Research Note • DOI: 10.15212/ijafr-2020-0117 IJAFR • 59(1) • 2020 • 201-205

Irish Journal of Agricultural and Food Research

Research note: Mitigation of phosphorus, sediment and *Escherichia coli* losses in runoff from a dairy farm roadway

R.W. McDowell[1,2], K. Daly[3], O. Fenton[3†]

[1]AgResearch, Lincoln Science Centre, Private Bag 4749, Christchurch 8140, New Zealand
[2]Faculty of Agriculture and Life Sciences, P O Box 84, Lincoln University, Lincoln 7647, Christchurch, New Zealand
[3]Teagasc, Environment Research Centre, Co. Wexford, Ireland

Abstract

Dairy cow deposits on farm roadways are a potential source of contaminants entering streams. Phosphorus (P), suspended sediment (SS) and Escherichia coli **(E. coli)** ***loads in 18 runoff events over 12 mo from two-halves of a section of dairy farm roadway that spilt into an adjacent P-impacted stream were measured. The runoff from one half was untreated while the other half was directed through a filter of steel melter slag [termed aluminium chlorohydrate (ACH)-altered slag] sprayed with 1% ACH solution to improve P sorption capacity. An uncertainty analysis was conducted to ascertain potential loads of P lost from roadways considering variation in deposit weight, number and P content. Over the monitoring period, the total load decreased P (92%), SS (98%) and*** **E. coli** ***(76%) from the ACH-altered slag roadway compared to the control. However, uncertainty analysis showed that the amount of dung-P deposited on the roadway could be 10-fold greater.***

Keywords

Contaminants • dung • laneway

Introduction

Roadways are used by vehicles and livestock within farm boundaries (termed lanes in New Zealand). On dairy farms, roadways are used by dairy cattle twice daily to go to the milking shed, during which time they deposit dung on the roadway. Runoff from these roadways is acknowledged as a significant source of contaminant transfer to nearby streams (Hively *et al.*, 2006). However, apart from a few studies they remain an understudied part of farm scale losses. Monaghan & Smith (2012) found that the concentration of faecal contaminants in runoff decreased with increasing distance away from the milking parlour. Furthermore, McDowell *et al.* (2007) found that in one case, runoff from a roadway accounted for 80% of the phosphorus (P) load in a nearby stream. However, the amount of dung-P available for loss is highly variable depending on deposit weight, P content and number (Vadas *et al.*, 2015).

Our objective was to determine the concentration of contaminants in roadway runoff and determine if the load and concentration of contaminants (P, *Escherichia coli* [*E. coli*] and suspended sediment [SS]) could be decreased with a filter material as has been used to reduce P and SS loads in runoff from grassland and cropland (Ballantine & Tanner, 2010; Buda *et al.*, 2012; Karczmarczyk *et al.*, 2016). We modified the filter material (steel melter slag) to increase P sorption given the sensitivity of downstream waterbodies in the study area. To place the P results of the present study into a broader context an uncertainty analysis was conducted using ranges of values gathered from the literature pertaining to deposit weight, P content and number of deposits per hectare of roadway.

Materials and methods

Site description and field setup

The Mangakino stream feeds into the P-limited Lake Rerewhakaaitu (Abell *et al.*, 2010, 2011), one of the Rotorua lakes. Land use within the catchment is representative of the surrounding area with a dominance of pastoral agriculture (70% dairy, 7% sheep and beef, 15% forestry blocks and 8% other mixed land uses). In July 2006, 200 L of a 1% solution of aluminium chlorohydrate (ACH; Orica Chemicals, Newmarket, Auckland, NZ) was sprayed onto 2 t of steel

---

[†]Corresponding author: O. Fenton
E-mail: owen.fenton@teagasc.ie

melter slag; hereafter called ACH-altered slag. We chose to use ACH-altered slag because of its high P sorption affinity and low cost and toxicity compared to other readily available products (McDowell *et al.*, 2008). The ACH-altered slag was placed in a 50-m long ditch that had been dug alongside one half of the roadway. On the other side, a similar ditch was dug but no ACH-altered slag installed (Figure 1). The roadway cut across the stream, resulting in runoff discharging into the stream. At these points, a perforated polyvinyl chloride (PVC) pipe was connected to tipping-buckets to collect ACH-altered slag and control runoff (Figure 1). The tipping buckets were calibrated to catch 1% of sample after each tip. Samples from the tipping buckets were collected in response to rainfall events (>10 mm) for 12 mo. A subsample of each water sample was filtered (<0.45 μm cellulose acetate syringe filter) and measured for dissolved reactive P (DRP) and, after persulfate digestion, total dissolved P (TDP). An unfiltered subsample was also measured for total P (TP) after persulfate digestion (Eisenreich *et al.*, 1975). All P determinations were made colorimetrically (Watanabe & Olsen, 1965). Dissolved organic P (DORP) was obtained by the difference of TDP−DRP, and particulate P by the difference of TP−TDP. SS was determined on the remaining sample volume by filtration through a GF/A glass fibre filter paper and weighing the oven-dried (105°C) residue. The faecal indicator bacteria – *E. coli* was enumerated using the Colilert® media and the Quanti-Tray® system (IDEXX Laboratories, Westbrook, ME, USA). Data from runoff events during the year were checked for normality and log-transformed if necessary, before being analysed via a two-tailed *t*-test of paired data. Data for concentrations were multiplied by the volume of runoff to calculate loads on a per event basis. These were summed across all events and expressed on a per hectare basis (viz. yield) assuming a catchment area of 200 m^2.

Figure 1. Picture showing one half of the road with ACH-altered slag and tipping bucket installed. Note the Mangakino stream tributary flows through a culvert located beneath and between the two tipping buckets.

Uncertainty analysis

An uncertainty analysis was conducted specifically for P using an Excel-based Monte Carlo model. Inputs into the model were data ranges as in Table 1 for dung wet weight (kg), number of deposits on roadway per ha, dung total P concentration (g/kg dry weight). For the Monte Carlo analysis of total P deposited on a 1 ha section of roadway, we assumed all input variables had a uniform distribution. The model was run 1,000 times, whereby the model selected a value for each parameter within the range specified in Table 1. This enabled a probability distribution of outcomes to be achieved. The results of the present study were then compared to this full range of possible outcomes and loads were compared using the actual runoff data.

Table 1: Weight, frequency and P concentration of dairy cattle dung used in the uncertainty analysis

Parameter	References
Dung wet weight (kg) range = 1.5–2.7	• 1.5–2.6 kg (Haynes & Williams, 1993) • 2.0 kg (Krol *et al.*, 2016; Bacher *et al.*, 2018) • 2.5 kg (Bell *et al.*, 2015)
No of deposits on roadway/ha cow = 1–1.5	• Dairy cattle defecate 10.5 times/cow per day (Oudshoorn *et al.*, 2008). Therefore, a low percentage of this could be on roadways • Davies-Colley *et al.* (2004) investigated a 200 m roadway, 245 cows and showed that five deposits were made during each milking event
Dung P concentration (g/kg dry weight) = 4–8 g P/kg	• Estimated for lactating dairy cattle (McDowell, 2006; Vadas *et al.*, 2015)

Results

Field trial

Data for the concentration of P fractions, SS and *E. coli* in runoff from the ACH-altered slag and control section of the roadway are given in Table 2, along with the probability of a significant difference ($P < 0.05$). In all cases, concentrations were greater in the control than in the ACH-altered slag sections for DRP, PP, TP and SS. Concentrations of DRP, TP and SS were well in excess of recommended limits for freshwater eutrophication in disturbed (i.e. agricultural) lowland streams and for aquatic ecosystems and *E. coli* concentrations deemed fit for contact recreation (Australian and New Zealand Governments, 2018). However, this should only be taken as an indication of the potential to enrich the receiving stream and runoff will likely be diluted during stormflow. Contextually, concentrations of at least TP were like those noted for septic tank discharge (1–14 mg/L) (Withers *et al.*, 2011).

The loads across all events along with the percent decrease due to treatment with ACH-altered slag are given in Table 3. Loads from the ACH-altered slag treatment were generally lower than from the control treatment except for DOP (−7%), a P species that commonly exhibits poor sorption characteristics (Andersen *et al.*, 2016). In contrast, 88% of DRP and 96% of PP species were mitigated by the ACH-altered slag inferring that efficient sorption and filtration occurred. The efficiency of removal decreased with increasing event size, becoming ineffective for events with about 1,000 L of runoff, which equated to about 40 mm of rainfall (Figure 2).

Uncertainty analysis

Using the parameter ranges in Table 1, a probability distribution based on 1,000 runs of the model is shown in Figure 3. This

Table 3: Loads of runoff, P fractions, sediment and *E. coli* in the ACH-altered slag and control roadways and the percentage mitigation (i.e. the fraction of load from ACH-altered slag vs. control roadways)

Parameter	Mean load for ACH-altered slag	Mean load for control	Percent decrease
Runoff (L/ha)	3,133,000	3,243,750	3
DRP (kg/ha)	0.3	2.5	88
DOP (kg/ha)	0.5	0.5	−7
PP (kg/ha)	0.7	17.0	96
TP (kg/ha)	1.5	20.0	93
SS (mg/ha)	0.4	21.6	98
E. coli (cfu)	4.44E+10	1.93E+11	77

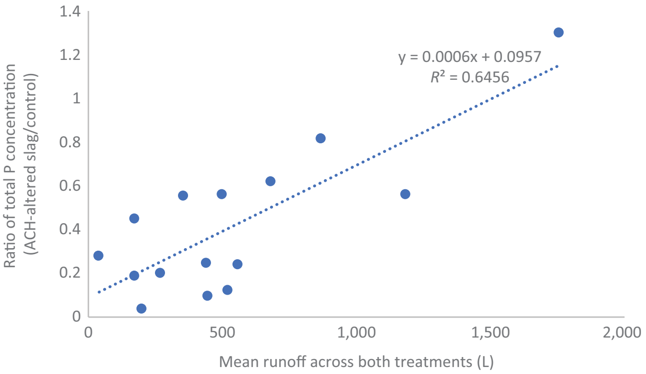

Figure 2. Relationship between mean runoff volume for the two treatments and the ratio of total P concentration in each event from the ACH-altered slag compared to the control treatments. Ratios above 1 indicate no removal occurred.

Table 2: Mean ± s.e. (range) of the volume of runoff and concentrations of P fractions, sediment and *E. coli*, and the probability of a significant difference between the ACH-altered slag and control roadways

Parameter	ACH-altered slag	Control	Significant difference (probability for *t*-test)
Runoff (L)[1]	367 ± 69 (53–1,760)	588 ± 123 (53–1,751)	0.574
DRP (mg/L)	0.060 ± 0.013 (0.004–1.560)	0.301 ± 0.053 (0.004–2.312)	0.001
DOP (mg/L)	0.149 ± 0.049 (0.001–0.906)	0.135 ± 0.022 (0.001–0.571)	0.798
PP (mg/L)	0.604 ± 0.048 (0.240–6.860)	2.046 ± 0.556 (0.010–6.640)	0.010
TP (mg/L)	0.813 ± 0.136 (0.176–7.623)	2.482 ± 0.590 (0.166–8.853)	0.006
SS (mg/L)	395 ± 93 (240–13,680)	2,720 ± 1,103 (26–4,950)	0.049
E. coli (cfu 100/mL)	3,629 ± 1,406 (100–24,190)	5,755 ± 1,901 (50–14,500)	0.088[2]

[1] Number of runoff events = 18.

[2] Data required log transformation for comparison of means.

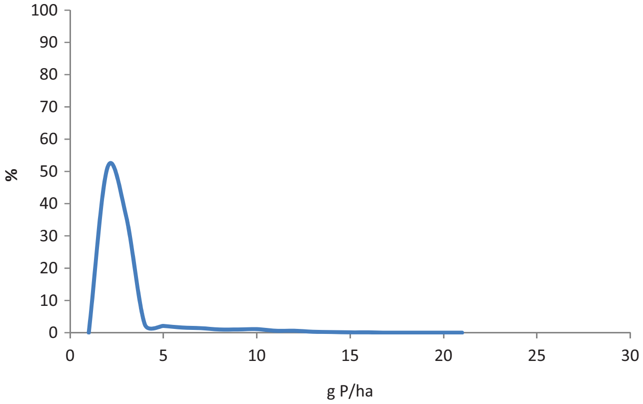

Figure 3. Probability distribution of g P day deposited on a hectare of roadway. This can be multiplied by cow number and time period; for example, 3 g TP × 100 cows × 365 days = ~100 kg.

shows that for approximately 50% of outcomes, 2.5 g TP/ha was available to be lost in runoff. When this is converted into a load using the runoff data presented in Table 3 the minimum TP load is 7.7 kg/ha and the maximum is 462.2 kg/ha.

Discussion

It should be noted that this trial did not have spatial replication nor directly measure manure loads. As such our findings should be taken as a proof of concept, informing the design of a more comprehensive study.

Considering that deposition of dung on the roadway occurs daily, concentrations of P lost are about one-fifth of those expected in runoff from a fresh dung patch as documented by McDowell (2006). However, this does depend on the rate of drying, diet and above all climatic conditions.

Installing ACH-altered slag alongside a roadway could be viewed as like an active filter bed used as pre or post treatment for wetlands (Ballantine & Tanner, 2010). These beds are influenced by flow rates and the concentration of inflowing water. In studying active filters with steel melter slag as a retention material, Shilton *et al.* (2005) noted that overall P removal from inflowing waste (piggery effluent or domestic effluent) water in two steel melter slag filters – one at Ashurst and another at Waiuku (New Zealand) – was 72 and 77%, respectively, but was most efficient in summer and autumn during low flows. Our data indicated a similar effect in that DRP load from the ACH-altered slag treatment was much lower than that of the control section at lower flows, but decreased at higher flows becoming ineffective when >1,000 L of runoff or 40 mm of rainfall occurred resulting in the filter being overtopped (Figure 2).

The cost per kg of P retained by unaltered steel melter slag when used in P-socks in a streambed to remove P from baseflow was estimated to be 30 USD (McDowell *et al.*, 2007), which precluded its use when compared to dosing streams with alum (at about 10 USD/kg P precipitated) (Pilgrim & Brezonik, 2005), but was more cost-effective than dosing the roadway directly with alum (McDowell & Nash, 2012; Smith & McDowell, 2016). However, the difference in cost between ACH-altered slag and unaltered slag is negligible as it is a waste material and the 1% solution of ACH is cheap. The resulting cost effectiveness would be about 3 USD/kg P retained. However, this would vary according to site characteristics (e.g. slope) and frequency of road use by cattle. It is also likely that the P-sorption capacity of the ACH-altered slag would decrease with time as sorption sites became occupied. Although the P sorption capacity of a similar product was not exceeded when included as part of a backfill for an artificial drainage network, this was exposed to much less sediment load (McDowell *et al.*, 2008). We have no data to confirm the longevity of our material beyond the 12-mo length of the trial. It should also be noted that other filter media may be more suited to a specific geographical location. Decision support tools are now available to help match pollutant and filter medium type (Ezzati *et al.*, 2019).

In terms of the uncertainty analysis results the present study falls within the lower range of potential losses, that is, 20 kg/ha. This range represents all possible sites where deposit weight, TP content and number of deposits range within a hectare of roadway. Preliminary data suggests that the use of ACH-altered slag was effective at mitigating P losses in runoff from a roadway (93% of TP) supplying P loss into the Mangakino stream. We therefore recommend the use of a material like ACH-altered slag to remove P from runoff from roadways entering streams. However, the uncertainty analysis did suggest that the deposition of dung-P on the roadway could be higher, which may reduce the efficiency of the ACH-altered to retain P. A reduced efficiency would need to be factored into the design of any filter bed.

Acknowledgements

The study was funded by Environment Bay of Plenty. Steel slag was supplied by SteelServ, Waiuku, South Auckland. We thank the Our Land and Water National Science Challenge for funding the first author while writing-up the study (contract C10X1507 from the New Zealand Ministry of Business, Innovation and Employment). The Monte Carlo analysis work was funded by the Irish Environmental Protection Agency and Department of Agriculture, Food and the Marine as part of the ROADRUNNER project; grant number 2018-W-MS-38.

References

Abell, J., Özkundakci, D. and Hamilton, D. 2010. Nitrogen and phosphorus limitation of phytoplankton growth in New Zealand lakes: implications for eutrophication control. *Ecosystems* **13**: 966–977.

Abell, J.M., Özkundakci, D., Hamilton, D.P. and Miller, S.D. 2011. Relationships between land use and nitrogen and phosphorus in New Zealand lakes. *Marine and Freshwater Research* **62**: 162–175.

Andersen, H.E., Windolf, J. and Kronvang, B. 2016. Leaching of dissolved phosphorus from tile-drained agricultural areas. *Water Science and Technology* **73**: 2953–2958.

Australian and New Zealand Governments. 2018. Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Governments and Australian state and territory governments, Canberra, ACT, Australia.

Bacher, M.G., Fenton, O., Bondi, G., Creamer, R.E., Karmarkar, M. and Schmidt, O. 2018. The impact of cattle dung pats on earthworm distribution in grazed pastures. *BMC Ecology* **18**: 59. doi:10.1186/s12898-018-0216-6.

Ballantine, D.J. and Tanner, C.C. 2010. Substrate and filter materials to enhance phosphorus removal in constructed wetlands treating diffuse farm runoff: a review. *New Zealand Journal of Agricultural Research* **53**: 71–95.

Bell, M., Rees, R., Cloy, J., Topp, K., Bagnall, A. and Chadwick, D. 2015. Nitrous oxide emissions from cattle excreta applied to a Scottish grassland: effects of soil and climatic conditions and a nitrification inhibitor. *Science of the Total Environment* **508**: 343–353.

Buda, A.R., Koopmans, G.F., Bryant, R.B. and Chardon, W.J. 2012. Emerging technologies for removing nonpoint phosphorus from surface water and groundwater: introduction. *Journal of Environmental Quality* **41**: 621–627.

Davies-Colley, R.J., Nagels, J.W., Smith, R.A., Young, R.G. and Phillips, C.J. 2004. Water quality impact of a dairy cow herd crossing a stream. *New Zealand Journal of Marine and Freshwater Research* **38**: 569–576.

Eisenreich, S.J., Bannerman, R.T. and Armstrong, D.E. 1975. A simplified phosphorus analysis technique. *Environmental Letters* **9**: 43–53.

Ezzati, G., Healy, M.G., Christianson, L., Feyereisen, G.W., Thornton, S., Daly, K. and Fenton, O. 2019. Developing and validating a decision support tool for media selection to mitigate drainage waters. *Ecological Engineering: X* **2**: 100010.

Haynes, R.J. and Williams, P.H. 1993. Nutrient cycling and soil fertility in the grazed pasture ecosystem. *Advances in Agronomy* **49**: 119–199.

Hively, W.D., Gérard-Marchant, P. and Steenhuis, T.S. 2006. Distributed hydrological modeling of total dissolved phosphorus transport in an agricultural landscape, part II: dissolved phosphorus transport. *Hydrology and Earth System Sciences Discussions* **10**: 263–276.

Karczmarczyk, A., Bus, A. and Baryła, A. 2016. Filtration curtains for phosphorus harvesting from small water bodies. *Ecological Engineering* **86**: 69–74.

Krol, D.J, Carolan, R., Minet, E., McGeough, K.L., Watson, C.J., Forrestal, P.J., Lanigan, G.J. and Richards, K.R. 2016. Improving and disaggregating N2O emission factors for ruminant excreta on temperate pasture soils. *Science of the Total Environment* **568**: 327–338.

McDowell, R.W. 2006. Contaminant losses in overland flow from cattle, deer and sheep dung. *Water, Air, and Soil Pollution* **174**: 211–222.

McDowell, R.W. and Nash, D. 2012. A review of the cost-effectiveness and suitability of mitigation strategies to prevent phosphorus loss from dairy farms in New Zealand and Australia. *Journal of Environmental Quality* **41**: 680–693.

McDowell, R.W., Hawke, M. and McIntosh, J.J. 2007. Assessment of a technique to remove phosphorus from streamflow. *New Zealand Journal of Agricultural Research* **50**: 503–510.

McDowell, R.W., Sharpley, A.N. and Bourke, W. 2008. Treatment of drainage water with industrial by-products to prevent phosphorus loss from tile-drained land. *Journal of Environmental Quality* **37**: 1575–1582.

Monaghan, R.M. and Smith, L.C. 2012. Contaminant losses in overland flow from dairy farm laneways in southern New Zealand. *Agriculture, Ecosystems & Environment* **159**: 1 70–175.

Oudshoorn, F.W., Kristensen, T. and Nadimi, E.S. 2008. Dairy cow defecation and urination frequency and spatial distribution in relation to time-limited grazing. *Livestock Science* **113**: 62–73.

Pilgrim, K.M. and Brezonik, P.L. 2005. Evaluation of the potential adverse effects of lake inflow treatment with alum. *Lake and Reservoir Management* **21**: 77–87.

Shilton, A., Pratt, S., Drizo, A., Mahmood, B., Banker, S., Billings, L., Glenny, S. and Luo, D. 2005. ‘Active’ filters for upgrading phosphorus removal from pond systems. *Water Science and Technology* **51**: 111–116.

Smith, L.C. and McDowell, R.W. 2016. The use of alum to decrease phosphorus loss from dairy farm laneways in southern New Zealand. *Soil Use and Management* **32**: 69–71.

Vadas, P.A., Busch, D.L., Powell, J.M. and Brink, G.E. 2015. Monitoring runoff from cattle-grazed pastures for a phosphorus loss quantification tool. *Agriculture, Ecosystems & Environment* **199**: 124–131.

Watanabe, F.S. and Olsen, S.R. 1965. Test of an ascorbic acid method for determining phosphorus in water and NaHCO3 extracts from soil. *Soil Science Society of America Journal* **29**: 677–678.

Withers, P.J.A., Jarvie, H.P. and Stoate, C. 2011. Quantifying the impact of septic tank systems on eutrophication risk in rural headwaters. *Environment International* **37**: 644–653.