Reducing the impacts of sheep winter forage crop grazing on water quality

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1. Executive Summary

Winter forage crop grazing is an important feature of many farms in New Zealand that experience cool and wet winters. It involves the use of a crop (usually brassica or fodder beet) that is grown during spring, summer and autumn and then fed to animals during winter as a standing crop. Winter grazing is known to make a significant contribution to total losses of contaminants transported from dairy farms to water. Very little information is available that documents losses when sheep are used to graze these crops, however. A paired catchment study approach was therefore used to quantify fluxes of Nitrogen (N), Phosphorus (P), sediment and *Escherichia coli* (*E. coli*) in surface runoff from fields where sheep were wintered on swede (*Brassica napus*) and kale (*Brassica oleracea*) crops. The effectiveness of a strategic grazing approach that protected critical source area (CSA) was examined to determine if these fluxes could be reduced. The CSA protection in this study involved establishing and maintaining a pasture cover in CSAs ahead of the winter grazing period and excluding livestock from those areas. Estimated fluxes of N, P and sediment in surface runoff recorded in winter (June – August) and early spring (September) were up to 0.64, 0.22 and 51 kg ha^{-1}, respectively, and up to 2 x 10^{11} MPN ha^{-1} for *E. coli*. These fluxes in surface runoff were much less than fluxes reported in other studies which may reflect the relatively benign weather conditions and grazing pressures recorded during the study period. Protection of CSAs considerably reduced contaminant fluxes compared to standard grazing practice. Reductions in N, P, sediment and *E. coli* were 38%, 48%, 55% and 63%, respectively. These reductions can largely be attributed to lower concentrations of contaminants in surface runoff when the CSA was protected.

2. Background

Winter forage crop grazing is a key part of many farms in New Zealand that experience relatively cool winters. The practice of grazing forage crops (usually brassica or fodder beet) during the winter can have significant adverse effects on the environment. Winter grazing usually involves the grazing of forage crops on relatively wet soils and at high stocking densities that can sometimes exceed the equivalent of 1000 cows ha^{-1}. These grazing intensities and wet soil conditions typically result in damage to the soil surface due to livestock treading which dislodges soil particles and causes pugging and/or compaction (Drewry and Paton, 2005; Houlbrooke et al., 2009; Monaghan et al., 2017). Soil compaction and pugging tend to restrict the infiltration of rainfall into soil, increasing the risk that contaminants found in livestock excreta (e.g., phosphorus (P), nitrogen (N), faecal microbes) and sediment will be transported to waterways in surface runoff.

Few studies have quantified contaminant losses in surface runoff from grazed winter forage crops. Suspended sediment loads of up to 3700 kg ha^{-1} $year^{-1}$ and phosphorus losses of 4 kg ha^{-1} $year^{-1}$ have been reported for winter-grazed forage crops (McDowell, 2006; McDowell and Houlbrooke, 2009; Burkitt et al., 2017; Monaghan et al., 2017). These losses were primarily attributed to periods of high rainfall (volume and intensity) on vulnerable soils that are often pugged and/or compacted by livestock, restricting water infiltration. Curran-Cournane et al. (2011) showed how reduced soil infiltration rates as a result of treading damage increased sediment, phosphorus and *Escherichia coli* (*E. coli*) in surface runoff that occurred after grazing. Monaghan et al. (2017) found that most (about 90 percent) surface runoff occurred after grazing due to increased soil damage. Stock type and grazing duration are two significant drivers of soil damage. When stocked at the same liveweight per hectare, the severity of hoof damage to the soil surface will be greater under cattle than under sheep (Houlbrooke and Laurenson, 2013). McDowell and Houlbrooke (2009) observed greater soil damage and subsequent sediment and phosphorus losses from cattle-grazed winter forage crops than from sheep-grazed winter forage crops.

The effectiveness of some management options that can help reduce the risk of soil damage or runoff and contaminant losses from grazed winter forage cropland have been documented in a limited number of studies. Drewry and Paton (2005) showed how back fencing reduced animal pacing and therefore minimised soil damage through unnecessary stock movement in winter forage cropland grazed by dairy cattle. Similarly, McDowell and Houlbrooke (2009) found that limiting cattle grazing times led to significant reductions in phosphorus and sediment losses in surface runoff from winter forage cropland. Studies have found a relatively small fraction of a farm or catchment can generate a large amount of runoff and contaminant loads (Gbruek et al., 2000; Pionke et al., 2000). Field observations have established that the majority of surface runoff during and shortly after rainfall (termed as stormflow) is generated by a small proportion of the catchment area (often <10%; Freeze, 1974). These areas are called critical source areas (CSAs) of runoff (Srinivasan and McDowell, 2009). CSAs are often persistently wet and include gullies, swales and seeps. Once CSAs are identified, management strategies can be more efficiently targeted to CSAs. Even small changes in management of these areas, especially in winter, can significantly lessen grazing impacts on soil and water quality. In a study in South Otago, Monaghan et al. (2017) showed that by grazing the at-risk CSA late in the winter season, sediment and nutrient loss were reduced significantly. As part of the study, the critical source area was the last break that was grazed at the end of the winter. This strategic grazing approach located most of the soil damage away from areas where water accumulated, hence maintaining the gully's soil infiltration rates. As a result, surface runoff from these protected catchments was

60 to 80 percent less than in un-protected catchments. Compared with standard grazing, strategic grazing reduced sediment loads in surface runoff by 94% and total phosphorus by 84%. This study used dairy cows; given that other stock classes may have different impacts, more research is needed to describe the impacts of other stock types, soil and climates.

This project aims to address the information gap around sheep winter grazing and determine the effectiveness and relevance of strategic grazing management practises for minimising sediment and nutrient loads in surface runoff from winter forage crops grazed by sheep. Using a paired catchment study approach, we investigated the effectiveness of protection of CSAs (animal exclusion during winter) in each catchment. This management strategy aimed to reducing soil damage in the CSAs of each catchment and, therefore, the volumes of surface runoff generated and associated fluxes of sediments and nutrients. Our specific objectives were to: (1) quantify sediment, P, N and *E. coli* losses in surface runoff from winter forage crops grazed by sheep over three successive winters, and (2) evaluate the effectiveness of CSA protection for reducing these losses.

3. Method

3.1 Study site and treatments

The study was conducted in west Otago (45^{o} 57’ 19”S, 169^{o} 46’ 15”), on rolling downland which has been used for pastoral grazing by sheep. The climate in the area is temperate. Mean annual rainfall in the area, as measured at 260 m asl elevation for the period between 1973-2018, was 850 mm. A paired catchment study approach was undertaken to evaluate the effectiveness of critical source area (CSA) protection for reducing contaminant losses from fields used for winter forage crop grazing by sheep. Two catchments, 0.81 ha (Catchment A) and 1.17 ha (Catchment B), at comparable elevations were selected for growing a winter forage crop for three consecutive years, before both fields were returned to pasture. Catchment B had slightly steeper slopes (6-30^{o}) compared to Catchment A (6-26^{o}). Both catchments are broadly comparable in terms of topography (Figure 1) and are ca. 1000 m apart. The predominant soil type in both catchments is a Timaru silt loam, which is classified as a Mottled Fragic Pallic soil (Hewitt, 2010).

A summary of crop sequence and yield, stock type, stock densities and feed requirements and grazing managements implemented are presented in Table 1. Practical management limitations, such as cultivation and avoiding the build-up of weed, insect pest or disease burdens, required a crop rotation of swedes followed by kale.

Table 1. Details of the crop sequence and yield, stock type, stock densities and feed requirements and grazing managements implemented in the paired catchments.

		Year 1	Year 2	Year 3
Catchment A	Crop type	Swedes	Kale	Kale
	Crop yield (T DM ha^{-1})	10.7	7.2	6
	Stock type	Sheep (ewe hoggets)	Sheep (ewe)	Sheep (ewe)
	Grazing intensity (RSU m^{-2} day^{-1})	0.50	0.74	0.39
	Liveweight (kg per animal)	43	65	59
	Target crop feed intake (kg DM $sheep^{-1}$ day^{-1})	2	1.5	1.25
	Supplementary feed (kg DM $sheep^{-1}$ day^{-1})	0.25	0.25	0.25
	Grazing method	Standard	Strategic	Standard
Catchment B	Crop type	Swedes	Kale	Kale
	Crop yield (T DM ha^{-1})	9.7	9.3	4
	Stock type	Sheep (ewe hoggets)	Sheep (ewe)	Sheep (ewe)
	Live weight (kg per animal)	43	65	59
	Grazing intensity (RSU m^{-2} day^{-1})	0.37	0.63	0.27
	Target crop feed intake (kg DM $sheep^{-1}$ day^{-1})	2	1.5	1.25
	Supplementary feed (kg DM $sheep^{-1}$ day^{-1})	0.25	0.25	0.25
	Grazing method	Standard	Standard	Strategic

Notes: Standard grazing had no protection of the CSA. Strategic grazing had protection of the CSA. Both treatments had a grazing front that moved in a downslope direction toward the catchment outlet (collection point, Figure 1). Livestock densities are converted to units of Relative Stock Units (RSUs) to derive a grazing intensity metric that can be used across different livestock classes. One RSU is assumed to be equivalent to an animal that requires 6000 MJME (megajoules of metabolizable energy) per annum.

In Year 1 of the study (2020), both catchments were managed in the same way (i.e., standard grazing practices) to quantify baseline losses and flows to examine whether they had comparable rainfall-runoff responses. In Year 2, after cultivation and sowing in kale, a strategic grazing method was implemented in Catchment A (described below), whilst standard grazing practice (i.e. as per Year 1) was followed in Catchment B. These grazing methods were switched between the catchments in Year 3 to quantify if a similar treatment response to that observed in Year 2 occurred.

The swede crop planted in Year 1 was established following spraying of the old pasture with Glyphosate and direct drilling. Kale crops in the subsequent years were established following spraying, surface cultivation and sowing with an air seeder. Crop establishment occurred in late spring of each year. Crop growth provided yields of 4 – 11 TDM ha^{-1} by the time of grazing during the following winter (Table 1). Crop yield was assessed by field survey using a 1 m^2 quadrat prior to grazing. Nine sampling locations corresponding to altitude and aspect within each catchment were selected by randomly throwing the quadrat across the crop at that location. Crop samples were taken from each quadrat by cutting the plant material to ground level and recording the total wet weight sampled. The crop samples were sub-sampled by a quartering technique with the sub-samples divided into leaf and bulb (swedes) or leaf and stalk (kale). The wet weights were recorded before drying the sub-samples at 65ºC to a constant weight for dry matter determination. The dry weights were then recorded, and the dry matter content (DM) calculated as a percentage of the wet weight of the sub-sample. Fields remained bare following grazing of the winter forage crops until soil conditions allowed cultivation and replanting in the following spring.

The strategic grazing approach had the aim of protecting CSA within each catchment. The approach to CSA protection in this study involved establishing and maintaining a pasture (Perennial ryegrass, Italian ryegrass & white clover) cover in CSAs ahead of the winter grazing period. Temporary electric fencing was used to exclude livestock from these areas throughout winter and early spring. The CSA was grazed with ewes (40 ewes ha^{-1}) prior to cultivation in the late spring, however the pasture by this time of year had become rank and less palatable to stock so grazing to ground level did not occur. The CSA was sprayed with Glyphosate followed by light surface cultivation prior to sowing in kale (Catchment A) or pasture (Catchment B) again in the third year.

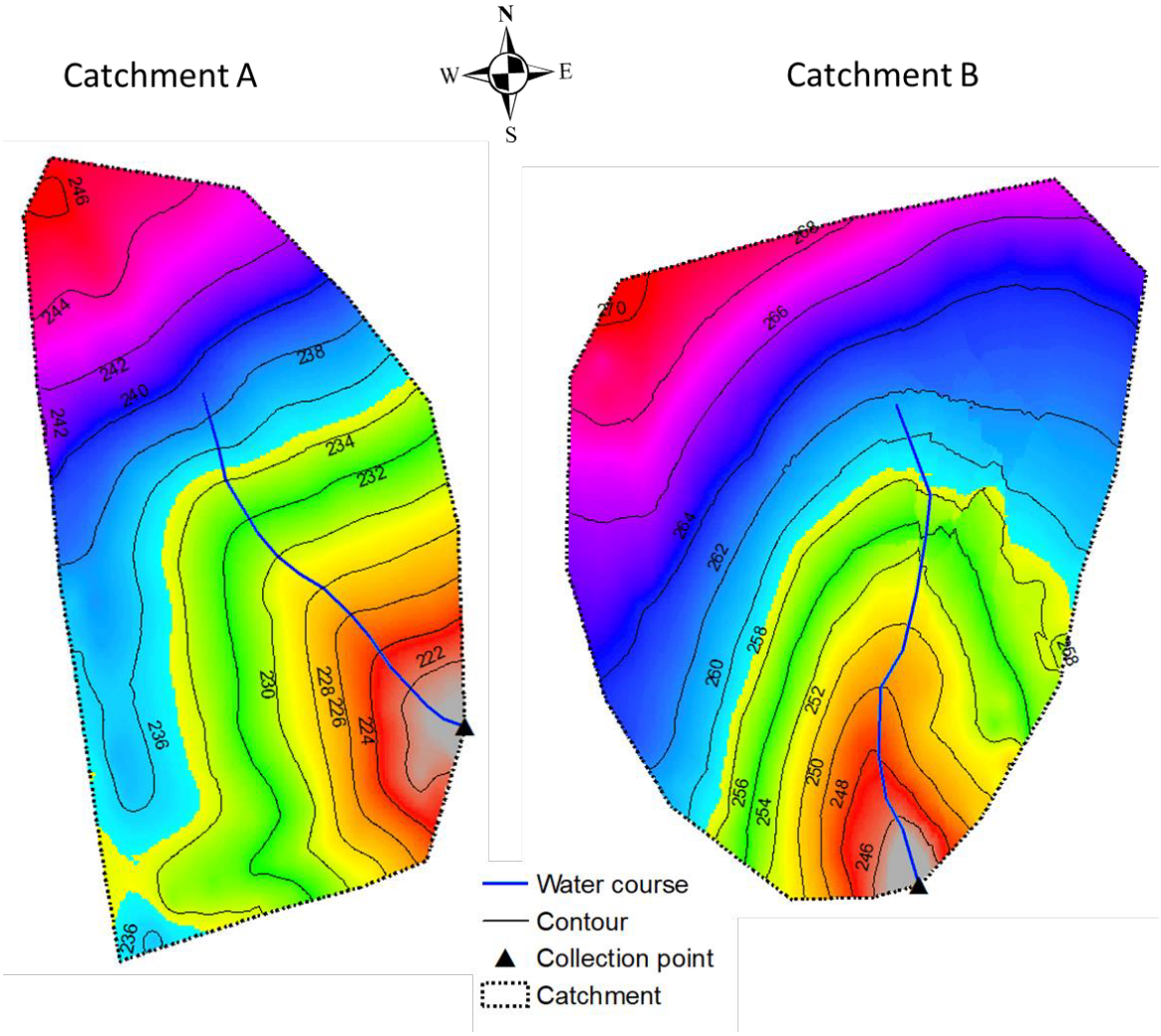

Figure 1. Elevation maps (contours every 2 m) and ephemeral water courses of paired Catchments A and B.

3.2 CSA delineation

Approaches for identifying CSAs range from simple inventories to complex models. In this study, topographic wetness index (TWI) in combination with slope steepness was used to delineate the CSAs in both catchments. TWI is an estimate of the potential for soil saturation – and therefore generation of surface runoff – of a land area that is based on slope and drainage area (Beven and Kirby, 1979). TWI was calculated by constructing a 1 x 1 m DEM for each catchment using ArcGIS, then using several tools within the programme to calculate flow direction, flow accumulation, slope and ultimately TWI using a simple equation (Beven and Kirby, 1979). The resulting TWI spatial data layer depicts areas where water is likely to accumulate (Figure 2). Smaller TWI values indicate less potential for overland flow generation, while larger values reflect a greater risk due to the drainage from a larger upslope area. In addition to TWI, slope criteria were also used to delineate high risk areas for soil mobilisation based on slope steepness (>25°) criteria. Finally, TWI maps and slope maps were superimposed to demarcate the extent of CSAs in

both catchments. This method of delineation identified approximately 35% of total catchment areas as CSAs.

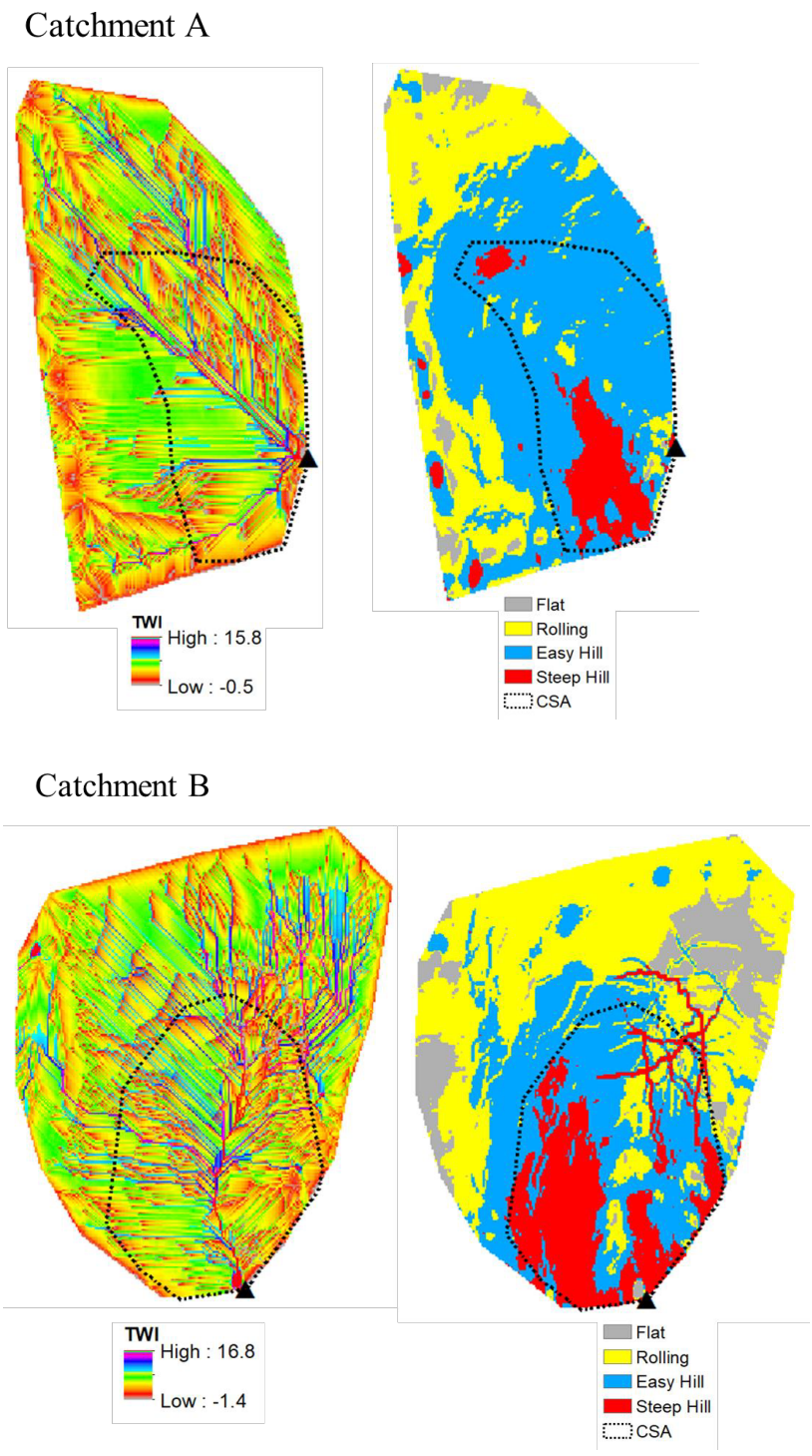

Figure 2. Maps of the TWI (left panel) and slope (right panel) categories for each catchment. Broken lines demarcate CSA boundaries.

3.3 Soil physical measurements

Soil sampling was carried out annually in both catchments before (c. April – May) and after (c. July – August) grazing of the winter forage crop. Three transects (1 upper, 1 mid, and 1 lower slope location) were established within each catchment. Soil core samples (50 x 100 mm dia.) were collected from soil profile depths of 0 – 50, and 50 – 100 mm at each location and four replicates were taken at each sampling depth. Soil samples were analysed for pore-size distribution (m^3 m^{-3}) and soil bulk density (gm cm^{-3}) using the method described by Drewry et al. (2000). Soil macroporosity (air-filled porosity at -10 kPa soil water pressure) was determined as the volumetric difference between total porosity and water content at -10 kPa. Total porosity was calculated from measurements of dry bulk density assuming a particle density of 2.65 gm cm^{-3} (Creswell and Hamilton, 2002). Field-saturated soil hydraulic conductivity (K_{sat}) of the topsoil (0 – 100 mm) was measured, before and after grazing, at six random locations in each catchment in the final year of the experiment using a single ring infiltrometer following the method outlined in Verbist et al. (2010).

3.4 Rainfall and runoff measurement systems

Incident rainfall (mm) at each site was recorded using a tipping bucket rain gauge (Davis Instruments USA; 0.2 mm per tip) backed by a manual gauge (100 mm orifice diameter). The orifices of all rain gauges were positioned 30 cm above the ground to avoid ground splash effects.

Surface runoff was continuously measured at the outlet of each catchment throughout the three-year study period. An H-flume was installed at each site for stage height recording. The stage-height data were converted to runoff rates using a rating curve for the H-flume. Attached to each flume was a stilling well with a water level sensor (NIWA Hydrologger) and ISCO auto-sampler. The system was controlled by a Unidata Neon Module, with stage height measured at one-minute intervals. The automated sampling system was connected to the water level sensor and programmed to automatically start collecting samples at 30-minute intervals when flow rates start to increase at the beginning of a runoff event. All data from the monitoring system was uploaded to the internet every 15 minutes to enable the system to be monitored remotely. When the automatic samplers were triggered, the samples were collected within 24 hours of the beginning of a runoff event and returned to the lab at AgResearch Invermay for analysis. Autosamplers were reset and sampling continued if a storm event lasted more than 24 hours. Due to sensor failure, robust stage-height data are only available for two periods in Catchment B: (i) 01 October 2020 to 28 August 2021; and (ii) 25 February 2022 to 31 October 2022.

3.5 Laboratory measurements

Collected water samples were returned to the lab and immediately analysed for *E. coli* using the Colilert / Quanit-Tray method (IDEXX, Maine, USA). Sub-samples were frozen, and then sent to the AgResearch Environmental Science laboratory at Ruakura for analysis of suspended solids (SS), total phosphorus (TP), total filterable phosphorus (TFP), filterable reactive phosphorus (FRP) and particulate phosphorus (PP) concentrations. A frozen sub-

sample was also sent to the Lincoln University Analytical Lab for total nitrogen (TN), total Kjeldahl nitrogen (TKN), ammonium-N (NH_4-N) and nitrate-N (NO_3-N) analysis. SS was determined by weighing the residue left after oven drying (105^{o}C for 24 h) a known volume of sample (APHA, 1998). TP and TFP concentrations were determined colorimetrically using a SEAL auto Analyser 3 using the phospho-molybdenum complex at 880 nm (Murphy and Riley, 1962) following a persulphate digestion for TP and a UV digestion for TFP. Particulate P was calculated as the difference between TP and TFP. Nitrate and ammonium analyses were undertaken colorimetrically using a Skalar SAN++ segmented flow analyser (Mulvaney, 1996). TKN was determined following acid digestion and distillation of samples (APHA, 1998) and TN was calculated as the sum of TKN and nitrate N.

3.6 Statistical analysis

Statistical analysis of soil physical data was carried out using the statistical package 'R' version 4.1.0 (R Core Team, 2021). Data were analysed using analysis of variance (ANOVA) and a nested analysis to acknowledge the structure in the data. Differences in soil physical properties were assumed to be significant with $p<0.05$.

4. Results

4.1 Soil physical measurements

Table 2 summarises soil physical properties for all three years of the forage cropping sequence. Grazing during winter significantly ($p<0.05$) decreased total soil porosity and macroporosity in the 0 – 100 mm soil depth relative to the pre-grazing soil condition. Such differences in soil porosity and macroporosity are likely to contribute to the observed differences in K_{sat} between pre- and post-grazing. Indeed, the pre-grazing K_{sat} values measured in both catchments in Year 3 of the study were significantly greater than that observed for post-grazing samplings (Table 2). Slope location did not contribute to the observed differences in total porosity and macroporosity in both catchments. There was no discernible significant downward trend in total porosity or macroporosity that would indicate an ongoing decline in soil aggregation due to winter grazing.

Table 2. Total soil porosity, macroporosity and soil saturated hydraulic conductivity (K_{sat}) values measured in Catchments A and B on pre- and post-winter crop grazing sampling occasions.

Depth (mm)	Pre or post grazing	Catchment A			Catchment B		
		2020	2021*	2022	2020	2021	2022*
Total porosity (m^3 m^{-3})							
0-50	pre	0.655[a]	0.669[a]	0.59[a]	0.658[a]	0.674[a]	0.587[a]
	post	0.636[a]	0.615[b]	0.568[b]	0.622[b]	0.614[b]	0.603[a]
50-100	pre	0.576[a]	0.620[a]	0.537[a]	0.594[a]	0.619[a]	0.523[a]
	post	0.606[b]	0.587[b]	0.548[a]	0.596[a]	0.594[b]	0.571[b]
Macroporosity (m^3 m^{-3})							
0-50	pre	0.269[a]	0.251[a]	0.28[a]	0.228[a]	0.25[a]	0.296[a]
	post	0.117[b]	0.125[b]	0.106[b]	0.109[b]	0.09[b]	0.139[b]
50-100	pre	0.218[a]	0.195[a]	0.21[a]	0.175[a]	0.182[a]	0.195[a]
	post	0.121[b]	0.153[b]	0.127[b]	0.116[b]	0.153[a]	0.153[b]
K_{sat} (mm h^{-1})	pre	nd	nd	200[a]	nd	nd	185[a]
0-100	post	nd	nd	22[b]	nd	nd	36[b]

Notes: Each value represents average of three transects and four replicates per transact. Years in which CSA was protected (Strategic grazing) are indicated by *. Within each catchment, annual differences between pre- and post-grazing samples for individual soil depths is determined to be significant (p<0.05) if the superscripts differ within a column. nd = not determined.

4.2 Rainfall and surface runoff

Total winter (June – August) rainfalls in Catchment A for each of the three years of the study were 129, 185 and 159 mm in 2020, 2021, and 2022, respectively. Corresponding values for Catchment B were 134, 159 and 134 mm (Table 3). These winter rainfall totals were below the long-term mean of 215 mm recorded close to the present study area at an elevation of 261 m asl. The presently observed differences in winter rainfall totals between the two sites that are located ~ 1000 m is not surprising given the complex topography of the study area which may be expected to induce local rain shadows (leeward slopes) and precipitation hotspots (windward slope). The highest 15 min-rainfall intensity recorded at Catchment A during the study period was 13.6 mm h^{-1} vs. 12.8 mm h^{-1} at Catchment B. As illustrated by Figure 4, low rainfall intensities appeared to be a common climate feature for the three years of this field study. Rainfall intensities ≤ 3 mm h^{-1} occurred for ~ 90% of the events in both catchments. Such low rainfall intensities combined with relatively high post-grazing K_{sat} values are likely to contribute to the low amounts of surface runoff volumes recorded at both sites (Table 3).

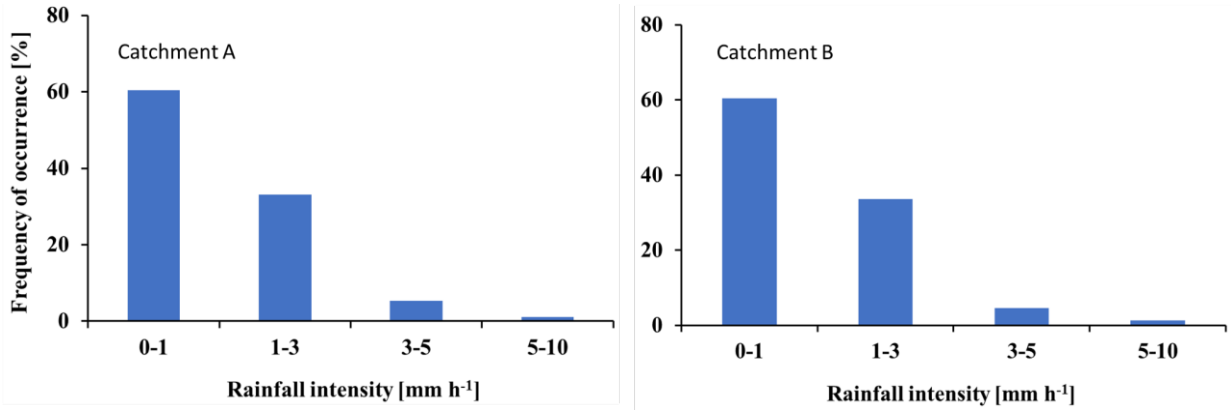

Figure 4. Frequency distribution of winter rainfall intensity at the study sites.

A total of 19 surface runoff events were recorded in Catchment A during the study period. Approximately 72% of total runoff events occurred in winter (June – July), with the remaining occurring in September (27%) and January (1%). Moreover, approximately 70% of the surface runoff events occurring in winter were recorded during the month of July. The total amount of surface runoff from Catchment A was 45 mm for during the study period. The majority of this runoff (85%) was recorded during winter and early spring (Table 3). Unfortunately, due to water level sensor failure in Catchment B on multiple occasions, surface runoff data were not available for the full winter cropping period (2020 – 2022). For missing surface runoff measurements in early spring of 2021, volumes were estimated using runoff coefficients derived for other events in 2021 where both rainfall and runoff data were available for Catchment B. It is acknowledged that such an approach provides only a rough estimate of runoff volume and is heavily biased towards rainfall total, and as such, it does not take into account various other important factors that can significantly affect surface runoff generation (e.g., treading damage to soil, rainfall intensity etc.). In order to partly address this uncertainty, a range in surface runoff volumes based on two contrasting

values (lowest and highest) of runoff coefficients is presented for winter in 2021 (Table 3). Due to the complete absence of runoff data (both event timings and volumes), it was, however, not possible to fill missing surface runoff data for the winter and early spring of 2020. A total of 13 surface runoff events were recorded in Catchment B during the 2021 and 2022 cropping sequence. The majority of surface runoff events (~70%) in Catchment B were also recorded in winter, with the remaining in September (23%) and January (7%). The estimated total amount of surface runoff from Catchment B in 2021 and 2022 ranged from 29 – 49 mm, again with the majority (~95%) recorded during winter and early spring (Table 3). There was a similar timing and number of runoff events observed in both catchments for 2021 and 2022, which suggests that their hydrological responses behaved in a broadly similar manner.

Most surface runoff events during the three-year cropping cycle occurred after the catchments had been grazed and accounted for 75% (Catchment A) to ~88% (Catchment B) of the total surface runoff. For the two measurement periods when the grazing treatments were imposed, cumulative surface runoff from the standard grazing treatment was in the range of 24 – 44 mm. The corresponding value for the strategic grazing treatment (i.e protection of CSA throughout winter and early spring) was 32 mm.

Table 3. Amounts of rainfall and surface runoff in Catchments A and B in winter (June-August) and early spring (September). Values in parentheses are total rainfall received in winter (June – August).

	Catchment A			Catchment B		
	2020	2021*	2022	2020	2021	2022*
Total rainfall [mm]	212 (129)	271 (185)	205 (159)	203 (134)	233 (159)	170 (134)
Number of surface runoff events	5	9	4	n.a.	10	2
Total surface runoff [mm]	6	28	4	n.a.	20 – 40	4

Notes: Years in which CSA was protected (Strategic grazing) are indicated by *. n.a. indicates data not available due to instrument malfunctioning.

4.3 Concentrations and fluxes of contaminants in surface runoff

A total of 312 water samples were collected from the 32 combined surface runoff events while the catchments were used for winter cropping from 2020 to 2022. The concentrations of SS, TP, TN and *E. coli* for the two treatments are shown in Figure 5. As expected, analyte concentrations varied widely across events and catchment treatments which can largely be attributed to plant cover type, grazing effects and individual surface runoff event

characteristics. The median concentration values of these analytes were higher for the standard grazing than for the strategic grazing treatment when the grazing treatments were imposed. These results suggest CSAs are effective at reducing sediment, nutrient and *E. coli* loads in surface runoff. Moreover, analytes concentrations tended to be greater for surface runoff events that occurred after grazing of the forage crop (data not shown).

Calculated total yields of contaminants transported in surface runoff for winter and early spring are documented in Table 4. Fluxes of all contaminants were considerably reduced when CSA protection was implemented. Reductions in SS, TP, TN and *E. coli* yields were 55%, 48%, 38% and 63%, respectively (Table 5). Given that surface runoff volumes from the standard and strategic grazing treatments were similar (Table 5), these reductions can largely be attributed to the lower analyte concentrations when the CSA was protected (Figure 5).

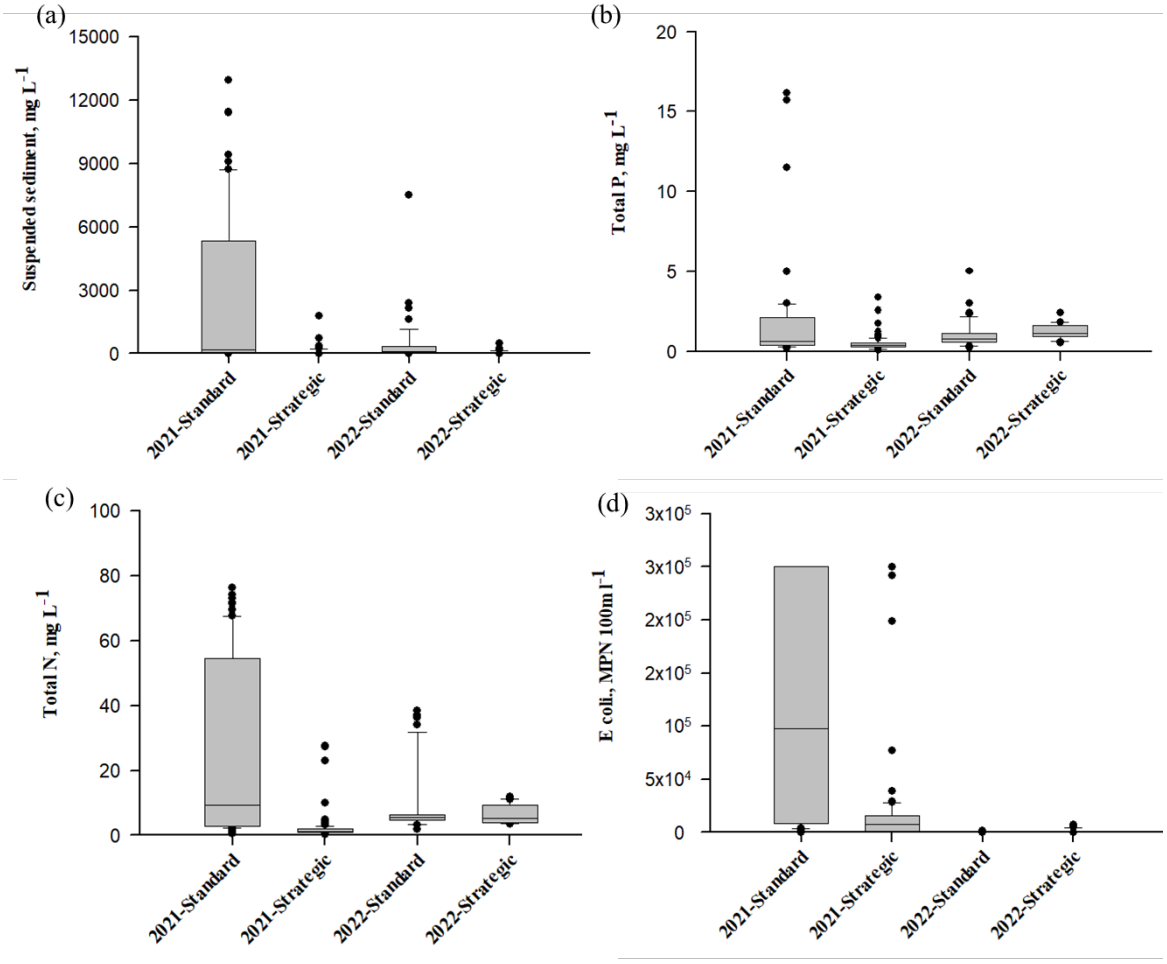

Figure 5. Concentration distributions of (a) suspended sediment, (b) total phosphorus, (c) total nitrogen and (d) *E. coli* concentrations in runoff samples collected over the three-year monitoring period. The grazing treatments shown are standard or strategic for 2021 and 2022. Boxes depict the 25th to 75th percentile values, with horizontal lines plotted within boxes representing median values; whiskers show the 10th and 90th percentile values and points the 5th and 95th percentile values.

Table 4. Total fluxes of suspended sediment (SS), total phosphorus (TP), total nitrogen (TN) and *E. coli* in winter (June – August) and early spring (September) runoff events.

Year	Catchment A					Catchment B				
	Treatment	SS (kg ha^{-1})	TP (kg ha^{-1})	TN (kg ha^{-1})	*E. coli* (MPN ha^{-1})	Treatment	SS (kg ha^{-1})	TP (kg ha^{-1})	TN (kg ha^{-1})	*E. coli* (MPN ha^{-1})
2020	Standard	3	0.030	0.11	0.14 x 10^{10}	Standard	n.a.	n.a.	n.a.	n.a.
2021	Strategic	43	0.2	0.45	12 x 10^{10}	Standard	80 – 100	0.3 – 0.5	0.7 – 1.5	(32 – 40) x 10^{10}
2022	Standard	11.5	0.037	0.18	4.26 x 10^{10}	Strategic	3	0.03	0.34	2.8x10^{10}

Notes: n.a. indicates data not available due to instrument malfunctioning. Fluxes of analytes from Catchment B in 2021 are presented in a range to indicate the uncertainty associated with some of the runoff volumes derived using a runoff coefficient method (see section 4.2).

Table 5. Mean annual fluxes of suspended sediment (SS), total phosphorus (TP), total nitrogen (TN) and *E. coli* under standard and strategic grazing (CSA protection) managements for winter (June – August) and early spring (September) runoff events recorded during 2021 and 2022.

Analyte	Treatment	Total flux (kg or MPN ha^{-1})	Runoff (mm)
SS	Standard	51	17
	Strategic	23	16
TP	Standard	0.22	
	Strategic	0.115	
TN	Standard	0.64	
	Strategic	0.40	
E. coli	Standard	20×10^{10}	
	Strategic	7.4×10^{10}	

5. Discussion

The presented results provide the first quantitative information on the effectiveness of strategic grazing to protect CSAs for minimising sediment, nutrient and *E. coli* loads in surface runoff from fields used for winter forage crop grazing by sheep.

Soil physical measurements showed significant differences in soil conditions between pre- and post-grazing samplings of the winter crops (Table 2). A large and significant reduction in macroporosity as well as high degree of visual soil deformation suggested that both compaction and pugging occurred simultaneously across both catchments following grazing of the crop (Drewry and Paton, 2005). This finding was further supported by the surface (0 – 100 mm) K_{sat} measurements that were made in 2022; major decreases in K_{sat} were observed in both catchments following grazing of the crop (Table 2). These findings thus support the strong linkage between soil physical condition and contaminant concentrations documented in other studies (McDowell et al., 2003; McDowell and Houlbrooke, 2009; Curran-Cournane et al., 2011; Monaghan et al., 2017) where increased concentrations or loads of TP or SS in surface runoff were reported in response to decreases in soil macroporosity and K_{sat}. The large reduction in surface K_{sat} during grazing is primarily related to the destruction or clogging of large drainage pores. Prolonged exposure of topsoil to the elements after grazing and trampling can also strongly diminish soil biological activity and soil aggregate strength (Collof et al., 2010). These observations of soil damage in response to winter forage crop grazing are comparable to findings reported by other studies (McDowell et al., 2003; Drewry and Paton, 2005). Monaghan et al. (2017) also reported significant increases in surface runoff after grazing of winter crops by dairy cows that was attributed to increased soil damage. McDowell et al. (2003) reported

that an increase in treading decreased K_{sat}, producing greater volumes of surface runoff. These results are not surprising since a decrease in soil infiltration rate would tend to produce a greater volume of infiltration-excess surface runoff. At only 22 – 36 mm h^{-1}, the surface K_{sat} measured following grazing in our study was still sufficient to accommodate high rainfall intensities, suggesting that winter crop grazing by sheep did not incur as much soil damage, and consequent surface runoff, as reported in the other studies on cattle and deer mentioned above. Although we did not measure soil moisture contents at our sites, these findings indicate that the majority of the winter runoff recorded in our study occurred under saturation- not infiltration-excess conditions (Curran-Cournane et al., 2011). These findings reinforce the use of mitigation strategies such as restricted or nil grazing in winter in areas that are persistently wet (e.g. CSAs) as a way to minimise contaminant losses in surface runoff.

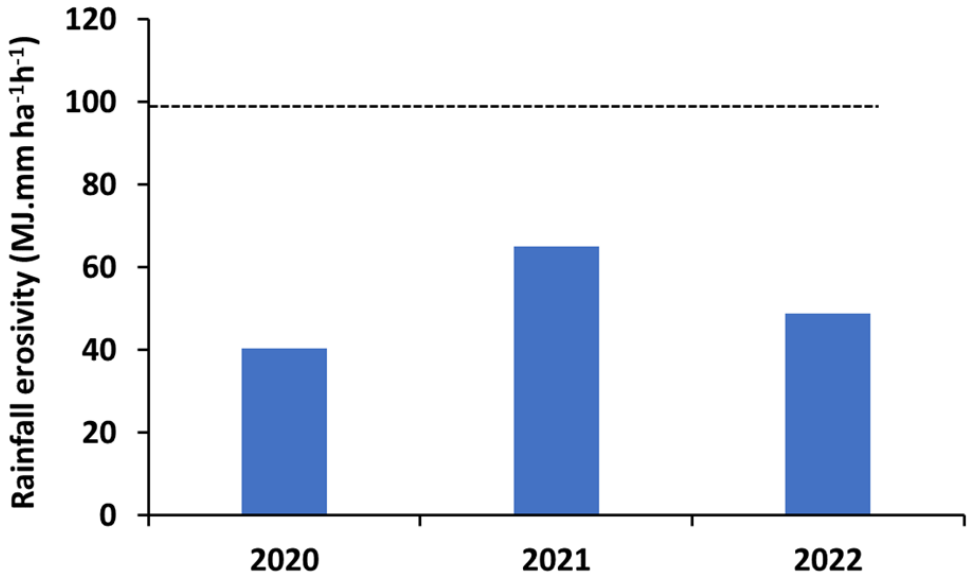

Figure 6. Winter rainfall erosivity values in the study area during the grazed winter forage cropping phase (2020 – 2022). The black dotted horizontal line indicates the long-term average rainfall erosivity in the area. Rainfall erosivity was calculated using the formula presented in Klik et al. (2015).

As evident from the data presented in Tables 4 and 5 and Figure 5, protecting CSAs from winter grazing considerably reduced the concentrations and loads of contaminants lost in surface runoff. These reductions can largely be attributed to: (i) the protection of soil structure in the CSA through animal exclusion, and (ii) vegetative cover in the CSA acting as a filter to both reduce rainfall splash energy and trap sediment that may have been dislodged from grazed areas. Contaminant losses, particularly the estimated sediment loss for the standard grazing treatment (51 kg ha^{-1}), were much less than losses reported in other studies (e.g. McDowell and Houlbrooke, 2009). This could be explained by the relatively low rainfall erosivity at our sites and the relatively low grazing pressures deployed when the crops were grazed (Table 1). Together with wind, rainfall is the most active agent of soil erosion due to its potential to break down aggregates, detach soil particles and produce runoff (Oliveira et al., 2013). The potential of rain to erode soil is characterised as rainfall erosivity which can be evaluated using the ability of a given raindrop to detach soil particles and produce runoff flow (Angulo-Martinez et al., 2012). Rainfall erosivity during the current study period was (much) lower as compared to the estimated long-term average (Figure 6). Grazing and treading pressures on soils were also low which allowed most of the rainfall to infiltrate (Table 1). Typical equivalent grazing pressures for winter crops grazed by cattle are in the range of 0.8 – 1.1 RSU m^{-2} day^{-1} (C. Smith, personal communication). As such, the relatively low losses of sediment and other contaminants observed in this study are not surprising.

6. Conclusions

A key message identified in this study is that protection of critical source areas (CSAs) from winter grazing can reduce fluxes of water contaminants. Soil physical data indicated significant differences in soil conditions between pre- and post-grazing samplings of the winter crop. However, this did not appear to translate into considerable differences in surface runoff volumes between the treatments. Much of the treatment response observed in contaminant fluxes could be attributed to the reduced contaminant concentrations observed in the strategic grazing treatment. This, in turn, can be attributed to: (i) the protection of soil structure in CSAs, and (ii) the preservation of a protective vegetative covering in the CSA that acted as a filter to trap sediment from grazed areas and slow surface runoff before it reached the catchment outlets. Combined with the relatively light grazing and treading pressures deployed in the catchments, most of the rainfall was able to infiltrate instead of flowing across the soil surface and entraining water contaminants such as sediment, phosphorus and faecal bacteria. We caution that the relatively low rates of contaminant loss in surface runoff may reflect the relatively benign weather conditions and grazing pressures recorded at the site from 2020 to 2022; further investigation is warranted to set our findings in the context of a wider range of weather conditions and grazing pressures.

References

APHA, 1998. Standard methods for the examination of water and wastewater. 20th ed. Washington (DC): American Public Health Association.

Angulo-Martínez, M., Beguería, S., Navas, A., Machín, J., 2012. Splash erosion under natural rainfall on three soil types in NE Spain. Geomorphology 175–176, 38–44.

Beven, K, Kirkby, M.J., 1979. A physically based, variable contributing area model of basin hydrology. Hydrological Science Bulletin 24, 43–69.

Burkitt, L.L., Winters, J.L., Horne, D.J., 2017. Sediment and nutrient losses under winter cropping on two Manawatū hill country soils. Proceedings of the New Zealand Grassland Association 79, 27–34.

Cresswell, H.P., Hamilton, G.J., 2002. Bulk density and pore space relations. In: McKenzie N, Coughlan K, Cresswell H, editors. Soil physical measurements and interpretation for land evaluation. Collingwood (VIC): CSIRO Publishing; p. 35–58.

Colloff, M. J., Pullen, K.R., Cunningham, S.A., (2010). Restoration of an ecosystem function to revegetation communities: The role of invertebrate macropores in enhancing soil water infiltration. Restoration Ecology 18(1),65–72.

Curran-Cournane, F., McDowell, R.W., Littlejohn, R., Condron, L.M., 2011. Effects of cattle, sheep and deer grazing on soil physical quality and losses of phosphorus and suspended sediment losses in surface runoff. Agriculture, Ecosystems & Environment 140, 264–272.

Drewry, J.J., Littlejohn, R.P., Paton, R.J., 2000. A survey of soil physical properties on sheep and dairy farms in southern New Zealand. New Zealand Journal of Agricultural Research 43, 251–258.

Drewry, J.J., Paton, R.J., 2005. Soil physical quality under cattle grazing of a winter-fed brassica crop. Australian Journal of Soil Research 43, 525–531.

Freeze, A,R., 1974. Streamflow generation. Reviews of Geophysics and Space Physics 12: 627–647.

Gburek, W.J., Sharpley, A.N., Heathwaite, L., Folmar, G.J., 2000. Phosphorus management at the watershed scale: A modification of the phosphorus index. Journal of Environmental Quality 29, 130–144.

Hewitt, A.E., 2010. New Zealand soil classification (revised). Lincoln: Manaaki Whenua Press.

Houlbrooke, D.J., Paton, R.J., Morton, J.D., Littlejohn, R.P., 2009. Soil quality and plant yield under dryland and irrigated winter forage crops grazed by sheep or cattle. Australian Journal of Soil Research 47, 470–477.

Houlbrooke, D.J., Laurenson., S., 2013. Effect of sheep and cattle treading damage on soil microporosity and soil water holding capacity. Agricultural Water Management 121, 81–84

Kilk, A., Haas, K., Dvorackova, A., Fuller, I.C., 2015. Spatial and temporal distribution of rainfall erosivity in New Zealand. Soil Research 53, 815–825.

McDowell, R.W., Drewry, J.J., Muirhead, R.W., Paton, R.J., 2003. Cattle treading and phosphorus and sediment loss in overland flow from grazed cropland. Australian Journal of Soil Research 41, 1521–1532.

McDowell, R.W., 2006. Phosphorus and sediment loss in a catchment with winter forage grazing of cropland by dairy cattle. Journal of Environmental Quality 35(2), 575–583.

McDowell, R.W., Houlbrooke, D., 2009. Management options to decrease phosphorus and sediment losses from irrigated cropland grazed by cattle and sheep. Soil Use and Management 25(3), 224–233.

Monaghan, R.M., Laurenson, S., Dalley, D.E., Orchiston, T.S., 2017. Grazing strategies for reducing contaminant losses to water from forage crop field grazed by cattle during winter. New Zealand Journal of Agricultural Research 60, 333–348.

Mulvaney, R.L., 1996. Nitrogen-inorganic forms. In: Sparks DL, editor. Methods of soil analysis: Part 3. SSSA Book Ser. 5. Madison, WI: SSSA; p. 1123–1184.

Murphy, J., Riley, H.P., 1962. A modified single solution method for the determination of phosphate in natural waters. Analytica Chimica Acta 27, 31–36.

Oliveira, P.T.S., Wendland, E., Nearing, M.A., 2013. Rainfall erosivity in Brazil: a review. Catena 100, 139–147.

Pionke, H.B., Gburek, W.J., Sharpley, A.N., 2000. Critical source area controls on water quality in an agricultural watershed located in the Chesapeake. Ecological Engineering 14(4), 325–335.

R Development Core Team, 2021. R: A Language and Environment for Statistical Computing, R Foundation for Statistical Computing, Vienna, Austria.

Srinivasan, M.S., McDowell, R.W., 2009. Identifying critical source areas for water quality: 1. Mapping and validating transport areas in three headwater catchments in Otago. Journal of Hydrology 379, 54–67.

Verbist, K., Torfs, S., Cornelis, W.M., Oyarzún, R., Soto, G., Gabriels, D., 2010. Comparison of single- and double-ring infiltrometer methods on stony soils. Vadose Zone Journal 9 (2), 462–475.

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