Effectiveness of Vegetated Buffer Strips in Controlling Legacy Phosphorus Exports from Agricultural Land

Reza Habibiandehkordi,* David A. Lobb, Philip N. Owens, and Don N. Flaten

Abstract
The continued phosphorus (P) impairment of freshwaters and the associated risk of eutrophication raise questions regarding the efficiency of current beneficial management practices (BMPs) for improving water quality. Vegetated buffer strips (VBSs) are widely encouraged BMPs for reducing P export from agricultural land. However, there is a lack of evidence regarding the long-term efficiency of VBSs for reducing legacy P losses. This research used soil analyses to investigate the P removal efficiency of an unmanaged VBS for controlling P loss from agricultural land in Manitoba, Canada, between 1954 and 2011. The results showed statistically significant retention of total P, Olsen extractable P, and 0.01 M CaCl$_2$ extractable P by a 5-m wide VBS compared with field soils. We found that surface soils at 5 m into the VBS had a significantly greater P sorption capacity and a smaller degree of P saturation (DPS) than adjacent field soils. The elevated DPS in field soils is likely associated with gradual P enrichment as a result of manure or fertilizer application over time and the strong affinity of P compounds for soil. Although P stratification in the VBS over 57 yr resulted in a significant increase (~11%) in DPS of VBS topsoil compared with VBS subsoil, our findings do not support the saturation of VBS soils with P. However, cutting and removal of vegetation from VBS could be a useful strategy to remove P from VBS and minimize possible P remobilization associated with vegetation senescence, especially where the climate is cold and runoff is dominated by snowmelt.

Core Ideas
• Vegetated buffer strips (VBSs) reduced legacy P losses from adjacent land.
• VBS soils had a significantly greater P sorption capacity than field soils.
• VBS soils had a significantly smaller degree of P saturation than field soils.
• VBS soils did not get saturated with P exports from adjacent land even after 57 yr.

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NITROGEN and phosphorus (P) inputs to agricultural land are required to ensure adequate levels of crop production to meet the food and fiber demands of a growing population. However, the transfer of surplus nutrients from agricultural land to water bodies following rainfall, irrigation, and snowmelt events can contribute to water quality impairment, resulting in the growth of extensive algal blooms, changes in community composition, and reductions in biodiversity within aquatic ecosystems (Smith and Schindler, 2009; Watson et al., 2016). Unlike nitrogen, P has a long residence time in soil due to the strong affinity of P compounds for soil and the lack of a significant gaseous pathway for P removal (Habibiandehkordi et al., 2015a). Despite significant efforts to reduce P export from agricultural land through balancing P inputs with crop requirements and implementing edge-of-field beneficial management practices (BMPs), little or no significant improvement in downstream water quality has been reported in many case studies (Meals et al., 2010; Sharpley et al., 2015). The lack of evidence for water quality improvement could be associated with a number of factors such as errors in implementing appropriate BMPs to reduce P losses, the lack of landowners’ full commitment to establish and maintain BMPs, climate variability, and the lag time in water quality response to BMPs which can be years to several decades (Meals et al., 2010; Sharpley et al., 2015). Continued water quality impairment has been reported for some of the world’s largest lakes, including Lake Winnipeg in Manitoba, Canada, where P concentrations and algae blooms have doubled since the mid-1990s (Schindler et al., 2012). This highlights the need to reconsider the efficiency of BMPs for tackling diffuse P pollution to understand whether existing BMPs are ineffective, or if more time is needed for achieving water quality targets (Jarvie et al., 2013).

The “P legacy” effect is hypothesized to be a primary reason for the delay in water quality response to BMPs and is associated with the prolonged accumulation of P in soils, as well as the complex pathways and forms of P loss in a catchment (Haygarth...
et al., 2012, 2014; Sharpley et al., 2013). For example, P loss generally occurs in certain areas of a catchment where sources with a large P content (e.g., land gradually enriched with P from past nutrient management) coincide with hydrological flows across the catchment (Sharpley et al., 2011). Such critical source areas, or “hotspots,” are spatially and temporally varied as, for example, in the Canadian prairies, where a large proportion of the entire landscape may generate runoff and P loss during snowmelt in the spring, whereas the critical source areas could be limited to areas close to streams during rainfall–runoff events in summer. As a result, fluctuations in P release from certain pools in a catchment have the potential to buffer or mask the actual impact of a BMP on P losses (Sharpley et al., 2013) because the performance of edge-of-field BMPs is varied depending on the concentrations and loads of P entering the BMP. Therefore, a greater insight into the complex response of BMPs to multiple and dynamic patterns of legacy P exports from agricultural land is crucial to design and implement practical and cost-effective mitigation strategies for water quality protection (Jarvie et al., 2013; Sharpley et al., 2015; Habibiandehkordi et al., 2017).

Among edge-of-field BMPs to reduce P exports from agricultural land, vegetated buffer strips (VBSs) are widely used in Europe, North America, and elsewhere, partly due to their relative ease of establishment and arguably their small footprints and maintenance costs (Hickey and Doran, 2004; Hussein et al., 2007; Habibiandehkordi et al., 2015a). Although VBSs appeared to be less effective or ineffective under concentrated flow conditions (Dillaha et al., 1988; Owens et al., 2007; Habibiandehkordi et al., 2017), several studies demonstrate that VBSs effectively reduce P export to downstream waters under diffuse, shallow flow conditions by intercepting surface runoff, promoting infiltration and sedimentation, and providing an opportunity for physical, chemical, and biological removal of P from surface runoff (Syversen, 2005; Dorioz et al., 2006; Roberts et al., 2012). However, the long-term efficiency of VBSs for minimizing P exports even under ideal circumstances remains uncertain, as VBS soils may become P saturated after some time, thereby releasing stored P to runoff (Roberts et al., 2012; Kieta et al., 2018). Understanding the magnitude and timescale over which P is retained or remobilized in VBSs is highly important to assess the efficiency of VBSs for reducing legacy P losses in the long-term, and to understand how this would impact downstream water quality.

Unlike water quality monitoring approaches, soil P analyses can be used as a fast and cost-effective approach to understand how VBSs respond to multiple and dynamic patterns of legacy P exports from agricultural land. It is assumed that greater soil P concentrations in VBS soils than adjacent land within and outside of flow paths could be evidence of a long-term VBS function in trapping sediments and associated P (Cooper et al., 1995; Sheppard et al., 2006; Habibiandehkordi et al., 2017). Furthermore, vertical P stratification in VBS soils could be also considered as evidence of how VBSs respond to the release of legacy P pools from adjacent land, and whether accumulated P in VBSs can become a source of P to runoff over the long term. However, vertical and/or lateral stratification of P in VBS soils may not be associated with runoff interception and retention of P transported from adjacent land because P can also accumulate in lower landscape positions as the result of vegetation growth, mineralization of grass residues at the soil surface, and soil formation processes over decades or centuries (Roberts et al., 1985), and/or accumulations of surficial crop residues or applications of fertilizer or manure, especially if the soil is not tilled intensively (Selles et al., 1999; Baker et al., 2017). Therefore, understanding the source of sediments and P accumulated in VBS soils is essential to evaluate the long-term effectiveness of VBSs in trapping sediments and associated P.

The cesium-137 ($^{137}$Cs) technique can provide robust evidence regarding sediment (and associated P) movement within agricultural catchments subject to careful planning and expert knowledge, as well as ensuring that fundamental assumptions of the technique are met (Ritchie and McHenry, 1990; Mabit et al., 2013; Alewell et al., 2014; Fulijart et al., 2017). Cesium-137 is an artificial gamma-emitting radionuclide that was produced primarily during the aboveground nuclear weapons tests in the 1950s and 1960s. The fallout of $^{137}$Cs from the atmosphere was first detected in 1954 and was mainly associated with precipitation and labeled surface soils and sediments. A test ban treaty in 1963 resulted in a reduction in $^{137}$Cs fallout except in the case of local and regional incidents like Chernobyl and Fukushima. Due to the rapid and tight sorption of $^{137}$Cs to soil particles (especially clays and silts) and organic matter and its long half-life (30.17 yr), most $^{137}$Cs is retained in the upper layers of the soil profile (Owens et al., 1996; Fulijart et al., 2017), and can be used as a tracer of soil loss, soil and sediment transport, and soil accumulation and sediment deposition (De Jong et al., 1983; Ritchie and McHenry, 1990; Fulijart et al., 2017). Thus, at sites of soil accumulation or sediment deposition like VBSs, the depth profile of $^{137}$Cs is likely to be deeper, and the inventory of $^{137}$Cs greater, than adjacent sites without soil accumulation or sediment deposition. Soil samples collected from stable locations that have experienced no or minimal erosion or deposition are used to determine a local “reference $^{137}$Cs inventory” against which other samples are compared so as to determine the magnitude of $^{137}$Cs, and thus soil, loss or gain (Owens and Walling, 1996).

We used a combination of $^{137}$Cs and soil P analyses with a unique soil sampling approach to determine the P removal efficiency of an unmanaged VBS for controlling the multiple and dynamic patterns of P loss from agricultural land during the period of 57 yr (1954–2011). Specifically, we examined (i) whether P removal efficiency of VBS varied over time in relation to legacy P exports from adjacent land, and (ii) whether VBS soils have the potential to act more as a source of P to surface runoff, rather than a sink in the long term.

**Materials and Methods**

**Site Description**

The study was conducted in 2011 in a 2.5-ha agricultural subcatchment of a small creek within the Whitemouth River watershed, situated near the town of Elma in the Rural Municipality of Whitemouth in southeastern Manitoba, Canada (49°51′18″N, 95°53′44″W) (Fig. 1a). The site was selected following our 6-yr detailed field investigation of sediment and P retention by VBSs in 22 sites across Manitoba (unpublished data, 2012), and it represents an ideal VBS scenario for trapping sediments and associated nutrients in the region. The site has regular surface runoff flow passing from the adjacent agricultural fields through
the VBS with visual evidence of sediment accumulation and lush vegetative growth and is not affected by floodwater from the creek or snowmelt generated within the VBS area itself. The soil is a Dark Gray Luvisol in the Canadian system of soil classification (MAFRI, 2010) that can be also classified as Boralfs and Gleyic Luvisol in the US and World Reference Base systems, respectively (Soil Classification Working Group, 1998). The surface soil (0–10 cm) is sandy loam with an organic carbon content of 2.0 to 2.5% and a pH range of 7.2 to 7.8. The fields at this site had a history of injected hog manure fertilization, but that practice ceased in recent years prior to this study. The fields within the area are classified as having low relief or being relatively flat; the relief of this subcatchment is <1 m, with slopes of <2%. The site had an established, unmanaged VBS that consisted of perennial grass and sedge with a width of 5 to 35 m. In the years leading up to and including the study, the upland field was planted with wheat (*Triticum aestivum* L.) and soybeans (*Glycine max* (L.) Merr.) and was conventionally tilled, maintaining a light cover of crop residues.

**Field Sampling and Laboratory Analyses**

Soil samples were collected from the VBS and the up slope field using a soil core sampler (JMC Backsaver with a 45.7-cm-long, 3.17-cm-diam. slotted sampling tube). Soil sampling was performed within and outside of the main surface water flow path, which originates in the field and travels through the VBS toward the creek (Fig. 1b). The field—VBS boundary was marked as the starting point and samples were taken at 0, 1, 2, 3, and 5 m into the VBS, as well as at 1, 2, and 5 m into the field (marked as −1, −2, and −5 m; Fig. 1b). Using a similar pattern, soil core samples were also taken within the VBS at 2 m (Transsects B and C) and 5 m (Transsects D and E) on either side of the main surface flow path (i.e., Transect A). Additional soil samples were taken at upper and middle slopes of the adjacent field in the contributing catchment area to understand erosion or sedimentation processes within the catchment (Fig. 1b, nonfilled circles). Soil core samples extended to the depth of parent material, 30 to 45 cm on the upper slopes and 60 to 95 cm on the lower slopes. The VBS soil core samples were subsampled at 5-cm intervals, and field soil core samples at 15-cm intervals. From two large, level, undisturbed sites (R1 and R2, Fig. 1c) near the study area, eight soil cores were collected for determining the $^{137}$Cs inventory as the noneroded reference. The reference sites were permanent pasture with no history of cultivation since the mid-1950s. This was confirmed through landowner interviews, field inspection, and historical aerial photography.

Soil samples were air-dried and sieved to <2 mm prior to laboratory analyses. Analyses included $^{137}$Cs, total P (TP) (Akinremi et al., 2003), Olsen-extractable P (Olsen-P) (Olsen et al., 1954), 0.01 M CaCl$_2$–extractable P (CaCl$_2$–P) (McDowell and Sharpley, 2001), and Mehlich-3-extractable Ca, Mg, and P (Ca$_{M3}$, Mg$_{M3}$, P$_{M3}$) (Mehlich, 1984). Cesium-137 was measured as a mass concentration of gamma radioactivity (Bq kg$^{-1}$) detected at 662 keV using Broad Energy Germanium gamma spectrometers (Canberra BE3830) with counting times ranging from 12 to 24 h, providing a detection error <10% (Li et al., 2007). Cesium-137 inventory values (Bq m$^{-2}$) were calculated based on $^{137}$Cs mass activity values (Bq kg$^{-1}$), the dry weight of
the core soil samples, and the sampling core area—i.e., the summation of area-based $^{137}\text{Cs}$ values of all the samples taken from different layers at a given sample location. To assess soil loss and accumulation or sedimentation, inventory values were compared to the noneroded reference level of $^{137}\text{Cs}$ for the site ($1502 \pm 63 \text{ Bq m}^{-2}$). Similarly, the soil P concentrations (mg kg$^{-1}$) were converted to P inventory (kg ha$^{-1}$). The sediment mass inventories (Mg ha$^{-1}$) were calculated according to the dry weight of core soil samples, soil depth (soil with $^{137}\text{Cs}$ activity), and the sampling core area. All processing and analyses were performed in the Department of Soil Science at the University of Manitoba.

We assumed that the first detection of $^{137}\text{Cs}$ in the soil–sediment profiles represents the start of weapon-derived $^{137}\text{Cs}$ fallout in the Northern Hemisphere in 1954 (Cambray et al., 1985). We have not corrected for downward migration and diffusion of $^{137}\text{Cs}$ in the profile (c.f., Owens et al., 1996), as it is not realistic to determine these values for sediment that was deposited decades ago. Consequently, values of $^{137}\text{Cs}$, sediment and P accumulation are likely to be overestimated by <5%, although the degree of overestimation is likely to be similar for all cores.

The particle size of soil samples was measured using a particle size analyzer (Mastersizer 2000, Malvern Instruments). For measuring TP concentrations as a long-term storage of P in soils, soil samples were digested by sulfuric acid and hydrogen peroxide following the procedure of Akinremi et al. (2003) and analyzed for P using an inductively coupled plasma optical emission spectrometer (ICP–OES; ICAP 6300, Thermo Electron). To determine medium- (plant available) and short-term (water extractable) storage of P in the soil, soil samples were extracted by 0.5 M NaHCO$_3$ solution (Olsen et al., 1954) and 0.01 M CaCl$_2$ solution (McDowell and Sharpley, 2001), respectively, and then PO$_4$–P concentrations in the extracts were measured colorimetrically following Murphy and Riley (1962). In addition, some of the soil samples—including 5-cm topsoil and 5-cm subsoil (i.e., the latter representing the initially deposited sediments in VBS in the 1950s with detectable $^{137}\text{Cs}$ activity)—were extracted by Mehlich-3 solution (Mehlich, 1984), and concentrations of Ca, Mg, and P in the extracts (Ca$_{M3}$, Mg$_{M3}$, and P$_{M3}$) were measured in these samples by the ICP–OES. The P sorption capacity (PSC) and the degree of P saturation (DPS) for these layers were calculated based on Ca$_{M3}$, Mg$_{M3}$, and P$_{M3}$ (Eq. [1] and [2]; Ige et al., 2005) recommended for soils of Manitoba:

$$
PSC = \alpha(Ca_{M3} + Mg_{M3})
$$

$$
DPS = 100(P_{M3}/PSC)
$$

where Ca$_{M3}$, Mg$_{M3}$, and P$_{M3}$ are Ca, Mg, and P extracted by Mehlich 3 solution (mmol kg$^{-1}$), as mentioned above, and $\alpha$ is a scaling factor ($\alpha = 0.2$).

**Statistical Analysis**

Statistical analysis was conducted in SPSS version 24.0 (IBM Corporation, 2016). The data were examined to ensure the assumptions of normal distribution and homogeneity of variances prior to statistical analysis. The independent $t$ test was used for comparison between the $^{137}\text{Cs}$ inventories of reference sites and the study site. The Pearson's correlation coefficient was used to test for correlations between soil redistribution ($^{137}\text{Cs}$ activity) and P accumulation along the sampled transects. One-way ANOVA followed by the Tukey post-hoc test was used for multiple comparisons between soil P properties or deposited sediments at different landscape positions. The paired $t$ test was used to statistically compare the differences in PSC and DPS of VBS topsoil (5-cm VBS soil surface in 2011) and VBS subsoil (5-cm VBS soil surface in 1950s as identified by $^{137}\text{Cs}$ analysis). Significant differences were accepted at $p \leq 0.05$.

**Results and Discussion**

**Effectiveness of VBS in Controlling Historical Patterns of Sediments and P Loss**

The mean $^{137}\text{Cs}$ activity of 1112 ± 146 Bq m$^{-2}$ (mean ± SE) at various locations in the upper and middle slopes of the study area was significantly ($p \leq 0.05$) smaller than the mean $^{137}\text{Cs}$ reference inventory value ($1502 \pm 63 \text{ Bq m}^{-2}$), indicating that substantial amounts of soil erosion occurred at these locations between 1954 and 2011. Figure 2 shows that the mean value of $^{137}\text{Cs}$ inventories for soil samples collected in the field at 5 m before the field–VBS boundary (i.e., all transects, $n = 5$) was 4507 ± 126 Bq m$^{-2}$, which is three times of the mean $^{137}\text{Cs}$ reference inventory, suggesting that soil accumulation or sediment deposition in lower landscape positions has occurred within the field portion of the study site since the mid-1950s. Such sedimentation in the field away from the field–VBS boundary is likely associated with reduced slope steepness that would promote infiltration of runoff into the soil and reduce its transport capacity. Further, surface runoff containing sediments and associated P slows down and becomes increasingly dispersive as it passes through the VBS. This created a backwater region a couple of meters before the field–VBS boundary where runoff flow velocity decreased rapidly at this ponded area, leading to deposition of sediments out of suspension. The mean $^{137}\text{Cs}$ inventory values (i.e., in excess of the reference inventory) provide robust evidence that sedimentation associated with runoff interception by VBS has started to occur before the field–VBS boundary and continued through the VBS where the greatest $^{137}\text{Cs}$ inventories were observed at 2 m into the VBS (Fig. 2).

Our field observations suggest that the presence of a topographic step (Supplemental Fig. S1) at the field–VBS boundary at this site, likely caused by tillage over time, may also have limited surface runoff movement beyond the boundary and into the VBS. Owens et al. (2007) also identified the importance of a tillage-induced step in causing sedimentation upslope of VBS in agricultural fields in the United Kingdom.

Table 1 presents Pearson’s correlation coefficients between $^{137}\text{Cs}$ inventories and TP, Olsen-P, CaCl$_2$–P, and total deposited sediment mass along the transects within and outside of the main flow path. Our data demonstrate statistically significant correlations ($p \leq 0.05$) between $^{137}\text{Cs}$ inventories and the mass of TP and CaCl$_2$–P deposited along all transects combined, suggesting that trapping of sediments and associated P occurred due to runoff interception, increased infiltration, and reduced transport capacity by VBS. There were relatively poor relationships between $^{137}\text{Cs}$ inventories and TP, Olsen-P, CaCl$_2$–P, and total sediments accumulated along individual Transects A (main flow path) and B, suggesting that the runoff had a greater velocity and transport capacity, thereby providing less opportunity for
sediment deposition at these positions. Apart from Transect E, there was a poor relationship between Olsen-P and \(^{137}\text{Cs}\) inventories along all transects. Such a poor relationship is expected because only a small fraction of sediment associated P is available in dissolved form, and in Manitoba soils, dissolved P losses are more closely related to Olsen-P than any other soil-test P.

Table 2 presents the mean TP, Olsen-P, and CaCl\(_2\)–P inventories at 5 m before, at the field–VBS boundary, and at 5 m into the VBS. The soils at the field–VBS boundary had significantly greater P contents than field soils, indicating substantial P storage at this landscape position due to runoff interception by VBS, as well as the topographic barrier caused by tillage over time. The VBS soils (i.e., 5 m into the VBS) had significantly lower P content than the field soils and soils at the field–VBS boundary. The substantial decline in TP, Olsen-P, and CaCl\(_2\)–P inventories with increasing distance from the field–VBS boundary into the VBS (Fig. 2, Table 2) is mainly associated with reduced runoff flow rate, increased runoff infiltration, and increased deposition of P into the VBS soils near the interface with the adjacent field. The significantly lower CaCl\(_2\)–P content of soils in the VBS than in the adjacent field soils shows that VBS soils have less potential to release P to runoff. However, researchers have reported that seasonal release of P from VBS soils to runoff could occur as the result of biogeochemical processes such as reductive dissolution of Fe-bound P and dissolved P release from soil organic pools after rapid wetting and drying of the VBS soil (Stutter et al., 2009; Roberts et al., 2012; Stutter and Richards, 2012; Habibiandehkordi et al., 2015b; Gu et al., 2017, 2018).

**Vertical Stratification of P in VBS Soils**

The depth distribution of the \(^{137}\text{Cs}\) activity concentrations in soil samples collected at the field–VBS boundary (0 m) and at 1, 2, 3, and 5 m into the VBS for all transects is shown in Fig. 3. During the period of 1954 to 2011, the greatest soil and sediment accumulation occurred at the field–VBS boundary where \(^{137}\text{Cs}\) activity was detected to a depth of 45 to 50 cm. As mentioned above, our field observations showed that tillage facilitated the creation of a topographic barrier at the field–VBS boundary, which limited surface runoff movement into the VBS. The relatively similar \(^{137}\text{Cs}\) activity concentrations with depth at

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**Table 1. The Pearson’s correlation coefficients between \(^{137}\text{Cs}\) inventories (Bq m\(^{-2}\)) and the deposited mass of sediments or associated P.**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Transect D</th>
<th>Transect B</th>
<th>Transect A</th>
<th>Transect C</th>
<th>Transect E</th>
<th>All transects</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total sediments (Mg ha(^{-1}))</td>
<td>0.857*</td>
<td>0.478</td>
<td>0.401</td>
<td>0.494</td>
<td>0.942*</td>
<td>0.623</td>
</tr>
<tr>
<td>Total P (kg ha(^{-1}))</td>
<td>0.838*</td>
<td>0.507</td>
<td>0.599</td>
<td>0.801*</td>
<td>0.958*</td>
<td>0.723*</td>
</tr>
<tr>
<td>Olsen-P (kg ha(^{-1}))</td>
<td>0.671</td>
<td>0.216</td>
<td>0.177</td>
<td>0.551</td>
<td>0.911*</td>
<td>0.424</td>
</tr>
<tr>
<td>0.01 M CaCl(_2)–P (kg ha(^{-1}))</td>
<td>0.930*</td>
<td>0.440</td>
<td>0.526</td>
<td>0.854*</td>
<td>0.951*</td>
<td>0.805*</td>
</tr>
</tbody>
</table>

* Significant at the 0.05 probability level.

**Table 2. Mean values of the \(^{137}\text{Cs}\) inventories, the mass of sediments and P accumulated over 57 yr at 5 m into the field, the field–vegetated buffer strip (VBS) boundary, and at 5 m into the VBS, and statistical analysis for the differences between these landscape positions. Data are presented as means ± SE (n = 5).**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>5 m into the field</th>
<th>Field–VBS boundary</th>
<th>5 m into the VBS</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total P (kg ha(^{-1}))</td>
<td>3,956 ± 329(^{\dagger})</td>
<td>6,966 ± 130b</td>
<td>2,776 ± 250c</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Olsen-P (kg ha(^{-1}))</td>
<td>338 ± 31.1a</td>
<td>480.8 ± 30.1b</td>
<td>121.4 ± 19.8c</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>0.01 M CaCl(_2)–P (kg ha(^{-1}))</td>
<td>14.7 ± 2.2a</td>
<td>41.1 ± 2.3b</td>
<td>3.2 ± 0.7c</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>(^{137}\text{Cs}) (Bq m(^{-2}))</td>
<td>4,507 ± 126a</td>
<td>10,461 ± 321b</td>
<td>4,941 ± 742a</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Sediments (Mg ha(^{-1}))</td>
<td>5,346 ± 498a</td>
<td>10,702 ± 631b</td>
<td>3,932 ± 424a</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

\(^{\dagger}\) Means with similar letters within each row are not significantly different at the 0.05 probability level.
the field–VBS boundary (Fig. 3) indicate the role of tillage in mixing the deposited sediments associated with runoff interception by VBS with subsoil. The decline in $^{137}$Cs activity concentration with depth for sampling positions within VBS suggests the progressive accumulation of topsoil eroded from adjacent land in the VBS, which have higher $^{137}$Cs concentrations (Fig. 3).

Supplemental Fig. S2 represents vertical P profiles within the VBS along the main flow path. Between 1954 and 2011, P has built up in the VBS, mainly due to deposition of sediments and P transported from adjacent land. The fluctuations in P stratification in the VBS soil column could be evidence of legacy P export from adjacent land, as P loading and concentrations entering VBS are varied depending on runoff hydrology, manure and fertilizer application rate, sediment concentration in runoff, and soil conditions. Apart from a few cases (e.g., concentrations shown in bold in Supplemental Figs S3–S6), upper soil layers in the main flow path generally had greater P concentrations than the lower soil layers. The P fluctuations in the VBS soil profile appeared to be more evident outside the main flow path (see Supplemental Figs S3–S6). The expected P fluctuations in VBS soil profiles could have been compromised due to mineralization of grass residues at the soil surface coupled with a lack of VBS management, resulting in increased P concentration in topsoil over time. Long-term P accumulation in VBSs is a cause of increasing concern, as VBS soils may become saturated with P and may act more as a source of P rather than a sink (Roberts et al., 2012); therefore, PSC and DPS of VBS soils are explored in the section below.

**Phosphorus Sorption Capacity and Degree of P Saturation of Field Soils and VBS Soils**

Estimates of DPS were used to evaluate the potential of P loss from field soils, as well as from the soils accumulated within VBS between 1954 and 2011. Table 3 shows that surface soils at the field–VBS boundary and at 5 m into the VBS had significantly smaller DPS than field soils. The greater saturation of field soils with P than the VBS soils could be associated with historic applications of synthetic fertilizer and hog manure that led to gradual enrichment of field soils with P. Furthermore, VBS soils had significantly greater PSC than the field–VBS boundary and field soils (Table 3). This appears to be due to deposition of fine sediments within VBS with greater surface area, and thus capacity, for sorption (Stutter et al., 2009; Habibiandehkordi et al., 2015b). As coarse particles settle faster than finer ones, most of the coarse sediments were deposited at the field–VBS boundary, whereas finer particles moved further into the VBS (Supplemental Fig. S7). This finding is supported by the $^{137}$Cs activity concentration data shown in Fig. 3, as the $^{137}$Cs content of the deposited sediment, especially for the upper soil layers, increases with distance into the VBS, reflecting the known relationship between $^{137}$Cs activity concentration and specific surface area (i.e., particle size; He and Walling, 1996).

### Table 3. Mean values of P sorption capacity (PSC) and degree of P saturation (DPS) in surface (0–5 cm) soil samples collected from transects located 5 m into the field, at the field–vegetated buffer strip (VBS) boundary, and 5 m into the VBS and statistical analysis for the differences between these landscape positions. Data are presented as means ± SE ($n = 5$).

<table>
<thead>
<tr>
<th>Soil P characteristics</th>
<th>5 m into the field</th>
<th>Field–VBS boundary</th>
<th>5 m into the VBS</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>PSC (g kg$^{-1}$)</td>
<td>0.75 ± 0.01a†</td>
<td>0.78 ± 0.04a</td>
<td>1.08 ± 0.04b</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>DPS (%)</td>
<td>29.7 ± 1.0a</td>
<td>21.6 ± 0.7b</td>
<td>9.9 ± 1.1c</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>

† Means with similar letters within each row are not significantly different at the 0.05 probability level.

### Table 4. Mean P sorption capacity (PSC) and degree of P saturation (DPS) in the 5-cm topsoil layer of the vegetated buffer strip (VBS) in 2011 and in 5-cm subsoil layer of the VBS (initially deposited sediments within the VBS in 1950s) for all transects and the results of statistical analysis. Data are presented as means ± SE ($n = 25$).

<table>
<thead>
<tr>
<th>Soil P characteristics</th>
<th>5-cm topsoil</th>
<th>5-cm subsoil</th>
<th>t</th>
<th>p value</th>
</tr>
</thead>
<tbody>
<tr>
<td>PSC (g kg$^{-1}$)</td>
<td>0.91 ± 0.03</td>
<td>0.84 ± 0.04</td>
<td>1.61</td>
<td>0.121</td>
</tr>
<tr>
<td>DPS (%)</td>
<td>16.4 ± 1.0</td>
<td>5.6 ± 0.5</td>
<td>13.3</td>
<td>&lt;0.001</td>
</tr>
</tbody>
</table>
Table 4 reports the differences in mean PSC and DPS values between topsoil (5-cm VBS soil surface in 2011) and subsoil (5-cm soil deposited in VBS in 1950s) of the VBS at different sampling locations for all transects. The DPS in topsoil of VBS was significantly greater than VBS subsoil, and there was no significant difference between mean PSC of topsoil and subsoil of the VBS. Research on noncalcareous soils in the Netherlands showed that soils with DPS >25% have the potential to release P to runoff (Breeuwsma et al., 1995). Therefore, this VBS does not show evidence that it has become saturated with incoming runoff P. Furthermore, the VBS in our study was unmanaged and potential cutting and removal of vegetation from VBS could likely offset part of the legacy P stored in VBS in the long term. It should be noted that although PSC and DPS are widely used it is used by researchers in the literature, they cannot fully predict the risk of P loss from agricultural soils.

Conclusions

Our study demonstrated that VBSs have the potential to effectively reduce the export of legacy P from agricultural land under diffuse, shallow runoff flow conditions. Most of the sediments and associated P were trapped at the field–VBS boundary or within the first 2 m of VBS due to runoff interception by VBS and a broad topographic step likely caused by historical tillage, which caused ponding and sedimentation. The surface soils of VBS appeared to have a significantly greater PSC and a significantly lower DPS than adjacent field soils. Our findings showed that P accumulation in unmanaged VBS over 57 yr resulted in a significant increase in DPS of VBS topsoil, but only by ~11% compared with VBS subsoil. Although DPS alone is not expected to fully predict soil P loss potential, our data show that incorporation of P from VBS soils due to saturation with P. Management techniques such as occasional harvesting of vegetation and removal of vegetation clippings from the landscape could slow down P stratification in VBS soils, delay possible saturation of VBS soil in the longer term, and minimize P leaching associated with vegetation senescence, particularly in our region where freeze–thaw cycles can accelerate these processes. Further recommendations regarding VBS management in cold-climate regions can be found in Kieta et al. (2018).

Supplemental Material

Seven supplemental figures are available online for this article.

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References


