**	**17**	2	**15**	**51**	**56**
Fully fenced	**88**	**5**	**32**	-60	**45**	**87**	**93**

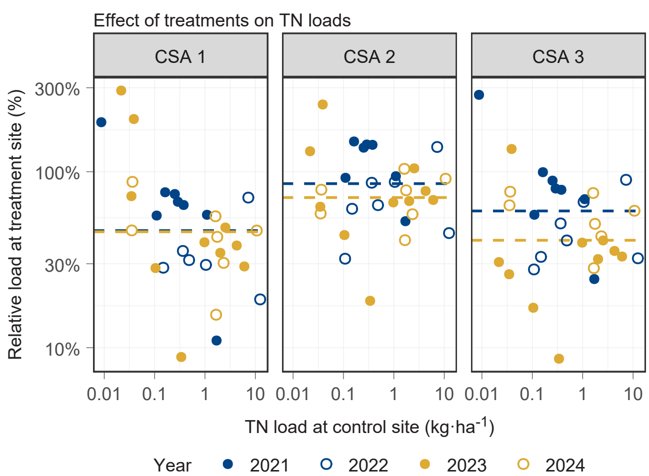

Fig. 8. Total nitrogen event loads in the treatment catchments relative to the control catchment for each year. The blue dotted line represents the relationship pre-treatment, and the yellow dotted line represents the post-treatment relationship. The difference between the yellow and blue lines represents the effectiveness of the different treatments. Note the log scale on the Y axis.

the runoff of TP, but it is important to note that the TP losses from this catchment were much lower than the control catchment to start with (i.e. the pre-treatment line in Fig. 9 was approximately 50 %). The partial and fully fenced mitigation options significantly reduced TP runoff loads by 15 and 45 %, respectively (Table 4 and Fig. 9). The temporary, partial and full fencing mitigation options significantly reduced sediment runoff by 28, 51 and 88 %, respectively (Table 4 and Fig. 10). As with the TP results, the sediment runoff loads from CSA1 were much less than the control catchment (CSA4) at the beginning of the experiments (Fig. 10) due to the lower levels of bare ground.

The temporary fencing mitigation did not significantly reduce the storm event *E. coli* loads (Table 4). The partial and fully fenced mitigations significantly reduced the storm-flow loads of *E. coli* by 56 and 93 %, respectively (Table 4 and Fig. 11).

3.4. *Annual losses*

As there were no treatments applied in the first two years of all 4 catchments, and all 4 years from CSA4, there are effectively 10 "catchment years" of data of contaminant losses from these pastures grazed by deer without implementation of CSA mitigations (Table 5). The total nitrogen losses ranged from 2.0 to 23.1 with an average of 11.1 kg ha^{-1} year^{-1} which encompassed the estimate from the Overseer model of 10 kg ha^{-1} year^{-1}. The Overseer model is a farm scale decision support tool that estimates the long-term average of nitrogen losses and

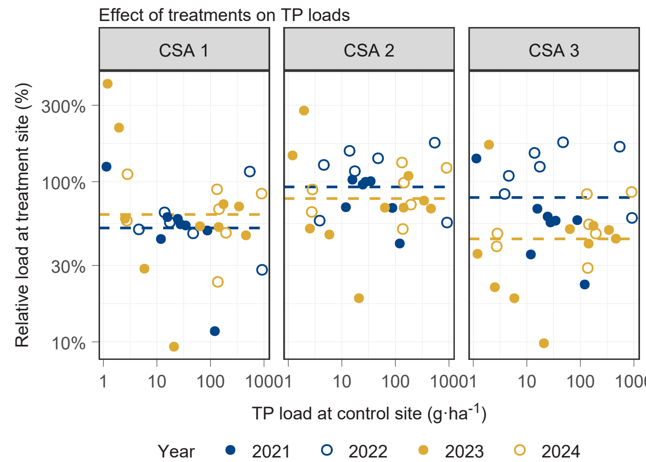

Fig. 9. Total phosphorus event loads in the treatment catchments relative to the control catchment for each year. The blue dotted line represents the relationship pre-treatment, and the yellow dotted line represents the post-treatment relationship. The difference between the yellow and blue lines represents the effectiveness of the different treatments. Note the log scale on the Y axis.

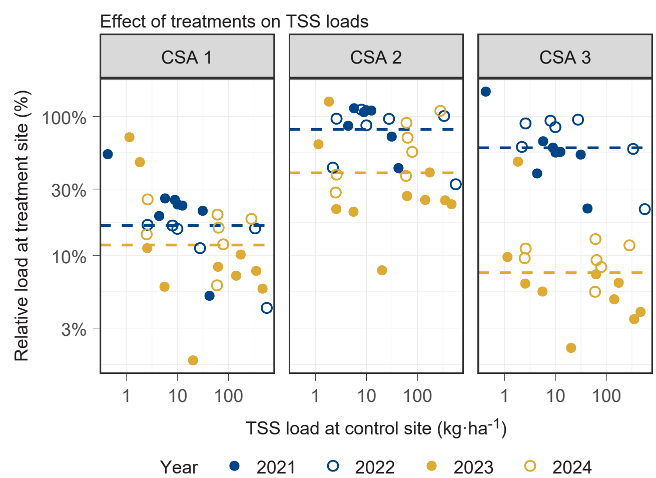

Fig. 10. Total suspended solids event loads in the treatment catchments relative to the control catchment for each year. The blue dotted line represents the relationship pre-treatment, and the yellow dotted line represents the post-treatment relationship. The difference between the yellow and blue lines represents the effectiveness of the different treatments. Note the log scale on the Y axis.

therefore, is more appropriate to compare to the average of the 10 "control years" of data, rather than individual years. Total phosphorus losses from the 10 "control years" ranged from 0.15 to 1.68 with an average of 1.0 kg ha^{-1} year^{-1}. In the Overseer model there are options for deer farm managements to indicate the presence of deer wallowing on a farm and deer wallowing did occur in the unprotected CSAs in the experimental catchments. With the deer wallowing behaviours option activated the long-term estimate of P losses from the Overseer model was 1.0 kg ha^{-1} year^{-1} matching the average measured data (Table 5). This measured data indicates that the Overseer model appears to be representing the long-term average nitrogen and phosphorus losses from the deer grazed pasture system in this landscape.

Annual sediment losses measured from untreated catchments in this study ranged from 23 to 1250 with an average of 421 kg ha^{-1} year^{-1} (Table 5). This study represents the first published data on the annual losses of *E. coli* from deer grazed pastures and ranged from 1.5×10^8–9×10^9 with an average of 2.7×10^9 MPN ha^{-1} year^{-1} (Table 5).

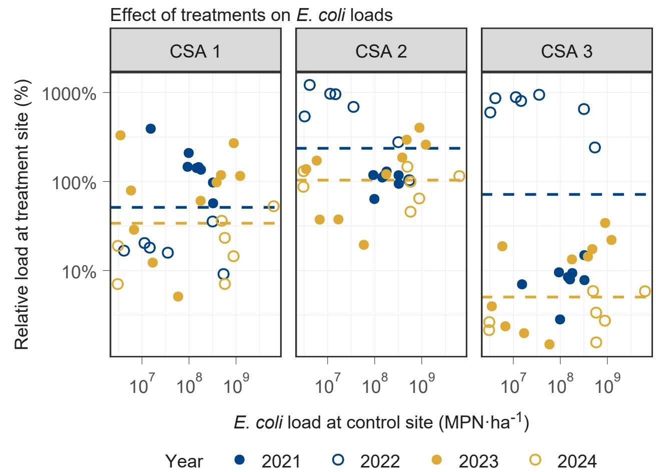

Fig. 11. *E. coli* event loads in the treatment catchments relative to the control catchment for each year. The blue dotted line represents the relationship pre-treatment, and the yellow dotted line represents the post-treatment relationship. The difference between the yellow and blue lines represents the effectiveness of the different treatments. Note the log scale on the Y axis.

4. Discussion

There is a need for the development of additional mitigation options to reduce contaminant losses from grazed pastoral farming systems (McDowell et al., 2021). Here we investigated three mitigation approaches targeting CSAs, to reduce contaminant runoff from pasture grazed by deer. Furthermore, mitigation effectiveness was assessed for reducing low flow concentrations and storm event loads to understand the potential impact of these mitigations on down-stream water quality metrics.

This study conducted measurements of water quality contaminants in runoff from four individual catchments over four years and therefore, represents a large dataset on contaminant losses from deer grazed pastures. McDowell and Wilcock (2008) reviewed data from 38 catchment studies to investigate the contaminant losses from different land uses in NZ. Their study identified that mean nitrogen losses from deer farms (8 kg ha^{-1} year^{-1}) were much less than from dairy farms (27 kg ha^{-1} year^{-1}). Annual nitrogen losses from deer farms ranged from 3 to 19 kg ha^{-1} (McDowell and Wilcock, 2008) which aligns with the individual year results from this study of 2–23 kg ha^{-1}. The nitrogen losses are likely a result of the more extensive deer farming system with relatively low nitrogen fertiliser inputs compared to dairy farms.

Phosphorus yields measured in this study, from catchments without mitigation, ranged from 0.15 to 1.68 kg ha^{-1} year^{-1} which is similar to the range reported by McDowell and Wilcock (2008) of 0.6–1.8 kg ha^{-1} year^{-1} for deer grazed pastures. McDowell and Wilcock (2008) found that deer farms had the highest reported losses of sediment from all the land uses and was much higher than for dairy farms. They attributed the high sediment losses from deer farms to deer behaviours of wallowing and fence-line pacing and the more erosion prone hilly landscapes where deer are typically farmed. This study's measured sediment yields which ranged from 23 to 1250 with an average of 421 kg ha^{-1} year^{-1}, were lower than the range from 158 to 3950 with an average of 2034 kg ha^{-1} year^{-1} reported by McDowell and Wilcock (2008).

There is much less published data on *E. coli* losses from grazed pastures than other contaminants (McDowell and Wilcock, 2008). These annual losses of *E. coli* appear to be similar to the reported losses from cows grazing winter forage crops (6.7 $\times 10^{8}$–1.1 $\times 10^{10}$ MPN ha^{-1} year^{-1}) but lower than for sheep grazed forage crops or pasture (2 $\times 10^{11}$ and 1.2 $\times 10^{11}$ MPN ha^{-1} year^{-1}, respectively) (Monaghan et al., 2017; Ghimire et al., 2024). *E. coli* losses from sheep grazed pastures are high due to the high counts of *E. coli* in sheep faeces relative to other grazing animal species such as cattle and deer (Muirhead, 2023; Moriarty et al., 2015). Interestingly, these deer grazed pasture-measured losses (and from other studies) are all less than modelled estimates for all pasture areas (8.8 $\times 10^{11}$ MPN^{-1} ha^{-1}) derived from river water quality samples (Elliott et al., 2016).

These annual loads indicate that the contaminant losses measured in this study are typical of losses from dry-stock farming systems across NZ. Therefore, the impacts of the mitigations applied in this study should have applicability to other grazing species in rolling to hilly landscapes. This study indicates that at a large catchment scale with multiple land uses, deer farms have the potential to make a significant contribution to

Table 5
Annual yields of total nitrogen (TN), total phosphorus (TP), sediment (TSS) and *E. coli* from the four catchments for each year. The greyed-out cells represent the control catchments when the 4 catchments were managed according to traditional farm management practices with un-protected CSAs. Only the greyed-out cells are used to calculate the "control average" annual contaminant losses from the deer grazed pasture.

Catchment	Year	TN (kg/ha)	TP (kg/ha)	TSS (kg/ha)	*E. coli* (MPN/ha)
CSA 1	2021	2.0	0.15	23	3.2 x 10^9
CSA 2	2021	3.9	0.25	97	1.6 x 10^9
CSA 3	2021	2.6	0.17	61	1.5 x 10^8
CSA 4	2021	4.1	0.34	122	1.5 x 10^9
CSA 1	2022	8.5	0.95	86	1.8 x 10^8
CSA 2	2022	18.3	1.65	597	2.4 x 10^9
CSA 3	2022	12.7	1.68	404	4.6 x 10^9
CSA 4	2022	23.1	1.60	980	9.7 x 10^8
CSA 1	2023	6.4	0.73	96	4.9 x 10^9
CSA 2	2023	13.0	0.95	337	9.2 x 10^9
CSA 3	2023	6.0	0.59	60	7.5 x 10^8
CSA 4	2023	16.8	1.24	1250	3.3 x 10^9
CSA 1	2024	8.0	1.15	96	3.8 x 10^9
CSA 2	2024	15.5	1.70	500	9.5 x 10^9
CSA 3	2024	10.7	1.14	63	4.6 x 10^8
CSA 4	2024	18.9	1.58	593	9.0 x 10^9
Control average		**11.1**	**1.0**	**421**	**2.7 x 10^9**

phosphorous, sediment and *E. coli* losses to water.

Overall, the temporary fencing mitigation option had no effect on the storm event runoff loads of oxidised and total nitrogen, but partial and fully fenced mitigations significantly reduced storm event runoff loads. All three mitigations significantly reduced ammonia concentrations in low and event runoff. However, ammonia concentrations in low flow and storm-flows were predominantly lower than the water quality standards in NZ and therefore, ammonia reductions would not be a key target of CSA mitigation options. Total oxidised nitrogen is a much larger proportion of the total nitrogen load than ammonia. The median concentration of oxidised nitrogen in all runoff samples indicated that the runoff from these deer grazed pastures would achieve the median water quality standards downstream (Figure S5). The low flow runoff concentrations would be likely to achieve the 95th percentile standards downstream but the storm event runoff could exceed the 95th percentile standards downstream. The percent reductions of the partial and fully fenced mitigations were relatively small for the storm runoff and therefore, even with mitigations, runoff from deer grazed pastures could impact on downstream 95th percentile nitrate water quality metrics. The observation of lower oxidised nitrogen losses from CSA1 catchment prior to the treatments and the relative effectiveness of the temporary fencing mitigation in this catchment supports the research from lysimeter studies that have demonstrated the urine patches on wet soils and low forage plant uptake are key drivers of nitrate leaching from farm systems (Talbot et al., 2020). These observations from lysimeter studies have been supported by studies that have shown that cover crops (minimising bare ground) can reduce nitrogen losses at a catchment scale (Vincent et al., 2025). As oxidised forms of nitrogen species (e.g. nitrate) are the dominant species of total nitrogen transported in water, the patterns of total nitrogen losses from these deer grazed pastures, reflect the patterns in oxidised nitrogen losses described above.

The effectiveness of the mitigation options for reducing FRP was highly variable with the data indicating that the mitigation may have increased FRP losses from these catchments. This variability in the FRP data is likely to be due to chemical reactions in the soils and water indicating rapid turnover of phosphorous chemical species in ephemeral streams. There were high concentrations of FRP in the storm event loads that may have downstream impacts and the mitigation options did not significantly reduce the event loads of FRP (Figures S7 and S16). The temporary fencing mitigation did not reduce TP concentrations under low flows but it should be noted that the pre-treatment storm event loads in CSA1 (that had the temporary fencing mitigation applied) where approximately half the loads in the control catchment which is likely due to the higher pasture cover in CSA1 pretreatment. The fully fencing mitigation was more effective for reducing TP concentrations under low flow and storm event loads than the partial fencing mitigation. These combined results strongly support the observation that plant cover is important to reducing TP loss from landscapes (Butler et al., 2006).

A key driver of sediment erosion is the energy of rain drop impact on a soil surface that dislodges soil particles (Renard et al., 1997). Therefore, it is not surprising that TSS concentrations under low flow runoff (without rainfall) is much lower than under storm event conditions (Fig. 4). Consequently, it is not surprising that the mitigations were much more effective at reducing storm event loads of TSS than low flow concentrations (Tables 3 and 4 and Figs. 6 and 10). The effectiveness of the mitigations for TSS followed the same pattern as for TP reinforcing the importance of plant cover for reducing sediment erosion (Renard et al., 1997; Butler et al., 2006).

All mitigation options were effective at reducing *E. coli* losses for both low flow counts and storm event loads. There was a clear pattern of increasing effectiveness from the temporary, to partial, to fully-fenced mitigation options (Tables 3 and 4) which is likely driven by the reduction of faecal pats deposited into the CSAs. Of note is that the low flow runoff counts from CSA3 with the fully fenced mitigation option would achieve an A grade for swimming water quality standards in NZ (MfE, 2024). However, the *E. coli* counts during storm runoff are well above the swimming water quality standards. How these storm flow counts impact on downstream swimming water quality standards requires further research (Rafi et al., 2018), particularly as it is recognised that people generally avoid contact with rivers in flood.

Overall, these results show a general pattern of the CSA mitigation options being more effective for phosphorous, sediment and *E. coli* than for nitrogen. This is consistent with the CSA concept of targeting surface flow pathways that are the dominant transport pathway for phosphorous, sediment and *E. coli* (McDowell et al., 2025). The effectiveness of the three mitigation options tested generally showed an increasing effectiveness from the temporary, to partial to fully-fenced option. The temporary and partial fencing options still allowed animals to graze, defecate and urinate in parts of the CSA at different times. Therefore, it is logical that the fully fenced option, preventing animal access to the CSA, would be the most effective option. Furthermore, it is important recognise that these mitigations can reduce the losses from multiple water quality contaminants and hence will have multiple benefits downstream i.e. for human health, habitat and stream health. However, it needs to be recognised that the fully fenced option is also the most expensive option in terms of fencing cost and loss of grazing area. A cost benefit analysis of these results is presented in supplementary materials 3 which showed that sometimes the partial or temporary fencing mitigations could be more cost effective for nitrogen, sediment or *E. coli*. It is recognised that there are a lot of variables associated with installing fences in hilly landscapes and actual costs will need to be assessed on a farm-by-farm basis. While the temporary fencing option may be cost effective the lower effectiveness needs to be considered with regards to the level of mitigation needed to achieve downstream water quality standards which will be catchment specific.

Another way to think of these mitigation options is as a progression. By starting with the temporary fencing option, the level of CSA protection could be determined, whether pastures recover and preventing areas of bare soil. If the temporary fencing option is effective for protecting the soils, and the downstream water quality in the catchment is good, then it could be a long-term solution. However, if the animal treading damage during the non-runoff period continues to cause soil pugging and patches of bare soil in the CSA then progress to permanent fencing might occur. Progression of the permanent fencing could start at the bottom moving up the CSA. If a CSA runs through multiple paddocks, permanent fencing could initially start within the lower paddock progressing up to spread the capital cost of fencing over several years.

5. Conclusions

Critical source areas have been identified as the most cost-effective areas of a catchment to target for mitigation actions on farms. This study evaluated the effectiveness of applying temporary, partial and full fencing of the critical source areas in a paired catchment experimental design. The effectiveness of the three mitigation options for reducing low flow concentrations ranged from not effective for filtered reactive phosphorus, total phosphorus, and sediment to 83 % effective for *E. coli*. The effectiveness of the three mitigation options for reducing storm-flow event loads ranged from not effective for filtered reactive phosphorus to 93 % effective for *E. coli*. The critical source area managements all mitigate multiple contaminants and hence will have multiple water quality benefits downstream.

CRediT authorship contribution statement

Peter Green: Writing – review & editing, Software, Formal analysis. **Muirhead Richard:** Writing – review & editing, Writing – original draft, Visualization, 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. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agwat.2025.110106.

Data availability

Data will be made available on request.

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