# A review of on-farm roadway runoff characterisation and potential management options for Ireland

**RESUBMISSION**

# A review of on-farm roadway runoff characterisation and potential management options for Ireland

O. Fenton[a,*], P. Tuohy[b], K. Daly[a], T. Moloney[a], P. Rice[a], J. G. Murnane[c]

[a]Environmental Research Centre, Teagasc, Johnstown Castle, Co., Wexford, Ireland

[b]Animal and Grassland Research and Innovation Centre, Teagasc, Moorepark, Fermoy, Co., Cork, Ireland

[c]School of Engineering, University of Limerick, Limerick Ireland

*Corresponding author:

Email address: owen.fenton@teagasc.ie (O. Fenton)

## ABSTRACT

Runoff from farm roadways within farm boundaries are acknowledged as a year round source of pollutants discharging to surface water, particularly during the main grazing season (Feb-Nov) when their usage is high. **These losses are considered to be a significant catchment scale pressure and have led to recent legislation in Ireland to prohibit direct runoff from farm roadways to waters with similar guidance in the UK and New Zealand. However roadway runoff (RR) remains an undocumented and understudied part of the transfer continuum where knowledge gaps remain in relation to its quantity and composition, and in relation to the impacts of RR management options on adjacent surface waters**. **Indeed some information on mitigation measure design and efficacy is only available from non-agriculture land uses e.g. forestry and needs to be presented in a farm specific context. The present review brings together knowledge on RR in terms of content on- or off-roadway management options and proposes alternative mitigation measures that may require pilot scale testing.** Studies show that RR contains a mix of legacy (surface materials) and incidental (fresh urine and faeces) constituents such as

phosphorus (P), Escherichia coli (*E. Coli*) and sediment which become temporarily mobilised during rainfall events. Once mobilised, the roadway network can quickly transport pollutants providing connectivity between farmyards, hard standings, underpasses where attached to a watercourse, fields and even public roadways to watercourses. Its contents are not dissimilar to dilute slurry or dairy soiled water with loads being highest where animals congregate (within 100 m of the farmyard or where roadway configuration or quality impedes stock movement along the network). On-roadway management options include regularly spaced diversion structures that divert runoff for passive treatment in a field or drainage ditch or the application of chemical amendments to roadway surfaces. Off-roadway options can include natural or enhanced features, which attenuate flow leading to settlement of suspended sediment and in some cases treatment of a percentage of the load before delivery to a watercourse. It should be noted that the options presented in this review need to be retrofitted to site specific conditions. Therefore, a number of factors, including their effectiveness, cost and management needs on a particular farm, as well as local regulation requirements will need to be considered prior to implementation. **Therefore, only the measures that divert RR away from waters are applicable to Ireland.** Future research should develop and validate a farm roadway risk assessment tool to identify vulnerable roadway sections that need management and/or re-design, and identify means of incorporating RR into catchment scale risk assessment models. In addition, design of on- and off- RR management options should consider the wide variations in farm configuration and management with regard to size, topography, soil type/drainage class, enterprise, intensity, scale, stage of development and the potential impacts of climate change.

**Keywords**: catchment, agriculture, environment, pathway, farm roadways, runoff

## *1.0 Introduction*

*1.1 Farm roadway networks on dairy farms*

Farm roadways, which service private lands but are never considered public roadways, are a specific unsealed subset of the overall catchment road network (Kröger et al., 2012). Internationally farm roadways are also referred to as farm laneways (Monaghan and Smith, 2012; New Zealand), farm tracks (Adams et al., 2014; USA) and stock lanes (Lucci et al., 2010: EU). Within farm boundaries urine and faecal matter deposited by animals, pollutants from machinery movement and run-on from farmyards are deposited on roadway surfaces (Oudshoorn et al., 2008) and with the onset of rainfall are mobilised along surface slopes, as saturation or infiltration excess is exceeded. Farm roadways are typically constructed so that they can easily drain surface runoff to adjacent land or alternatively deliver this temporary runoff to a watercourse thereby alleviating ponding and waterlogging. **To protect water quality however, opportunities to intervene and prevent direct or indirect RR delivery to a watercourse or public roadway should be explored (Sharpley et al., 2015). An increased awareness of the potential contribution of farm roadway runoff (RR) to surface water impairment is reflected in recent legislation in Ireland (SI 605 of 2017) prohibiting direct runoff from farm roadways to waters with similar guidance in the UK (UK Environment Agency) and New Zealand (Ministry for the Environment). However, while such restrictions are being imposed, it is somewhat surprising that RR management is underrepresented in the literature and key knowledge gaps remain in relation to i) the seasonal content of RR, ii) current and future RR management options and iii) understanding of the long term and seasonal short term impacts of RR on adjacent surface waters. The present review focuses on the first two of these knowledge gaps and its objective is to a) report on the varied content and composition of RR and b) assimilate knowledge of on-roadway (e.g. diversion, discharge and treatment) and off-roadway management options while examining their**

**suitability and adaptability to break the direct connectivity between RR and water courses, with emphasis on sediment, N and P loss remediation. In addition, future options that may have potential for RR management are briefly discussed. The third knowledge gap, impacts of RR on surface waters, is currently being assessed separately.**

The primary function of a roadway network on grassland farms is to facilitate stock movement and maximise grass utilisation which is a key driver in increasing net profit and farm sustainability. An intensive roadway system facilitates increased labour efficiency and ease of operations at all times and makes grazing management easier during adverse weather conditions. **In Ireland,** Farm roadways typically occupy 1-2% of the grazing land area and their spatial layout and configuration varies significantly within and between farms. This variability will be due to enterprise type (e.g. beef versus dairy, grassland versus tillage), landscape and farm shape (roadway slope or length) or farmer preference. Farm roadways, open drainage ditches, and culverts influence hydrological and geomorphological processes significantly (Duke et al., 2003). Farm roads represent a category of semi-pervious areas with potential to generate infiltration-excess surface runoff (Srinivasan and McDowell, 2009).

**The significance of RR as a contribution to spatially distributed diffuse source pollution of waters is clear. For example, Buchanan et al. (2013a) indicate that manmade drainage networks (e.g., road- or agricultural-ditches) have a high level of connectivity with waters and contribute significantly to non-point source pollution. Finding where connectivity to waters exists along these networks enables possibilities for targeting water quality protection practices around or within these networks. For example, Buchanan et al. (2013b) found that 94% of connected roadway drainage ditches discharged to natural streams, effectively doubling the drainage density. Endreny and Wood (2003) allocated an export coefficient of 0.49 kg P ha$^{-1}$ for roadways**

**within the Paines Creek catchment in USA. In fact P-loading from RR can be substantial and, indeed, can approach or exceed that of agricultural runoff from fields in some settings (e.g. Easton et al., 2007; 2008; Wu et al., 1998).**

Farm roadway design and construction has not, to date, incorporated and prioritised diversion of runoff away from watercourses. While surface drainage of roadways is considered, especially where steep gradients promote the potential for scouring and loss of road surface layers, surface water would normally discharge to watercourses where adjacent and accessible. Typically, farm roadways closer to the farmyard tend to be better developed and maintained while those servicing the outer extremities can be underdeveloped or neglected over time. There also tends to be incremental development as new land is acquired or the farm layout is reconfigured. As such, a patchwork of different roadway types can be pieced together to form the roadway network with specific sections varying in terms of configuration, length, width, gradient, cross-fall, surface condition and constituent materials. Livestock will therefore encounter many different roadway types and configurations when walking. In many cases the core road network may have been in place for many years and originally designed for lower livestock numbers than currently exist. In this scenario, the roadways, even if well maintained, tend to be compromised in terms of width and as a result reduce livestock flow and comfort when in use (Teagasc, 2017; Dairy NZ, 2019). Efficient and comfortable livestock movement will be hindered by the extent of maintenance of the roadway, such as uneven/damaged surfaces, potholes, excessive dung/dirt, grassy margins, excessive shelter/shading (obscures view and promotes dirty surfaces and dampness) and with obstacles such as poor placement of water drinking troughs or excessively close fencing.

1.2 *Roadway runoff source, content and hotspots*

A dairy herd can make up to 600 return journeys per year between grazing areas and the milking parlour (Clarke, 2016). Several studies employing different grazing systems have also investigated where and how many times cows urinate and defecate (Oudshoorn et al., 2008; Hirata et al., 2011; Villetttaz Robichaud et al., 2011; Orr et al., 2012). Most relevant to the current review is the study of Oudshoorn et al. (2008), which measured the frequency of urination and defecation in pasture, in barns and in races (i.e. roadways). Results showed that the time spent by animals in a certain area correlated with the frequency of urination and defecation. Defecation occurred on roadways and in pastures, but there was greater variation of defecation along roadways than in pastures. Similarly, in a study to map the spatial and temporal distribution of dairy cattle excreta, White et al. (2001) also observed that the number of defecations correlated with time spent in a particular area. Highest concentrations of nutrients were in the vicinity of drinking troughs in fields and along roadways, particularly during warm weather.

Therefore, identifying roadway sections that receive greater loads and increasing the speed (i.e. flow and therefore time spent) of animals through these sections would likely decrease the load of source materials on roadways, thereby decreasing load losses in RR as mobilisation occurs. To aid with animal flow, consideration should be given to the roadway surface (smooth for animal hoof health, compact and water resistant to minimise infiltration), width (should increase as herd size increases), gateways (should be a consistent width) and configuration of the roadway network. The avoidance of “bottle necks” at particular points i.e. before the collecting yard, at junctions or through incorrect placement of water troughs along roadways (either in the field or roadside), and avoiding sharp bends or changes in direction will further facilitate ease of stock movement. White et al. (2001) showed that rousing animals in pasture before allowing them onto roadways could decrease on-road

153 urination and defecation. This practice could increase flow on roadways as cows need to stop
154 to urinate (Aland et al., 2002). All of these factors will minimise time spent travelling to and
155 from the milking parlour and **reduce** the potential for cow lameness. **In terms of RR**
156 **management, roadways should not be considered as stand-off areas and individual road**
157 **specifications in different jurisdictions should be followed closely.**

158

159 Depending on the location of inspection, farm RR will always be a combination of
160 different nutrient sources which can include farmyard run-on, spillage of organic fertilisers
161 from machinery, nutrients stored in the roadway surface from past losses and concentrated
162 nutrient deposition from manure and urine by cattle (Ledgard et al., 1999; Lucci et al., 2012;
163 Burchill et al., 2019). However, only a few studies have examined particulate and dissolved
164 runoff losses derived from such depositions in terms of concentrations or loads. Ledgard et al.
165 (1999) noted that approximately 5% of cow excreta N were deposited on roadways in a dairy
166 farmlet trial in Waikato, New Zealand. An assessment of P and suspended sediment (SS)
167 loads from sources within a dairy farm reported highest concentrations from farm roadways
168 (5.98 mg TP $m^{-2}$; 875.6 mg SS $m^{-2}$) and areas around drinking troughs (4.20 mg TP $m^{-2}$;
169 190.2 mg SS $m^{-2}$) with lowest concentrations from pastures (0.78 mg TP $m^{-2}$; 73.14 mg SS $m^{-2}$)
170 (Lucci et al., 2010).

171

172 Monaghan and Smith (2012) showed that RR concentrations of nutrients and E. coli
173 increased near farmyards (within 100 m) or along roadway sections up to 700 m from
174 farmyards due to deposition of animal excreta in these more highly trafficked sections of the
175 farm roadway network. In addition, these concentrations were higher during sampling events
176 that occurred when animals were outside and lower during the winter period where animals
177 were kept indoors. Total nutrient loads in runoff expressed as g $m^{-2}$ were as follows: TP 3.3,

178 DRP 1.3, ammonium-N ($NH_4$-N) 3.8, dissolved organic N 3.1, total SS 358 while the *E.coli*
179 load in runoff was 3.8 x $10^7$ MPN $m^{-2}$.
180

181 Cow diet and cow type also have a role to play in the nutrient content of faeces and
182 this should not be considered homogeneous. As depicted in Powell et al. (2013), when dairy
183 feed N exceeds 26.4 g $kg^{-1}$ dry matter (DM), there is a propensity for the 'nitrogen use
184 efficiency' (i.e. the proportion of dietary N (consumed feed crude protein (CP) divided by
185 6.25) converted to milk or live weight gain (kg)) to decline dramatically. This leads to an
186 increased amount (kg) of N being excreted (NEX), especially urinary N, mostly as urea, a
187 reactive form of N that is highly susceptible to environmental loss. Additionally, in research
188 carried out by Rice et al. (submitted), in relation to the environmental impact of three dairy
189 breeds (Holstein Friesian (HF), Jersey (J), or HF *J cows ($F_1$)), the $F_1$ cows had lower NEX
190 than the other 2 breeds per unit of dietary N fed (g N/ kg DM).
191
192 *1.3 Farm roadway runoff as a significant pressure at catchment scale*
193 Roadways within agricultural catchments are acknowledged as an important pathway
194 for nutrient and sediment loss providing connectivity within the land-water continuum
195 (Hively et al., 2006; Collins et al., 2010; Wemple, 2013). In a study conducted in New
196 Zealand by Monaghan and Smith (2012) across a range of dairy farms within the Bog Burn
197 study catchment, it was shown that runoff from farm roadways accounted for < 1% of the
198 annual catchment loads of total P (TP), dissolved reactive phosphorus (DRP), total suspended
199 nitrogen (TSN) and total suspended solids (TSS). However, if loads measured over summer
200 are expressed as a proportion of measured summer catchment loads, roadways could
201 contribute between 4 and 76% of TP loads, or between 6 and 100% of TSS loads, assuming

that 5 or 100% of laneway areas discharge directly to surface water, respectively. Similar patterns were found in another study by McDowell et al. (2020) who examined roadway runoff from a section of roadway trafficked by cows twice daily on a dairy farm in New Zealand across 18 rainfall events where a total of 3,243,750 L of runoff was collected. Results showed load losses of 2.5, 0.5, 17.0, 20.0, 21.6 and 1.93E+11 for DRP, dissolved organic P, particulate P, TP, SS (kg $ha^{-1}$) and *E.coli* (colony forming units, cfu*)*, respectively. For approximately 50% of outcomes, ≥2.5 g TP $ha^{-1}$ was available to be lost in runoff. Using uncertainty analysis and default data for cow dung weight (kg), number of deposits on a roadway section per hectare and the TP content of a dung deposit, the estimated TP load, from a 100 cow dairy herd, for any roadway ranged from 7.7 to 462.2 kg $ha^{-1}$. Once again in that study the greatest losses occurred during summer months. This season is interesting in Ireland as over the past 30 years there has been a shift in weather patterns with more episodic high intensity rainfall events occurring in the summer period (Tuohy et al., 2018).

*1.4 Potential on-roadway and off-roadway runoff management measures*

It is likely that for maintenance purposes RR is already being diverted and discharged from existing farm roadway networks and consequently their starting position from a pollution perspective is not a neutral one. At present, instead of retaining water and slowing the flow, the agricultural landscape has been modified to speed up flow to surface water. Many roadway sections act as hard surface conduits speeding up sediment and dissolved pollutant losses from farms to watercourses. During rainfall events, critical (stored in the roadway media) and incidental (farmyard run-on and on-roadway micturition and defecation) losses of nutrients can be incorporated into RR, which may discharge directly to a field, ditch (at breakthrough discharge points), public roadway or a surface waterbody (delivery

discharge point) (Thomas et al., 2017). However, roadways can also be seen as potential contaminant retention features if managed correctly.

On-roadway management of runoff may include measures such as (i) flow attenuation through diversion or (ii) use of roadway surface chemical amendments (e.g. alum, ferric chloride or polyaluminium chloride) to flocculate nutrients out of suspension and bind them to roadway sediments. Flow attenuation can be achieved to varied degrees by introducing diversion structures such as grade breaks or channels installed at regular intervals on sloped roadways. Other diversion techniques are through cross-fall manipulation of the roadway (LEAF, 2010). It should be noted that diversion along a roadway to a ditch or direct discharge to a field will be easier at some locations than others. For example, diversion of runoff may become complicated at locations where ponding could occur, due to soil type or relief, or in locations directly adjacent to a watercourse, where discharge away from such a feature would be problematic. Diverting RR **away from waters** to a discharge point and into a field **or towards waters into a ditch (not applicable under current regulations in Ireland)** slows water movement from land to surface waters during peak rainfall. The degree to which this phenomenon occurs naturally varies due to inherent soil or sediment conditions after the discharge point. Slowing the flow can also occur through mechanical solutions e.g. sub-soiling in fields or installation of engineered solutions such as weirs or gates in ditch systems. The infiltration rate of soils in fields may not always be adequate to prevent diverted runoff from ponding or reaching a watercourse. In some tillage scenarios using these ponded areas as sinks (Thomas et al., 2018) by creating a vegetated scrape bund (LEAF, 2010) or a soakage (DAFM, 2019) or vegetated area, may facilitate water retention and attenuation of nutrients in the RR whilst still breaking connectivity. Engineered treatment systems which intercept diverted RR at or near surface runoff pathways are now incentivised and are in

place on many landscapes (e.g. in forestry (Akbarimehr and Naghdi 2012)). For example, in-ditch engineered structures in different parts of the world such as woodchip biological denitrifying bioreactors (Schipper et al., 2010) can treat ~20% of the nitrogen (N) during peak flow conditions through conversion of nitrate ($NO_3^-$) into di-nitrogen ($N_2$) gas. Other examples include a variety of structures (e.g. combined bed) filled with a permeable phosphorus (P)-sorbing media treating ~20-30% of the dissolved reactive phosphorus (DRP) load in ditch networks (Penn et al., 2007).

*2.1 Existing on-roadway management options*

Given the potential of farm roadways to increase the hydrological connectivity of diffuse pollution with receiving waters (Shore et al, 2013; Thomas et al, 2017) it seems logical that their design should facilitate measures which retain pollutant loads along their length, for example by controlled discharge points for off-road treatment. A key aspect of this treatment is the separation of nutrient enriched sediment from the liquid portion of the runoff (LEAF, 2010). This can be achieved by flow attenuation which mutes runoff at peak rainfall periods and thereby decreases load transfer of nutrients and sediment along the transfer continuum (Haygarth et al., 2005).

The captured solids can potentially be reused as a fertiliser while the liquid portion will require further treatment (natural or engineered) to reduce nutrient concentrations prior to connecting with the receiving watercourse. An overarching consideration for on-road management options are the impacts of climate change and weather patterns such as storm events (Mellander et al., 2013) and seasonal as well as inter-annual variations (Jennings et al., 2013; Sherriff et al., 2015), although the extent to which these impact on the relationship between agriculture and aquatic ecosystems remains somewhat uncertain (Doody et al.,

2019). There are relatively few examples of on-roadway management options available in the literature (e.g. Scheetz and Bloser, 2009; STTI, 2018) and most of these relate to roadway surface modifications which can be applied to new or existing roadways.

### *2.2 Roadway surface modifications or amendment*

The overall objective of carrying out roadway surface modifications is to manage and direct RR away from watercourses to selected discharge points where flow can be attenuated, preferably in an adjacent field, prior to off-roadway treatment. These modifications are generally cheap and easy to implement; however their effectiveness is dependent on selecting the most appropriate option for the scenario in question.

Surface profile modifications may involve provision of a single camber to the roadway away from watercourses to direct runoff to adjacent fields where it can be naturally attenuated by the soil. Where the roadway is located away from a watercourse a double camber comprising a continuous slope from the roadway centreline to the roadway edge may be employed to allow fields on both sides of the roadway be used for attenuation (Scheetz and Bloser, 2009). The depth of water retained by roadway surfaces is influenced by the slope and length of the flow paths taken by the runoff, the rainfall intensity and the roadway texture depth (Galloway et al., 1971). The flow paths are dictated by both longitudinal and cross sectional gradients and where the longitudinal gradient is relatively low; the flow path is predominantly influenced by the cross-fall gradient. In situations where there is a steep longitudinal gradient however, there is an increased risk of high velocity flow (flow channelling) bringing with it the risk of surface erosion and transport of nutrients to a downstream discharge point. In cases where the roadway surface is (or has become) lower than the surrounding terrain it may also be necessary to raise the roadway profile to eliminate

299 additional flow contributions from adjacent fields (Sheetz and Bloser, 2019). Where this
300 measure is applied, it may be necessary to provide hydraulic connectivity between both sides
301 of the roadway by constructing a “french mattress” or underdrain (DGVLR, 2019). This is to
302 prevent build-up of water in cases where the roadway acts as a barrier to natural water
303 movement. In a study to measure the impacts of forest roadways on the hydrologic response
304 of subsurface flow in a steep forested landscape, Wemple and Jones (2003) found that many
305 roadway sections altered the timing and magnitude of the catchment hydrograph, particularly
306 during large storm events when flows were diverted more quickly to drains. The authors
307 concluded that their study approach may be a useful hydrologic tool when assessing removal
308 or restoration of roadway segments. Similarly, in a study to assess the hydrological
309 connectivity between $NO_3^-$ and P to streams, Outram et al. (2016) noted that in addition to
310 antecedent ground conditions, the amount and timing of rainfall events had a significant
311 influence on runoff and subsequent amounts of mobilised N and P fluxes. Removal of all or
312 sections of roadside berms to allow surface water drain to adjacent fields is also a simple and
313 effective roadway management option; however, care should be taken not to remove all of
314 the grassy edge as this may be required to maintain the structure and integrity of the road.

315 On-roadway surface diversions generally consist of physical barriers or structures to
316 intercept and direct surface water to designated outlets. One such barrier, termed a grade
317 break, comprises an elongated hump formed on the road to create a reverse gradient so that
318 the surface water is physically directed to a designated outlet. The spacing of grade breaks
319 depends on the slope of the roadway as well as the number of available outlets for the runoff,
320 however their effectiveness on roadways with a longitudinal slope of >10% is reduced
321 (DGVLR, 2019 (Grade Break), Fig 1). As an alternative to grade breaks, the use of mine
322 conveyor belts (DGVLR (Conveyor Belt Diversion), Fig 1 (2) embedded into the roadway at
323 an angle to divert longitudinal flow has also been trialled in the USA (Sheetz and Bloser,

2009). While these belts are reported to be effective and flexible enough to allow vehicles to pass there is a lack of information available on their impact on the movement of farm animals. The use of open channel drains to intercept and redirect longitudinal flows along the roadway surface has also been proposed as an effective way of catching runoff and has the advantage that they are visible and can be easily cleared of solids as necessary. However, consideration must be given to the location of these open channels as they pose a potential risk for animals accidentally stepping into them and thus they may be a more viable option for roads trafficked with machinery (Fig 3). Similar to the use of grade breaks, the depth and spacing of these channels will depend on the amount of surface runoff they are required to intercept (STT1, 2018; Fig 4); however, their overall success does not seem to have been reported in the literature to date and a potential drawback of this type of measure is that the channel sections might fill up with RR or other loose material and reduce their effectiveness.

The use of overflow structures such as swales, infiltration areas and buffer areas such as scrape bunds (LEAF, 2010) to attenuate flows may be suitable in cases where the RR volume is high. These overflows may be implemented by simply blocking or partially blocking downstream flow and diverting it to these overflow structures. An example of the use of a scrape bund to retain RR and diffuse pollutants from a sloping field on a farm in Cambridgeshire, UK was found to be highly effective (LEAF, 2010) with the authors suggesting that the captured water should drain from the retention area over a 24-hour period in preparation for the next rainfall event. In an evaluation of constructed overflow areas to attenuate diffuse pollution from forest drains, Liljaniemi et al. (2003) reported no significant differences between concentrations of chemical oxygen demand, $PO_4^+$, TP, TN, $NH_4^+$, $NO_2^-$, $NO_3^-$, Fe and Al upstream and downstream of their overflow areas and suggested that wider buffers with more extensive overland flow areas would be needed to control diffuse pollution.

Amendments such as ferric chloride, poly-aluminium chloride, alum, lime and other chemical amendments have been studied extensively in terms of their ability to coalesce and trap nutrients, carbon and sediment in both surface and subsurface runoff after field application (Brennan et al., 2011, O'Flynn et al., 2012; Murnane et al., 2015). Results have proven the efficacy of these amendments to decrease both critical and incidental losses of P and N in runoff. In addition, the build-up of metals in soil and plant material is not an issue (Healy et al., 2018). However, in terms of manure management these amendments were not perceived as cost effective for broad scale application. This viewpoint has now changed as in-storage amendment is considered as a greenhouse gas and ammonia abatement measure (Teagasc, 2019). The efficacy of using off-the-shelf chemical amendments during the storage of slurry to prevent gaseous emissions has been investigated (Kavanagh et al., 2019). Smith and McDowell (2016) showed that roadways amended with chemical flocculants minimised P loads in runoff while alum amendment rates of 25 and 50 kg $ha^{-1}$ initially lowered DRP losses for four events, over 60 days, following application. A similar approach has been deployed in ditch networks through amendment of sediments (Penn et al., 2007).

*2.3 Roadway re-routing*

In cases where poaching and compaction of soil is a direct source of pollution at the locations of field gates on roadways, it may be necessary to reroute the roadway and gate to a location where soil disturbance is less likely (STT1, 2018) or allow for multiple access points to decrease the level of traffic at any one location. In addition, it may be necessary to incorporate a runoff trap or attenuation area to capture sediment and retain RR, particularly where the soil has poor attenuation properties and therefore limited capacity for RR purification and nutrient recycling (Schulte et al., 2014).

## *2.4. Existing off-roadway management options*

### *2.4.1 Adjacent fields*

One of the most effective and obvious sources of treatment for RR are the adjacent fields, which provide indirect diffuse pathways to watercourses. The field boundary closest to roadways should be conducive to allowing RR to enter fields. Often slight berms can form due to cow movement along the boundary between the roadway and the field (Fig 2). The fate of RR entering a field is then dependent on the nature of the soil and drainage properties in the field. A quantification of soil hydrological response to precipitation requires knowledge of the rate at which water infiltrates vertically through the soil surface and into the soil profile. The parameters with a direct effect on infiltration rate are antecedent soil moisture, rainfall rate and soil physical properties. The thickness of the soil/subsoil and its permeability (low, moderate or high) are important site specific factors to consider the ratio of runoff to infiltration (Misstear et al., 2009). To calculate recharge volume a recharge coefficient (30-90%) is multiplied by the total available water (Fitzsimons and Misstear, 2006). Therefore, heavier textured soils, with a shallow water table and a low recharge coefficient will be more prone to ponding. In some soils, the infiltration rate of RR may be impeded by a landscape feature e.g. man-made feature such as a wall. This results in a localised and often temporary settlement pond, which may or may not need to be fenced off and managed periodically to remove sediment. Such features have been recognised as having a denitrification capacity as natural seepage features or as artificially created equivalents (Chibuike et al., 2019). Where this feature needs to be drained, piping to a “dry closed drain” may be an option. This assumes infiltration within the drain will attenuate nutrients without connectivity to a watercourse (DAFM, 2019).

Knowledge of recharge volumes is also important in terms of nutrient transport. For example, in terms of P in a well-drained soil (with a high recharge coefficient), shallow subsurface P concentrations typically mirror those of surface runoff where macropore networks connect these flow pathways during the early stages of a rainfall event whereas once saturation occurs matrix flow dominates. In heavier textured soils, matrix flow induces slower transport conditions and concentrations of P are influenced by bulk soil P concentrations in the deeper soil profile. These have lower concentrations of available soil P compared to surface values that receive applications of organic or chemical fertilisers. In free draining soils greater infiltration and less contact time can result in less desorption and mobilisation by overland flow along the surface of fields. However, P loads and export rates from fields are largely driven by the hydrological load (Kurz et al., 2005) and high rainfall rates and initial moisture contents can produce large surface overland flow rates and volumes. Infiltration rates across soil types are very important and should be incorporated into a multi-criteria P risk assessment scheme. External factors may also change the recharge coefficient such as the degree of compaction where increased soil bulk density means, by definition, a reduction in porosity. A change in bulk density affects many properties and processes in the soil including: water retention, saturated and unsaturated hydraulic conductivity as well as chemical processes, plant growth and soil workability. Poorly timed agricultural practices such as machinery traffic and high stocking rates in adverse conditions can result in the formation of soil horizons of high bulk density and low porosity in soils which ordinarily have good physical properties (Creamer and O'Sullivan, 2018).

Whilst soil physical quality influences the magnitude and direction of hydrological pathways for nutrients, soil chemical processes interact with these pathways releasing concentrations of P and N from the soil matrix. All soils can perform multiple functions of primary productivity (food, fuel and fibre), carbon sequestration, habitat provision, recycling

nutrients, as well as water purification (Coyle et al., 2015). The supply of each function highly depends on land use management and local soil properties including soil drainage (Schulte et al., 2015). Water purification (the capacity of a soil to remove harmful compounds) (Wall et al., 2018) is an important soil function where RR discharges to a field.

Physico-chemically adsorbed nutrients within the soil matrix can desorb into solution and along with physical detachment of soil particles/sediment into flow paths, these can contribute to the soluble and particulate losses of P from fields (Sandström et al., 2020). In terms of N, Fenton et al. (2009) proposed a model where nitrate concentration and denitrification rate were correlated with soil and subsoil permeability. Therefore, in terms of water purification heavier textured soils offer more protection but some studies have shown nitrate conversion to ammonium or gaseous phases in both mineral and organic soils (Clagnan et al., 2019). In terms of P, fields with a high soil test P, or high amounts of legacy P pose a greater risk of P loss into a connected hydrological pathway compared to fields that are low or deficient in P (Kurz et al., 2005). Other factors that influence these processes are largely those that describe the chemical composition of soil including soil acidity (i.e. pH), organic matter content, clay mineralogy and presence of amorphous forms of Al, Fe and Ca (Daly et al., 2015). These constituents can control the rates at which nutrients, specifically P, can be solubilised and removed from the soil matrix into solution. For example, low pH soils will retain nutrients in an insoluble form, whereas neutral soils will provide the right conditions for P solubilisation and desorption processes to occur. Acidic soils typically have high concentrations of soil Al, and if the Al:P ratio is high, these soils can fix P into insoluble forms (Daly et al.,2015). Likewise, the presence of clay minerals and amorphous forms of Ca in high concentrations can react with P to form Ca-P precipitates which remain largely insoluble (McLaren et al., 2014; Daly et al., 2015; McLaughlin et al., 2011). Therefore, RR

447 diverted into fields with a high propensity to fix P or generate insoluble P precipitate will be
448 advantageous to reducing diffuse P losses. Whilst most of the reactions with P in soil take
449 place on the surface of clay minerals without interference from other soil constituents,
450 organic matter (OM) if present in large amounts (>20% by loss on ignition) can occlude these
451 reaction (sorption/desorption) sites. This results in high concentrations of soluble P in soil
452 solution that cannot be physico-chemically adsorbed to the soil matrix (Daly et al., 2000;
453 Gonzalez et al., 2018; Gonzalez et al., 2019). Hydrological pathways that interact with high
454 OM soils are likely to have high concentrations of P in the soil solution and have a limited
455 ability to adsorb any additional P originating from RR. Therefore, diverted RR into fields
456 with high OM would exacerbate diffuse P losses from these fields, and are unlikely to
457 assimilate any P entering the field in RR. Physical detachment of soil particles into overland
458 flow contributes to the particulate P (PP) losses from agriculture, and diverted runoff onto
459 tillage fields or those with poor grass cover, could potentially increase the risk of PP losses to
460 water bodies (Roberts et al., 2017).

461

462 *2.4.2 Drainage ditch*

463 As an alternative to, or in conjunction with fields, RR can also be diverted to a
464 discharge point connected to a drainage ditch. However, it is important to note that RR
465 should never be discharged directly into first or second order small streams. First or second
466 order small streams, which are typically defined by a channel width of < 3 m and may arise
467 from rainfall events or from seepage through soils, subsoils, aquifers and springs or lake
468 outlets (Biggs et al., 2017), may be perennial or temporary and positioned within the
469 headwater of a river extending a short distance, typically only a few kilometres, from the
470 source.

Early work on drainage ditches showed that these networks had potential in terms of sediment, nutrient and E. coli retention (Nguyen et al., 2002; Nguyen and Sukias, 2002). Dollinger et al. (2015) reviewed design and maintenance of ditches to optimise the mitigation potential of these often extensive networks. The following overarching points are important when considering ditch networks as a convenient place to discharge RR. Firstly, ditch networks are not homogeneous. The individual ditch sections or 'reaches' that make up a ditch network can be defined according to their landscape position and chemistry. Using this approach Moloney et al. (2020) categorised five ditch types: farmyard connection, outlet, out-flow, secondary and disconnected. Furthermore, ditch categories are ranked according to their efficiency as connections between agricultural sources of P (both point and diffuse) and nearby surface waters. Implementing the right measure in the right place at the right time is key to water quality improvement and a ditch classification system like that of Moloney et al. (2020) could be useful in informing decisions regarding the selection and implementation of in-ditch mitigation measures on farms. For example, the mitigation potential of a particular measure placed in an outlet ditch that connects a large area of land to a stream compared to the same measure placed in a disconnected ditch is likely to differ significantly in favour of the former.

Ditch maintenance, typically dredging or deepening to maximise flow to the outlet, can have both positive and negative effects on nutrient attenuation. For example, over engineered ditch networks (e.g. Shore et al., 2015a) with low slopes had potential to retain fine sediment and ditches with exposed sediment and subsoil could in some cases immobilise nutrients in drainage waters (Shore et al., 2015b). Ezzati et al. (2020) found that P inputs can impact the source-sink P dynamics of sediment along an agricultural ditch network and therefore attenuation should not be assumed and where it occurs, can change over time. Provision of additional storage can dampen peak flows and mitigate flooding as well as

diluting nutrient concentrations. In-ditch vegetation at the discharge point could take up nutrients diverted from RR. However, this attenuation capacity is not uniform across nutrient species e.g. in vegetated versus non-vegetated ditches of similar size and landform. Moore et al. (2010) found no significant difference in $NH_4$, nitrate, or dissolved inorganic P mitigation but reported significant differences in total inorganic P loads. Other studies have found similar or contrasting results (Smith and Papas, 2007; Kröger et al., 2007; 2008).

Water retention features can be used to slow down water flow, thereby increasing the residence time to give further opportunities for runoff retention and biogeochemical N transformation to occur. Kroger et al. (2012) looked at the installation of low-grade weirs along ditch networks in the U.S. and measured the associated N load. Attenuation of $NO_3^-$ loads was reduced in ditches with (79%±7.5) and without (73%±9) weir installation (Kroger et al. 2012) or even as high as 92% in a six-year study in Italy (Tolomio and Borin 2018). Some of the most commonly used engineered measures are constructed wetlands (Blackwell et al., 2002; Kadlec, 2012), buffers (Braskerud, 2002; Fogg et al., 2005), controlled drainage and in-ditch techniques (Strock et al., 2010; Woli et al., 2010; Puckett, 2014). Ditches provide an ideal location for an engineered structure to mitigate nutrients in drainage waters such as ditch filters or in-ditch cartridges. There are of course many methods to trap sediment in ditches (Fig 5) or in fields (Fig 6), permanent options which need maintenance or temporary that need replacement e.g. silt curtain). There are also options to build in extra capacity (e.g. over engineer the ditch to a trapezoidal shape before allowing it become vegetated) and slow down water (e.g. control weirs), but herein systems used for treatment of nutrients in drainage waters are reviewed. In-ditch structures filled with filtration materials and the use of different adsorbents (Ezzati et al., 2019) have gained popularity as a cost-effective, easy to install and low-maintenance technology (Delgado and Berry, 2008; Bhatnagar and Sillanpaa, 2010; Bibi et al., 2015). To date, the performance of in-ditch

structures has mainly been evaluated in terms of the ability to treat a single contaminant, predominantly $NO_3^-$ (Addy et al., 2016; Christianson and Schipper, 2016) and dissolved reactive P (Penn et al., 2007). Recent studies have examined combining individual nutrient removal technologies (Goodwin et al., 2015; Ahnen et al., 2016; Gottschal et al., 2016) to treat both N and P (Ibrahim et al., 2015; Christianson et al., 2017). Although the present review focuses on nutrients in RR, ditches as a receptor of runoff can treat these nutrients and sediment either through natural or enhanced attenuation.

Engineered options target discharging ditch nutrients directly using reactive filters to remove a percentage of the nutrient load. For P treatment the filter contains P sorbing material (PSM), which can then be extracted once it becomes saturated (Penn and Bowen 2017). Saturation of the material will depend on the characteristics of the material (Ezzati et al. 2019). Fig.7. shows an example of an in-ditch structure used to remove a target DRP load in the USA. The installation of the pipes within in-ditch structures provides a gradient for water to flow through the media allowing removal rates of approximately 30%. Ideal site conditions for in-ditch systems should allow water to be channelled through a structure, at a gradient to an outlet, with upstream DRP concentrations in the region of 0.2-0.3 mg $L^{-1}$ (Penn and Bowen 2017) to enable good removal targets to be reached. Therefore, the highest concentrations of DRP should be targeted along a ditch which is likely to occur down gradient from the roadway discharge point. Seasonal filtered grab samples can be taken along a ditch to establish the DRP concentrations. If these are not available, estimates may be established from relationships between water extractible phosphorus and soil test P in adjoining fields. Denitrifying bioreactors are engineered structures filled with carbon sourced materials such as woodchip, straw, corn cob or compost and have been effectively used to convert $NO_3^-$ into di-nitrogen gas within ditch systems (Fig. 8). Research in this area has now

545 moved to treat both sediments and nutrients (N & P) simultaneously in such systems without
546 pollution swapping (Fenton et al., 2016).
547 Insert Fig 7
548 Insert Fig 8.
549
550 *3.1 New options* needing pilot scale testing
551 It is important that new farm roadway containment/mitigation measures are explored
552 and assessed with a view to their use as a viable and effective option for farmers. Such
553 measures should be suitable for retrofitting to existing roadways or installation in a new farm
554 roadway system. It is envisaged that such a level of intervention would only be required for a
555 small proportion of circumstances where simpler interventions are not possible due to site
556 specific limitations (roadway configuration relative to water courses, soil type, topography,
557 etc.). In this section two possibilities are presented. The first is a containment measure in the
558 form of a sedimentation tank. The overriding principle for this option is that the RR would be
559 routed through an underground two-compartment tank similar to a conventional passive
560 septic tank system. The first compartment would operate as a sedimentation chamber while
561 the second compartment would contain the supernatant liquid which would discharge via a
562 high level outlet pipe or opening (Fig 9). For this option the roadway could be constructed
563 with either a single or double cross sectional camber to route the flow into trapezoidal
564 channels running along its length. 'Grade breaks' may be incorporated at intervals along the
565 roadway, for example for longitudinal gradients > 5-10%, to prevent stream flow which may
566 result in channelling and erosion along the length of the roadway. A local depression at the
567 outlet with a concrete apron finish to the surface might be included in the design. The outlet
568 channel should be constructed to divert RR to the sedimentation tank and could also facilitate
569 an 'overflow' to the field, which would act as a visual indicator for the tank to be emptied. To

facilitate ease of emptying and slurry transport for spreading on suitable local land the tank would be located parallel to the roadway. The size of the sedimentation tank would depend on the length of roadway it serves and on the extent of anticipated pollutant load, which would vary depending on proximity to farm yards or holding areas for cattle, location of drinking troughs, junctions and field gates etc. (Oudshoorn et al., 2008, White et al., 2001). The separated liquid outflow from the sedimentation tank might be discharged to the adjacent field or dry ditch with sufficient nutrient attenuation capacity or alternatively to a swale downstream of the tank. While there might be an initial outlay associated with this type of system, one of the many perceived advantages is its low maintenance and capacity to contain valuable slurry solids in a fixed chamber of the tank which can be easily emptied as required.

The second alternative containment measure could be described as a bioretention area (Fig 10). This management option is similar to that of a sedimentation tank except that the tank is replaced with swales or bermed holding areas to attenuate the outflow and facilitate sedimentation. Whilst potentially relatively cheap to install, this system would require a larger land area and may require higher maintenance than the sedimentation tank system due to build-up of sediment over time. It may also not be as easy to clean out using vacuum slurry tankers. Additionally, the retention areas may lead to nutrient leaching through subsoil into groundwater (Murnane et al., 2018) unless impermeable barriers are incorporated into the design. On the other hand, the retention areas would act in an environmentally sustainable way, enhancing quality and promoting biodiversity similar to integrated constructed wetlands (Woods Ballard et al., 2015).

## *4.0 Discussion*

### *4.1 Local costs and considerations*

595 Any attempt at RR management must be considered as a long term management
596 objective which will necessitate a heightened level of total farm enterprise management
597 (USA ARS 2006). There are many diverse RR management options available which give
598 land owners flexibility. All management options involve costs and this is one of the many
599 complex considerations, which influences land owners' willingness to adopt best
600 management practices (BMPs) or voluntary measures (Bragina et al., 2019). Liu et al. (2018)
601 created a conceptual framework for BMP adoption decisions which highlights scale, the
602 tailoring or targeting of information and incentives and expected farm profits as being
603 important factors. Therefore, fair and equitable financial incentive schemes could be an
604 option to manage such an important pollutant loss pathway effectively.
605
606 In the present work RR management options were divided into on- and off-roadway
607 management options. The associated costs of such options will of course vary considerably
608 due to a host of variables (e.g. materials used, cost of labour, method of construction etc.).
609 Indeed, the level of after care and maintenance of conservation practices will also vary and
610 may be a deterrent to adoption (Welch et al., 2001; Tosakana et al., 2010). There have been
611 many projects and databases developed with BMPs to minimise nutrient loss to waterways
612 e.g. COST 869 Mitigation options for nutrient reduction in surface water and groundwater,
613 SERA17 (Innovative Solutions to Minimise Phosphorus From Agriculture) or Goeller et al.
614 (2020). These examine options which in some cases deal with runoff on an agricultural
615 landscape, but are not directly related to RR.
616 Insert Table 1.
617 The European Union Water Framework Directive (EU WFD, 2000/60/EC) requires
618 all Member States to protect and improve water quality in all waters. Member States must
619 prevent the deterioration of waters and all waters must achieve at least good status. Taking

Ireland as a case study, recent Irish Environmental Protection Agency (EPA) figures (2013-2018) reported 52.8% of surface water bodies in satisfactory ecological health; being in either 'Good' or 'High' ecological status. The remaining 47.2% of surface water bodies are in 'Moderate', 'Poor' or 'Bad' ecological status. This compares with 55.4% at satisfactory status for the last assessment period of 2010-2015, a decrease of 2.6% (EPA, 2019). The Nitrates Directive (ND) sets out a programme of measures relating to agriculture under the EU WFD. The ND aims to prevent pollution of waters from N and P arising from agricultural sources by ensuring good agricultural practices. Member States are required to implement a Nitrates Action Plan (NAP) and this is reviewed every 4 years. In Ireland, treatment of RR before discharge to a stream may require a licence under the Water Pollution Act (Local Government Water Pollution Act 1977). An example here is that discharges to a surface water body from integrated constructed wetlands which treat dairy soiled water on Irish farms already require a discharge licence. In Ireland, the onus is on the licence holder to demonstrate that the discharge to a surface water body is not causing water pollution. This would necessitate water testing at an accredited laboratory with an added level of cost and inspection that some landowners may not want. Interestingly, in Ireland new water protection measures within the NAP regulations include prevention of direct runoff from farm roadways to waters (S.I. No. 605 of 2017). **It should be noted that the definition for "waters" used in the regulations is defined as a) any (or any part of any) river, stream, lake, canal, reservoir, aquifer, pond, watercourse, or other inland waters, whether natural or artificial, b) any tidal waters, and c) where the context permits, any beach, river bank and salt marsh or other area which is contiguous to anything mentioned in paragraph (a) or (b), and the channel or bed of anything mentioned in paragraph (a) which is for the time being dry, but does not include a sewer. This policy instrument is effective from 1st January 2021. Therefore, in terms of the present review, options for RR may or may**

**not be applicable due to regulations. Table 1 contains all the mitigation measures applicable for RR but in an Irish context none of the measures that promote connectivity with waters are applicable. These may be relevant in the future where such regulations change.**

**This policy instrument is effective from 1st January 2021 and the most likely and immediate measures that will be most impactful are those that will break the connection between the source/pathway and the receptor (stream). The initial step in this process is to establish the pollution risk to surface waters from RR and this must be done in a methodical way, perhaps using a risk assessment tool that ranks a wide variety of parameters, such as farm road type, usage and characteristics, surface water proximity, soil type and its assimilative capacity, as well as the nature and frequency of precipitation. In that context a targeted approach is suggested and a hierarchical suite of potential RR management options could be considered. In Ireland, it is likely that the most widely applicable of these management options without incurring excessive costs will be use of adjacent fields to attenuate flows and assimilate nutrients such as N and P, SS, *E-coli* and emerging contaminants. Thus creation of a buffer zone (typically ≥ 1.5 m) between surface waters and farm roadways in combination with re-cambering the roadway so that runoff will be directed away from the stream are two basic measures that should contribute to this objective. In order for this to remain effective however it is important that the buffer zone is maintained for its purpose by stock proof fencing. Other relatively straightforward measures that would be beneficial are the removal of obstructions to animal movement along the farm roadway (e.g. sharp bends, drinking troughs, inadequate width and inappropriately located gateways) in order to reduce the time spend by animals travelling the road and hence the source pollution load. Additional to these measures, incorporation of grade breaks along steep lengths of the**

**roadways may also enhance protection by directing flow to diffuse sections of adjacent fields thereby mobilising a greater natural treatment area (i.e. soil) than would be the case if all the RR were discharged at a single location off the farm roadway. There may be cases however where all of these relatively straightforward measures may not be feasible, for example in situations where the capacity of the soil is insufficient for either flow or load attenuation, or the measures are simply insufficient to achieve the required outcome (i.e. reduce the source pollution load and break the connectivity). In such cases, depending on the situation, more complex and expensive management options may be required, ranging from on-roadway measures such as buffer zones, earth bunding, overflow structures and application of chemical amendments to off-roadway measures such as in-ditch treatments or engineered attenuation measures (Table 1), while in extreme cases roadway relocation may be required. In practical terms however, it is likely that low maintenance and visible measures will be initially favoured by both farmers and the regulation authorities in Ireland. It is important to recognise however that each site is unique and therefore generalisation of RR management options to all farms would be a misguided approach where a strategy of "right place, right management option" should instead be followed.**

*4.2 Future research needs*

Although acknowledged as a pollutant source there is a surprisingly little literature with respect to testing and costing of on- or off- roadway management options. There are examples which can be inferred from other land uses such as forested areas around the world (e.g. Grayson et al., 1993; Laurance et al., 2009; Lane and Sheridan 2002; Hill and Pickering 2009).

Future research in Ireland should focus on field work and high resolution data collection. Pilot scale setups could intercept RR on a flow weighted basis thereby providing spatial and temporal characterisation data. As farms vary considerably due to geographic location (soil type and rainfall distribution), stocking rate and infrastructure, datasets should be built that can compare and contrast systems. For example, in a dairy management system decisions may be quite different into the future as many transition to an automated system or change the traditional routines i.e. once-twice-thrice a day milking. Such dairy systems would all have different knock on effects regards animal access to roadways, time spent on the roadway network, volume of traffic and associated defecation. A robotic milking approach enables a "semi-forced" or "free cow traffic" system which has implications for farm roadway defecation. Furthermore, for many reasons, livestock farming systems are continuously changing in terms of intensity and expansion/contraction of animal numbers.

There is also a need to consider RR in the context of a changing climate. Wetting and drying patterns on farms are changing and this affects a soils' capacity to store, transit and attenuate pollutants. Such changes will also have an effect on the spatial and temporal mobilisation and connectivity of RR. In times of flooding and drought, farm roadways may be negatively affected in terms of quality and on/off management options designed for normal conditions are likely to suffer functional failure when encountering extremes (e.g 1 in 100 year events or higher). Development of a farm roadway classification system which can be used to divide farm roadway networks into prescribed sections with an associated risk in terms of nutrient, sediment and *E. coli* losses in RR is needed. As an added layer a visual tool to describe the quality of individual roadway sections should also be developed as this could be used as a parsimonious tool for land owners or advisors to quickly identify roadway sections in need of intervention. These sections could then be matched with management options proposed herein. As lameness in cows is a significant animal welfare concern,

719 research on hoof health with an emphasis on the role of farm roadways both in terms of
720 runoff and surface quality is important.
721
722 **5. Conclusions**
723 Farm RR can contain nutrients, sediment and pathogens akin to dairy soiled water.
724 Farm roadways in close proximity to farmyards have the highest concentration of pollutants.
725 Such sources are temporarily mobilised and transported as runoff during rainfall events and
726 may discharge to fields, into drainage ditches or onto public roadways or watercourses.
727 Options to prevent direct or indirect runoff to waters may include installation of on- or off-
728 roadway options. For example, diversion structures at regular intervals along steep roadway
729 sections can discharge runoff to a field or a drainage ditch where its contents can be naturally
730 or artificially retained, rerouted and in some cases be treated. As every site is unique a
731 strategy of "right place, right management option" should be followed. Such a management
732 option must follow local regulatory frameworks. **This is especially important in Ireland**
733 **where all the available options to manage roadway runoff are not allowed under present**
734 **regulations.** Future work should concentrate on impacts of climate change, varied farming
735 systems and intensity, classification of roadway networks, alternative management option
736 development and providing a roadway risk assessment tool to enable landowners and
737 advisors to identify points of connectivity where roadway runoff enters waters.
738
739 **Acknowledgement**
740 This project 5173: Roadway Runoff and Nutrient-loss Reduction was co-funded by
741 EPA/DAFM. The authors would also like to thank those contributors for photos and to those
742 who provided local information in Ireland, UK and USA.
743

## References


Adams, R., Arafat, Y., Eate, V., Grace, M.R., Saffarpour, Sh., Weatherley, A.J., Western, A.W., 2014. A catchment study of sources and sinks of nutrients and sediments in south-east Australia. J. Of Hydrol. 515, 166-179.

Akbarimehr, M., Naghdi, R. Reduction of erosion from forest roads and skid trails by management practices. J Forest Sci., 58, 165-169.

Aland, A., Lidfors, L., Elesbo I., 2002. Diurnal distribution of dairy cow defecation and urination. Appl. Anim. Behav. Sci. 78, 43-54.

Bjerg, P.L., Albrechtsen, H.J., Kjeldsen, P., Christensen, T.H., Cozzarelli., 2003. The groundwater geochemistry of waste disposal facilities. Treatise on Geochem. 9, 579-612.

Biggs, J., von Fumetti, S., Kelly-Quinn, M., 2017. The importance of small waterbodies for biodiversity and ecosystem services: implications for policy makers. Hydrobiologia,793, 3-39.

Bragina L, Micha E, Roberts WM, O'Connell K, O'Donoghue C, Ryan M, Daly K., 2019. Spatial and temporal variability in costs and effectiveness in phosphorus loss mitigation at farm scale: A scenario analysis. J. Environ. Manage. 245, 330-337.

Brennan, R.B., Fenton, O., Rodgers, M., Healy, M.G., 2011. Evaluation of chemical amendments to control phosphorus losses from dairy slurry. Soil Use and Manage., 27, 238-246.

**Buchanan, B.P., Archibald, J.A., Easton, Z.M., Shaw, S.B., Schneider, R.L., Walter, T.A. 2013a. A phosphorus index that combines critical source areas and transport pathways using a travel time approach. Journal of Hydrology, 486, 123-135.**

**Buchanan, B.P., Falbo, K., Schneider, R.L., Easton, Z. M., Walter, M.T. 2013b.Hydrological impact of roadside ditches in an agricultural watershed in Central New York: implications for non-point source pollutant transport. Hydrological Processes, 27, 2422-2437.**

Burchill, W., Reville, F., Misselbrook, T.H., O'Connell, C., Lanigan, G.J., 2019. Ammonia emissions and mitigation from a concrete yard used by cattle. Biosyst Eng. 184, 181-189.

Chase, E.H., Schneider, R.L., Baker, N.J., Dunn, S.J., 2019. Retrofitting the Rural Roadside Ditch Network to Treat Nitrogen from Agricultural Runoff Using Woodchip Bioreactors. 12th International Conference on Low-volume Roads. Transportation Research Circular September 15-18, Montana, USA. Drainage and Stream Crossings Section, pg 320-324.

Chibuike, G., Burkitt, L., Bretherton, M., Camps-Arbestain, M., Singh, R., Bishop, P., Hedley, C. & Roudier, P., 2019. Dissolved organic carbon concentration and denitrification capacity of a hill country sub-catchment as affected by soil type and slope. New Zealand J. Ag. Research 62(3), 354-368.

Clagnan, E., Thornton, S.F., Rolfe, S.A., Tuohy, P., Peyton, D., Wells, N.S., Fenton, O., 2018 Influence of artificial drainage system design on the nitrogen attenuation potential of gley soils: Evidence from hydrochemical and isotope studies under field-scale conditions. 2018. J. Environ Manage. 206, 1028-1038.

Clagnan, E., Thornton, S.F., Rolfe, S.A., Wells, N.S., Knoeller, K., Murphy, J., Tuohy, P., Daly, K., Healy, M.G., Ezzati, G., von Chamier, J., Fenton, O., 2019. An integrated assessment of nitrogen source, transformation and fate within an intensive dairy system to inform management change. PLoS One 14, 7, e0219479

Clarke, P. 2016. Chapter 21: Infrastructure. In Dairy Farm Manual, Section 8: Cattle Health. Teagasc.

Collins, A.L., Zhang, Y., Walling, D.E., Grenfell, S.E., Smith, P. 2010. Tracing sediment loss from eroding farm tracks using a geochemical fingerprinting procedure combining local and genetic algorithm optimisation. Sci. Tot. Environ. 48, 5461-5471.

Coyle, C., Creamer, R.E., Schulte, R.P.O., Sullivan, L.O., Jordan, P., 2016. A Functional Land Management conceptual framework under soil drainage and land use scenarios. Environ. Sci. Pol. 56, 39-48.

Creamer, R., O'Sullivan, L. 2018. The Soils of Ireland. World Soils Book Series. Springer International Publishing AG, DOI: 10.1007/978-3-319-71189-8

Daly, K., Styles, D., Lalor, S., Wall, D.P., 2015. Phosphorus sorption, supply potential and availability in soils with contrasting parent material and soil chemical properties. European J. Soil Sci. 66, 792-801.

Daly, K., Tuohy, P., Peyton, D., Wall, D.P., Fenton, O. 2017. Field soil and ditch sediment phosphorus dynamics from two artificially drained fields on poorly drained soils. Ag. Water Manage. 192, 115-125.

Deakin, J., Flynn, R., Archbold, M., Daly, D., O'Brien, R., Orr, A., Misstear, B., 2016. Understanding pathways transferring nutrients to streams: review of a major Irish study and its implications for determining water quality management strategies. Biol. Environ.: Proceedings of the Royal Irish Academy Vol. 116B (3), 233-243.

DGVLR, Dirt Gravel and Low Volume Roads, Center for Dirt and gravel roads studies, Penn State University, USA. Technical Bulletins (2019). Accessed on 13.02.20 at https://www.dirtandgravel.psu.edu/general-resources/technical-bulletins.

Dollinger, J., Cécile Dagès, Jean-Stéphane Bailly, Philippe Lagacherie, Marc Voltz. 2015. Managing ditches for agroecological engineering of landscape. A review. Agronomy for Sustainable Development, 35 (3), 999-1020.

Doody, D., Cross, P., Withers, P., Cassidy, R., Augustenborg, C., Pullin, A., Carton, O., Crosse, S. 2019. EPA Research Report (2013-W-DS-13). AgImpact Project: A Systematic and Participatory Review of Research on the Impact of Agriculture on Aquatic Ecosystems in Ireland. Report NO 194. www.epa.ie

**Duke, G.D., Kienzle, S.W., Johnson, D.L., Byrne, J.M. 2003. Improving overland flow routing by incorporating ancillary road data into Digital Elevation Models. Journal of Spatial Hydrology, Vol.3, No.2 Fall 2003.**

**Easton, Z.M., Gerard-Marchant, P., Walter, M.T., Petrovic, A.M., Steenhuis, T.S., 2007. Identifying dissolved phosphorus source areas and predicting transport from an urban watershed using distributed hydrologic modeling. Water Resour. Res. 43 (11).**

**Easton, Z.M., Walter, M.T., Steenhuis, T.S., 2008. Combined monitoring and modeling indicate the most effective agricultural best management practices. J. Environ. Qual. 37 (5), 1798–1809.**

**Endreny, T.A., Wood, E.F., 2003. Watershed weighting of export coefficients to map critical phosphorus loading areas. J. Am. Water Resour. Assoc. 39, 165–181.**

Ezzati, G., Healy, M.G., Christianson, L., Feyereisen, G.W., Thorton, S., Daly, K., Fenton, O., 2019. Developing and validating a decision support tool for media selection to mitigate drainage waters. Ecol. Eng. X2.

Ezzati, G., Fenton, O., Healy, M.G., Christianson, L., Feyereisen, G., Thornton, S., Chen, Q., Fan, B., Ding, J., Daly, K., 2020. Impact of phosphorus inputs on the source-sink dynamics of sediment along an agricultural ditch network. J. Environ. Manage. 257 (109988).

Fenton, O., Healy, M.G., Brennan, F.P., Thornton, S.F., Lanigan, G.J., Ibrahim, T.G., 2016. Holistic evaluation of field-scale denitrifying bioreactors as a basis to improve environmental sustainability. J. Environ. Qual., 4, 788-795.

Galloway, B.M., Schiller, R.E., Rose, J.G., 1971. The effects of rainfall intensity, pavement cross slope, surface texture, and drainage length on pavement water depths. Research Report 138-5. Texas Transport Institute, Texas A&M University, College Station, Texas. Accessed on 13.02.20 at https://static.tti.tamu.edu/tti.tamu.edu/documents/138-5.pdf

Goeller, B.C., Febria, C.M., McKergow, L.A., Harding, J.S., Matheson, F.E., Tanner, C.C., McIntosh, A.R., 2020. Combining Tools from Edge-of-Field to In-Stream to Attenuate Reactive Nitrogen along Small Agricultural Waterways. Water, 12, 383.

Gonzalez Jimenez, J. L., Healy, M. G., Roberts, W. M. and Daly, K.,2018. Contrasting yield responses to phosphorus applications on mineral and organic soils from extensively managed grasslands: Implications for P management in high ecological status catchments. *J.* Plant Nutr. Soil Sci. 1-9.

González Jiménez, J. L., Healy, M. G., and Daly, K.,2019. Effects of fertiliser on phosphorus pools in soils with contrasting organic matter content: A fractionation and path analysis study. Geoderma 338, 128-135.

Grayson, R.B., Haydon, S.R., Jayasuriya, M.D.A., Finlayson, B.L. 1993. Water quality in mountain ash forests – separating the impacts of roads from those of logging operations. Journal of Hydrology, 150 (2-4), 459-480.

Haygarth, P.M., Condron, L.M., Heathwaite, A.L., Turner, B.L., Harris, G.P., 2005. The phosphorus transfer continuum: Linking source to impact with an interdisciplinary and multi-scaled approach. Sci. Tot. Environ., 344, 5-14.

Hill, W., Pickering, C.M. 2009. Evaluation of impacts and methods for the assessment of walking tracks in protected areas. Cooperative Research Centre for Sustainable Tourism.

Hively, W.D., Gérard-Marchant, P. & 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.

Jennings, E., Allott, N., Lenihan, D., Quirke, B., Taylor, D., Twomey, H., 2013. Drivers of long-term trends and seasonal changes in total phosphorus loads to a mesotrophic lake in the west of Ireland. Marine and Freshwater Research 64, 413-422.

Kröger, R., Holland, M.M., Moore, M.T., Cooper, C.M., 2007. Hydrological variability and agricultural drainage ditch inorganic nitrogen reduction capacity. J. Environ. Qual, 36, 1646-1652.

Kröger, R., Holland, M.M., Moore, M.T., Cooper, C.M., 2008. Agricultural drainage ditches mitigate phosphorus loads as a function of hydrological variability. J. Environ. Qual. 37, 107-113.

Kröger, R., Dunne, E.J., Noval, J., King, K.W., Mclellan, E., Smith, D.R., Strock, J., Boomer, K., Tomer, M.D., Now, G.B., 2012. Downstream Approaches to Phosphorus Management in Agricultural Landscapes: Regional Applicability and Use. Sci. Tot. Environ. 442, 263-274.

Lane, P.N., Sheridan, G.J. 2002. Impact of an unsealed forest road stream crossing: water quality and sediment sources. Hydrological Processes, 16, 2599-2612.

Laurance, W.F., Goosem, M., Laurence, S.G. 2009. Impacts of roads and linear clearings on tropical forests. Trends in Ecology and Evolution, 24, 659-669.

LEAF 2010. SUDS – Sustainable Drainage Systems to explore the effectiveness of different pathway options in slowing down the flow of surface run-off and trapping sediment from different farm and field locations. https://archive.leafuk.org/eblock/services/resources.ashx/000/557/755/LEAF_EA_SuDS_report_final2.pdf

Ledgard, S.F., Penno, J.W. and Sprosen, M.S., 1999. Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. J. Ag. Sci. 132, 215-225.

Lucci, G.M., McDowell, R.W., Condron, L.M., 2010. Potential phosphorus and sediment loads from sources within a dairy farmed catchment. Soil Use and Manage. 26, 44-52.

Lucci, G.M., McDowell, R.W., Condron, L.M., 2012. Phosphorus source areas in a dairy catchment in Otago, New Zealand. Soil Research 50, 145-156.

Liljaniemi, P., Vuori, K.-M., Tossavainen, T., Kotanen, J., Haapanen, M., Lepistö, A., Kenttämies, K., 2003. Effectiveness of constructed overland flow areas in decreasing diffuse pollution from forest drainages. Environ. Manage. 32, 602-613.

Liu, T., Bruins, R.J., Herberling, M.T., 2018. Factors Influencing Farmers' Adoption of Best Management Practices: A Review and Synthesis. Sustain, 10, 432.

Majsztrik, J.C., Fernandez, R.T., Fisher, P.R., Hitchcock, D.R., Lee-Cox, J., Owen, J.S. Jr., Oki, L.R., White. S.A., 2017. Water use and treatment in container-grown specialty crop production: A review. Water Air Soil Pol., 228.

McDowell, R.W, 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. J. Environ. Qual. 41, 680-693.

McDowell, R.W., Norris, M., 2014. The use of alum to decrease phosphorus losses in runoff from grassland soils. J. Environ. Qual. 43, 1635-1643.

McDowell, R.W., Daly, K., Fenton, O. 2020. Mitigation of phosphorus, sediment and Escherichia coli losses in runoff from a dairy farm roadway. Irish J. Food and Agri. Sci., DOI 10.15212/ijafr-2020-0117.

McLaren, T., Guppy, C., Tighe, M., Moody, P., Bell, M. 2014., Dilute acid extraction is a 505 useful indicator of the supply of slowly available phosphorus in Vertisols. Soil Science 506 Society of America Journal, **78**, 139-146

McLaughlin, M.J. McBeath, T.M., Smernik, R., Stacey, S.P., Babasola A., Guppy, C., 2011. The chemical nature of P accumulation in agricultural soils – implications for fertiliser management and design: an Australian perspective. Plant Soil 349: 69-87.

Mellander, P-E., Jordan, P., Melland,A.R., Murphy, P.N.C., Wall, D.P., Mechan, S., Meehan, R., Kelly, C., Shine, O., Shortle, G., 2013. Quantification of Phosphorus Transport from a Karstic Agricultural Watershed to Emerging Spring Water. Environ. Sci. Technol. 47, 6111-6119.

**Ministry for the Environment, A guide to sustainable water and riparian management in rural New Zealand. Accessed at https://www.mfe.govt.nz/sites/default/files/managing-waterways-jul01.pdf on 13th January 2021**

Monaghan, R.M., Smith, L.C., 2012. Contaminant losses in overland flow from dairy farm laneways in southern New Zealand. Ag. Ecosyst. Environ. 159, 170-175.

Moore M.T, Kröger R, Locke M.A, Cullum R.F, Steinriede R.W Jr, Testa S III, Lizotte R.E Jr, Bryant C.T, Cooper C.M., 2010. Nutrient mitigation capacity in Mississippi Delta, USA drainage ditches. Environ. Pollut. 158, 175-184.

Mulligan, C.N., Yong, R.N., 2004. Natural attenuation of contaminated soils. Environ. International, 30, 587-601.

Murnane, J.G., Brennan, R.B., Healy, M.G., Fenton O., 2015. Use of zeolite with alum and polyaluminum chloride amendments to mitigate runoff losses of phosphorus, nitrogen, and suspended solids from agricultural wastes applied to grassed soils. J. Env. Qual. 44(5), 1674-1683.

Murnane, J.G., Fenton, O., Healy, M.G., 2018. Impacts of zeolite, alum and polyaluminum chloride amendments mixed with agricultural wastes on soil column leachate, and $CO_2$ and $CH_4$ emissions. J. Env. Mgt., 206, 398-408.

Nguyen L, Nagels J, Sukias JPS 2002. Faecal contamination and the removal of Escherichia coli (E. coli) in drainage ditches. In: Currie LD, Loganathan P ed. Dairy farm soil management. Fertilizer and Lime Research Centre, Massey University, Palmerston North. Occasional Report No. 15. Pp. 261-266

Nguyen, L., Sukias, J., 2002. Phosphorus fractions and retention in drainage ditch sediments receiving surface runoff and subsurface drainage from agricultural catchments in the North Island, New Zealand. Agric. Ecosyst. Environ. 92, 49-69.

O'Flynn, C.J., Fenton, O., Healy M.G., 2012. Evaluation of amendments to control phosphorus losses in runoff from pig slurry applications to land. Clean – Soil, Air, Wat, 40, 164-170.

Oudshoorn, F.W., Kristensen, T., Nadimi, N.S., 2008. Dairy cow defecation and urination frequency and spatial distribution in relation to time-limited grazing. Livest. Sci, 113, 62-73.

Outram, F.N., Cooper, R.J., Sunnenberg, G., Hiscock, K.M., Lovett, A.A., 2016. Antecedent conditions, hydrological connectivity and anthropogenic inputs: Factors affecting nitrate and phosphorus transfers to agricultural headwater streams. Sci. Tot Environ, 545-546, 184-199.

Penn, C.J., Bryant, R.B., Kleinman, P.J.A., Allen, A.L., 2007. Removing dissolved phosphorus from drainage ditch water with phosphorus sorbing materials. J. Soil Water Conserv. 62, 269-276.

Penn, C.J., Bowen, J.M., 2017a. Chapter 3: Phosphorus removal structures as a short-term solution for the problem of dissolve phosphorus transport to surface waters. In: Design and Construction of Phosphorus Removal Structures for Improving Water Quality. Springer International.

Penn, P., Chagas, I., Klimeski, A., Lyngsie G., 2017b. A review of phosphorus removal structures: How to assess and compare their performance. Water, 9, 583.

Rice, P., O'Brien, D., Shalloo, L., Holden, N.M., 2020. The influence of dairy cow breed on the environmental impact per unit of milk produced. Animal (*Under Review*).

Pluer, W.T., Geohring, L.D., Steenhuis, T.S., Walter, M.T., 2016. Controls influencing the treatment of excess agricultural nitrate with denitrifying bioreactors. J. Environ. Qual. 45, 772-778.

Powell, J.M., MacLeod, M., Vellinga, T.V., Opio, C., Falcucci, A., Tempio, G., Steinfeld, H. and Gerber, P., 2013. Feed–milk–manure nitrogen relationships in global dairy production systems. Livestock Science, 152(2), pp.261-272.

**Pratt, J.P., Macintosh, K.A., Brown, L., Burkitt, L. 2020. Effect of riparian widths for reducing contaminants from dairy-farm laneways. Farmed Landscape Research Centre Workshop, Massey University, University of New Zealand.**

Roberts, W. M., Gonzalez-Jimenez, J. L., Doody, D. G., Jordan, P., and Daly, K. (2017). Assessing the risk of phosphorus transfer to high ecological status rivers: Integration of nutrient management with soil geochemical and hydrological conditions. Sci. Tot. Environ. 589, 25-35.

Sandström, S., Futter, M.N., Kyllmar, K., Bishop, K., O'Connell, D.W., Djodjic, F., 2020. Particulate phosphorus and suspended solids losses from small agricultural catchments: Links to stream and catchment characteristics. Sci. Tot Environ., 711, 134616.

Schipper, L.A., Robertson, W.D., Gold, A.J., Jaynes, D.B., Cameron, S.C., 2010. Denitrifying bioreactors-an approach for reducing nitrate loads to receiving waters. Ecol. Eng. 36, 1532-1543.

Scheetz, B.E., Bloser, S.M., 2009. Environmentally sensitive maintenance on agricultural roads to reduce nutrient and sediment pollution in the Kishacoquillas watershed. U.S. Department of Agriculture. Conservation innovation grant program. The Pennsylvania State University. Accessed on 13.02.20 at
https://www.dirtandgravel.psu.edu/sites/default/files/Center/Research/CIG_Final_Report.pdf

Sharpley, A.N., Bergström, L., Aronsson, H., Bechmann, M., Bolster, Börling, K., Djodjic, F., Jarvie, H.P., Schoumans, O.F., Stamm, C., Tonderski, K.S., Ulén, B., Uusitalo, R., Withers, P.J.A., 2015. Future agriculture with minimized phosphorus losses to waters: Research needs and direction. AMBIO 44, 163-179.

Sharrer, K.L., Christianson, L.E., Lepine, C., Summerfelt, S.T., 2016, Modelling and mitigation of denitrification 'woodchip' bioreactor phosphorus releases during treatment of aquaculture wastewater. Ecol. Eng, 93, 135-143.

Sherriff, S.C., Rowan, J.S., Melland, A.R., Jordan, P., Fenton, O., O'hUallachain, D.O., 2015. Investigating suspended sediment dynamics in contrasting agricultural catchments using ex situ turbidity-based suspended sediment monitoring. Hydrol. Earth Syst. Sci. 19 (8). 3349-3363.

Shore, M., Murphy, P.N.C., Jordan, P., Mellander, P-E., Kelly-Quinn, M., Cushen, M., Mechan, S., Shine, O., Melland, AR.,2013. Evaluation of a surface hydrological connectivity index in agricultural catchments. J. Environ. Model Softw., 47, 7-15

Shore, M., Jordan, P., Mellander, P-E., Kelly-Quinn, M., Melland, A.R., 2015a. An agricultural drainage channel classification system for phosphorus management. Agric. Ecosyst. Environ., 199, 207-215.

Shore, M., Jordan, P., Mellander, P-E., Kelly-Quinn, M., Daly, K., Sims, J.T., Wall, D.P., Melland, A.R., 2015b. Characterisation of agricultural drainage ditch sediments along the

phosphorus transfer continuum in two contrasting headwater catchments. J. Soils and Sediments., 16(5), 1643-1654.

Smith, D.R., Pappas, E.A.2007. Effect of ditch dredging on the fate of nutrients in deep drainage ditches of the Midwestern United States. J. Soil Water Conserv., 62, 252-261.

Smith, L.C., McDowell, R.W., 2016. The use of alum to decrease phosphorus loss from dairy farm laneways in southern New Zealand. Soil Use Manage., 6(32), 69-71.

**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, New Zealand. Journal of Hydrology, 379, 53-67.**

STT1, 2018. Land Incentive Scheme Handbook. Source to tap, farming for water. Interreg (Northern Ireland – Ireland – Scotland). Accessed on 13.02.20 at https://www.sourcetotap.eu/wp-content/uploads/2018/07/STT1-Land-Incentive-Scheme-Handbook-1.pdf

Teagasc. 2017. Dairy Farm Infrastructure Handbook. Animal and Grassland Research and Innovation Centre, Moorepark. https://www.teagasc.ie/media/website/publications/2017/Dairy-Farm-Infrastructure-Handbook-Moorepark2017-(V3).pdf

Thomas, I.A., Jordan, P., Mellander, P.E., Fenton, O., Shine, O., hUallacháin, D Ó., 2016. Improving the identification of hydrologically sensitive areas using LiDAR DEMs for the delineation and mitigation of critical source areas of diffuse pollution. Sci. Tot. Environ., 556, 276-290.

Tosakana, N, Van Tassel,l L, Wulfhorst, J, Boll, J, Mahler, R, Brooks, E, Kane, S., 2010. Determinants of the adoption of conservation practices by farmers in the Northwest Wheat and Range Region. J Soil Water Conserv., 65, 404-412.

**UK Environment Agency, Guidance: Rules for farmers and land managers to prevent water pollution. Accessed at https://www.gov.uk/guidance/rules-for-farmers-and-land-managers-to-prevent-water-pollution#prevent-erosion-manage-livestock-and-soil on 13th January 2021.**

Villettaz Robichaud, M., de Passillé, A.M., Pellerinand, D., Rushen, J., 2011. When and where do dairy cows defecate and urinate? J. Dairy Sci., 94 (10), 4889-4896.

Wall, D.P, O'Sullivan, L., Debeljak, M., Trajanov, A., Schroder, J., Bugge Henriksen, C., Creamer, R.E., Cacovean, H., Delgado, A., 2017. Key indicators and management strategies for water purification and regulation. Technical Public Report D3.2, Landmark Project www.landmark2020.eu.

Welch, E.W., Frederick, J., Marc-Aurele Jr., P.E., 2001. Determinants of farmer behaviour: adoption of and compliance with best management practices for nonpoint source pollution in the Skaneateles Lake Watershed. Lake and Reservoir Manage., 17:3, 233-245.

Wemple, B.C., 2013. Assessing the effect of unpaved roads on Lake Champlain water quality. Final Report. June 2013. Lake Champlain Basin Program and New England Interstate Water Pollution Control Commission.

Wemple, B.C., Jones, J.A., 2003. Runoff production on forest roads in a steep, mountain catchment. Water Resources Research, 39(8), 1220.

White, S.L., Sheffield, R.E., Washburn, S.P., King, L.D., Green, J.T., 2001. Spatial and time distribution of dairy cattle excreta in an intensive pasture system. J. Environ Qual., 30, 2180-2187.

Wilkinson, M.E., Quinn, P.F., Barber, N.J., Jonczyk, J., 2014. framework for managing runoff and pollution in the rural landscape using a catchment systems engineering approach. Sci. Tot. Environ., 468-469, 1245-1254.

Woods Ballard, B., Wilson, S., Udale-Clarke, H., Illman, S., Scott, T., Ashley, R., Kellagher, R., (2015). The SuDS Manual. CIRIA C753, Griffin Court, 15 Long Lane, London, EC1A 9PN, UK. Accessed on 13.02.20 at http://www.hrwallingford.com.cn/pdfs/news/CIRIA%20report%20C753%20The%20SuDS%20Manual-v2.pdf

**Wu, J.S., Allan, C.J., Saunders, W.L., Evett, J.B., 1998. Characterization and pollutant loading estimation for highway runoff. J. Environ. Eng.-ASCE 124 (7), 584–592.**

## Captions for Figures

**Fig.1.** Schematic illustration of surface flow before and after installation of grade breaks (adapted from DGVLR, 2019 – Grade Break);

**Fig 2.** Conveyor belt diversion (Credit: DGVLR, 2019 – Conveyor Belt Diversion)

**Fig 3**.Open channel flow diversion on upgraded roadway (Credit: STT1, 2018)

**Fig 4**. Deep open channel flow diversion in N .Ireland. Open channel flow which is suitable in a scenario where machinery and not animals are trafficking a roadway section (25 cm deep, 10 cm wide) (Hillsborough farm, Northern Ireland Credit: Owen Fenton)

**Fig.5.** In ditch silt/sediment traps installed in freshly cleaned drains designed to drop sediment, carbon and some particulate nutrients from suspension (Example here is from Kilberrihert LIFE09 NAT/IE/000220 BLACKWATER SAMOK, Ireland). These also act as retention structures and slow the flow in the ditch system. Sediment can be removed in a removal bag or tray (Credit: Fran Igoe).

**Fig 6.** LEAF, UK examples of in-ditch retention structures to slow flow and reduce downstream sediment and nutrient loads (Credit: Paul Quinn, Newcastle).

**Fig.7.** A small drainage ditch filter developed at University Park, PA, USA. Left: Finished structure. Right: Steel slag infill to trap DRP (Credit: Gary Feyereisen, USDA).

**Fig 8.** Left: In-ditch bioreactor in Livingston County, Illinois, USA; looking up-stream during construction. Right: In-ditch bioreactor in Maryland, USA; looking upstream (Credit: Laura Christianson, UIUC).

**Fig.9.** Schematic layout of sedimentation tank system as a possible road runoff containment measure.

**Fig.10.** Schematic layout of bioretention area as a possible road runoff management measure.

**Captions for Tables**

**Table 1.** Explores each of the options mentioned in this review and documents the costs that appear in the related publication and potential positive and negative aspects to consider. It should be noted that as with all cases site conditions will dictate final costs and effectiveness.