



Influence of rainfall intensity and slope on suspended solids and phosphorus losses in runoff

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Abstract

Suspended solids (SS) and phosphorus (P) losses in rainfall generated runoff can lead to the deterioration of surface water quality. Simulated rainfall experiments were conducted to investigate the effects of rainfall intensity (30, 50, 65, and 100 mm h⁻¹) and land slope (0°, 5°, and 10°) on SS and P losses in runoff from experimental rigs containing bare land soil and soil planted with grass (tall fescue). In addition, total phosphorus (TP), particulate phosphorus (PP), and dissolved phosphorus (DP) losses in runoff were also measured. Results showed that tall fescue could reduce loads of SS by 86–99.5%, PP by 92–98.5%, and TP by 55–89.8% in runoff compared with losses from bare soil; this is due to a combination reduced raindrop kinetic energy at the soil surface, reduced soil erodibility in the presence of plant roots and shoots, and an increase in roughness and consequently reduced overland flow velocity resulting in the trapping of particles. Linear relationships between losses of SS and TP and between TP and PP in runoff were significant ($R^2 > 0.93$) in both bare soil and grass. In addition, SS and TP losses increased greatly significantly with rainfall intensity and slope. The influence of rainfall intensity on SS and P losses was greater than the influence of slope. Simple linear regressions were constructed between losses of SS and P, the rainfall intensity (30 to 100 mm h⁻¹), and land slope (0° to 10°). The multiple regression equations of SS and P losses in runoff established in this study can provide a simple predicting approach for estimating the non-point source pollution load of SS and P arising from rainfall.

Keywords Rainfall intensity · Slope · Bare soil · Grass · Suspended solids · Phosphorus

Introduction

Excess phosphorus can negatively affect the quality of ground and surface waters and can trigger eutrophication (Carpenter et al. 1998; Gao et al. 2010; Peacher et al. 2018). While phosphorus discharged from point sources (PS) such as wastewater treatment works can be controlled relatively easily (e.g., via tertiary treatment (Maher et al. 2015) or using tertiary constructed wetlands, albeit with variable results (Hickry et al. 2018), non-point source (NPS) phosphorus transfers from soil to water (e.g., from agricultural land) are harder to manage due to the complex (and variable) nature of the interactions

between sources and transport mechanisms and due to the fact that it is produced from many different farms in a catchment (Jiang et al. 2009), each containing many different fields with different crops, land use histories, and physical characteristics (Carpenter et al. 1998; Chen et al. 2012; Tediosi et al. 2012). Currently, the presence of NPS pollution due to rainfall-induced runoff is of increasing concern to the public in China (Chen et al. 2018; Wu et al., 2012a). The transfer of phosphorus and other nutrients from agricultural soils to water bodies via a range of runoff pathways is known to be a major cause of water pollution. Previous research has shown that the two main forms of phosphorus in surface runoff (overland flow) are particulate phosphorus (PP) and dissolved phosphorus (DP). PP typically accounts for more than 70% of the P losses via this pathway mainly through sorption of P to soil particles which are mobilized by rain splash and entrained by sheet wash and concentrated flow in rills (Wang et al. 2013). However, it should be noted that there are cases where dissolved P dominates P losses via surface runoff (e.g., Heathwaite and Dils 2000). It is also important to note that where artificial field drains are present (as is the case on much

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of the world's heavy clay soils; Gilliam and Skaggs 1994), P losses to drains (frequently via preferential flow) can greatly exceed losses via overland flow (e.g., Heathwaite and Dils 2000).

Relationships between soil erosion and runoff depend on site-specific factors such as rainfall intensity (usually mm h^{-1}), runoff rate (mm h^{-1}), topographic gradient (slope), soil physical properties (e.g., texture and organic matter content), antecedent soil moisture content, and land cover (Blanco-Canqui and Sharpley 1985; Martínez-Murillo et al. 2013; Mayerhofer et al. 2017; Serrano-Muela et al. 2013). Perhaps unsurprisingly, rainfall intensity has been found to be one of the most important factors influencing overland flow and nutrient losses (Cerdà 2002; Serrano-Muela et al. 2013). Wang et al. (2014a) found that kinetic energy from rainfall was one of the most important factors affecting material transport in runoff, in which mobilization by rainfall splash is usually the first step to particulate losses. An increase in P transport (mainly as PP) has been observed with increasing overland flow and erosion induced by greater rainfall intensities (Martínez-Zavala and Jordán 2010; Shigaki et al. 2007). The effect of slope on soil loss is also important, but its role is complex (Zhang et al. 2017). Soil loss generally increases as the slope becomes steeper, although interactions with soil properties and surface conditions are also believed to exist (Mah et al. 1992; Fu et al. 2011). Mahmoodabadi and Sajjadi (2016) explored the interaction between rainfall intensity and slope on soil erosion and suggested that the contribution of rainfall to rain-induced erosion may even decrease on steeper slopes (> 10%).

Another important and long-recognized control on overland flow and erosion is vegetation. Vegetation can intercept precipitation and hence reduce rain splash kinetic energy at the soil surface, plant roots can help to stabilize soil particles and promote aggregation, and stems can increase surface roughness and, hence, reduce overland flow velocities which can promote infiltration and enhance deposition of particulates (Melville and Morgan 2010). Laboratory (Blanco-Canqui et al. 2006; Zhao et al. 2016) and field (Lee et al. 2003; Hernandez-Santana et al. 2013; Udawatta et al. 2011; Webber et al. 2010) studies indicate that vegetative filter strips (VFS) significantly reduce sediment and nutrient losses in runoff (e.g., Coyne et al. 1995; Rankins et al. 2001). The most widely used vegetation type for VFS in the USA is grass (fescue) which is extensively used in the midwest (Hopkins et al. 2010; Rankins et al. 2001). Tall fescue has also been used in China (Xu et al. 2014).

Many studies have investigated the relationship between the P content in near-surface soil and the P transported in surface runoff (Carpenter et al. 1998; Djodjic 2004; Shigaki et al. 2007; Wang et al. 2017). These studies have often reported that the DP concentration in runoff does depend on the soil P concentration (Vadas et al. 2005; Vadas et al. 2011). However, fewer studies have quantitatively evaluated SS

and P losses in runoff from bare soil and grassland. In the paper, we describe a series of experiments in which artificial rainfall was applied to erosion rigs containing either bare soil or soil planted with tall fescue, in order to evaluate the influence of rainfall intensity (30, 50, 65, and 100 mm h^{-1}) and slope (0° , 5° , and 10°) on SS, TP, and PP losses in overland flow and the relationships between these variables.

Materials and methods

Soil collection and soil properties

Soil in this study was a sandy loam, collected from Changping District, Beijing, which is the most common soil type in the northern region of China. The soil is a sandy loam (USDA) with high permeability and is, therefore, unfavorable to the generation of surface runoff. According to historical weather records from the past 50 years in Beijing, the average annual temperature is between 10 and 12°C , and the annual precipitation is approximately 470–660 mm, which is quite unevenly distributed over the year. Precipitation from April to September accounts for over 90% of annual rainfall.

Soil moisture content was measured using time domain reflectometry (TDR; Hydra-Probe II, Beijing). Soil bulk density was determined using the standard ring method (e.g., McKenzie et al. 2004). Particle size distribution was determined using a Microtrac S3500 laser particle sizer after appropriate dispersion (e.g., Eshel et al. 2004). The size distribution was 74% sand, 26% silt, and < 1% clay. Air-dried, sieved (< 2 mm) sub-samples were used to measure the soil physical and chemical properties, including pH, organic matter (OM) content (Bowman and Cole 1978), total phosphorus (TP) (Zhu et al. 2017), and available phosphorus (AP) (Huang et al. 2016). The pH of the experimental soil was 7.34, and the OM, TP, and AP concentrations were 19.1 g kg^{-1} , 830 mg kg^{-1} , and 4.43 mg kg^{-1} , respectively.

Experimental design and water quality analysis

Simulated rainfall experiments were carried out indoors in a bespoke hall located at the Jiufeng test base of Beijing Forestry University, Changping District, Beijing. A schematic diagram of the rainfall simulating system is shown in Fig. 1. Air-dried and sieved (< 2 mm) soil was packed into erosion beds consisting of duplicated permeable-bottomed boxes fitted with permeable bases overlain with a layer of sand (1.0 m long, 0.3 m wide, and 0.5 m deep) to achieve a field bulk density of about 1.40 g cm^{-3} (consistent with the bulk density observed in the field). The slope of each bed could be adjusted. There were six beds in total which were evaluated under simulated rainfall at the same time. The rainfall simulator (Fig. 1a) was positioned 12 m above the erosion beds in

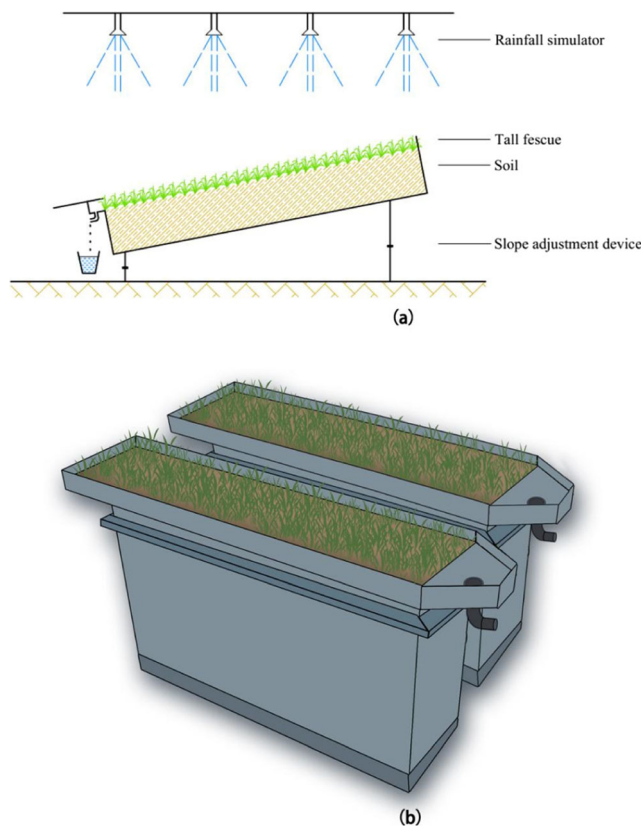


Fig. 1 Schematic diagram of rainfall simulation system (a) and the erosion beds (b)

order to allow rain drops to reach their terminal velocity and achieve a realistic kinetic energy (Lassu et al. 2015). The effective rainfall area is 256 m². It was used to generate rainstorms that were similar to high intensity natural rains forms. The rainfall process was automatically controlled by computer, and the potential range of simulating rainfall intensities was 10 to 300 mm h⁻¹. Here, soil was subjected systematically to intensities of 30, 50, 65, and 100 mm h⁻¹ with corresponding rainfall durations of 3, 3, 2, and 1 h, respectively. These intensity and duration combinations are representative of high intensity erosive storm events which are observed in the study area.

Topographic gradients in the study region are mostly gentle. Slope angles of 0°, 5°, and 10° were, therefore, selected for the experiments (Shi et al. 2012). Tall fescue (*Festuca arundinacea*), a typical grass which grows in northern China, was chosen as the experimental vegetation cover. In erosion beds planted with *F. arundinacea*, coverage rate was over 90%.

The experiment was arranged in a factorial design, with three factors (slope, rainfall intensity, and vegetation coverage) and two replicates per factor. Rainfall intensity in the experimental system was checked using a set of plastic containers which were deployed in the area receiving rainfall in grid with a 10-cm spacing between one container and the next. The spatial distribution of the simulated rainfall was assessed, and uniformity was estimated to be approximately 85% in a 30 min

period. Surface runoff was collected and the volume recorded every 5 min for the first 30 min (starting from the point at which runoff was observed to reach steady state) and then every 10 min for the next 30 min. Where duration was > 1 h, runoff was collected every 20 min for the period 60–120 min, and every 30 min for the period 120–180 min, where relevant. The time to reach steady state was slightly different in each run.

Runoff water samples were stored overnight at 4 °C and analyzed within 24 h after sampling (Harris et al. 1995). SS concentrations in runoff samples were determined gravimetrically via vacuum filtration through a 0.45-μm membrane and subsequent drying at 105 °C. SS was expressed as the particle mass per unit volume of sample. In order to determine TP, unfiltered samples were digested using potassium persulfate to oxidize the different phosphorus fractions to orthophosphate (PO₄-P) and then reacted with ammonium molybdate which forms a blue complex with phosphate. Concentrations were determined from a calibration curve with light absorbance at 700 nm using a UV/VIS spectrophotometer (Huang et al. 2016). DP was determined on the undigested 0.45 μm filtered water samples using the same molybdate method as described above. Organic phosphorus in 0.45-μm filtered samples was assumed to be negligible. PP was calculated by the difference between TP and DP. Each of measurements was conducted in triplicate.

Data analysis

Pollutant loads were calculated from the following:

$$L = (1/T) \frac{\sum_{i=1}^n C_i \cdot V_i}{S_0} \tag{1}$$

where *L* is pollutant load (mg m⁻² h⁻¹), *C_i* is the pollutant concentration at sampling time *I* (mg L⁻¹), *V_i* is the runoff volume collected between sampling time *i* – 1 and sampling time *i*, *n* is the number of sampling events, *T* is the total time, and *S₀* is area of the erosion (m²).

Figures were drawn using Origin 9.0, and statistical analysis was performed using the Microsoft Excel 2010 and SPSS for Windows 17.0. Multi-factor analysis of variance and regression analyses were used to examine the relationships between pollutants and explanatory factors. For all regression analyses, the level of significance was *p* < 0.05.

Results and discussion

The variation of SS and P concentrations in runoff

Changes in the concentrations of SS and different P fraction in runoff over time for different rainfall intensities and for different slope angles are shown in Fig. 2 for the bare soil treatment and for the vegetated surface in Fig. 3. The *x*-axes show the

time from the onset of rainfall but data are only plotted from the time runoff was first collected. In the case of the bare soil treatments (Fig. 2), the temporal patterns in SS and P concentrations were similar in all treatments (except for the zero slope, low intensity trial where concentrations varied less systematically). Concentrations of both SS and P were high in the early stages of runoff but then decreased asymptotically to a constant level—typically 30–40 min after the onset of rainfall. The initially high concentrations are similar to the “first flush effect” observed in urban storm water runoff (e.g., Buffleben et al. 2002) and may be due to the disruption of soil aggregate integrity by rain splash, followed by particulate entrainment once overland flow depths generate sufficient shear stress (Jiang et al. 2008). As well as the potential for exhaustion of fine mobilizable particles later in the event, particles dislodged by splash which fall back to the surface can produce a layer of that provides a degree of protection against the detachment of other particles underneath (Kinnell 2005). Furthermore, the effects of rain splash are typically reduced once water accumulates on the soil surface because the critical raindrop energy needed to cause

detachment increases. This is because raindrop energy is dissipated (to some extent) by the water film rather than by the soil particles themselves (Kinnell 2005).

As expected, peak SS and P concentrations increased with increasing of rainfall intensity and slope with the exception of Fig. 2. This is consistent with Kinnell (2000) who reported that sediment concentrations increased with increasing gradient, particularly when the gradient was over 10% (5.7°). Morgan (2005) also reported sediment yield increased with increasing rainfall intensity and slope, ascribing the effect of intensity to the increase in raindrop kinetic energy (splash effects) and the effect of slope to the increase in overland flow velocity and associated increase in shear stress. The patterns observed for P concentrations are consistent with the result of Shigaki et al. (2007), who reported that PP and TP concentrations were significantly greater ($p < 0.05$) in runoff generated by rainfall at 75 mm h^{-1} than generated by 25 mm h^{-1} . Kleinman et al. (2006) also observed that when rainfall intensity increased from 29 to 70 mm h^{-1} , the resulting runoff had a greater TP concentration (0.55 and 0.99 mg L^{-1} , respectively).

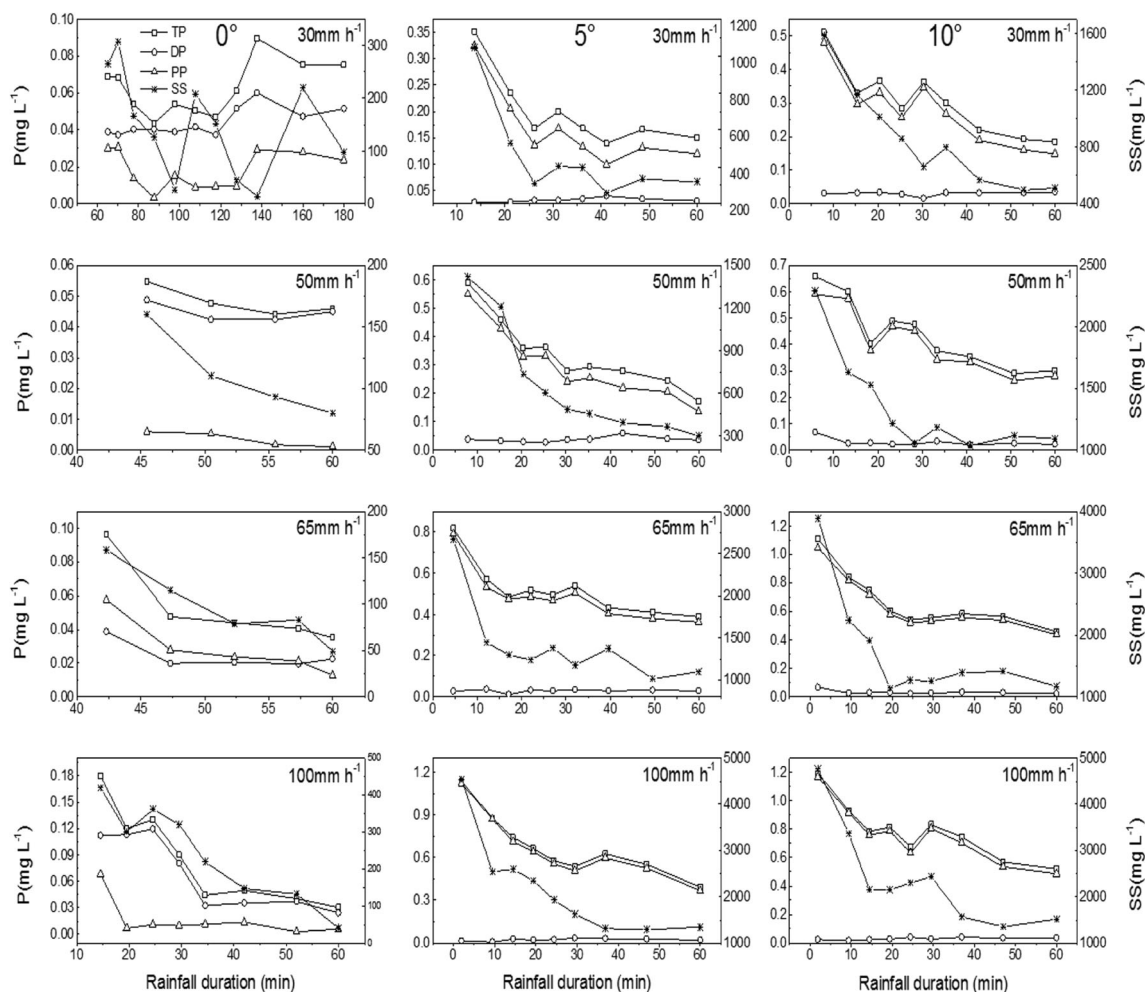


Fig. 2 Concentrations of SS, TP, DP, and PP in runoff from bare soil over time for different slopes and rainfall intensities. SS suspended solids, TP total phosphorus, PP particulate phosphorus, DP dissolved phosphorus. Note different axis scales

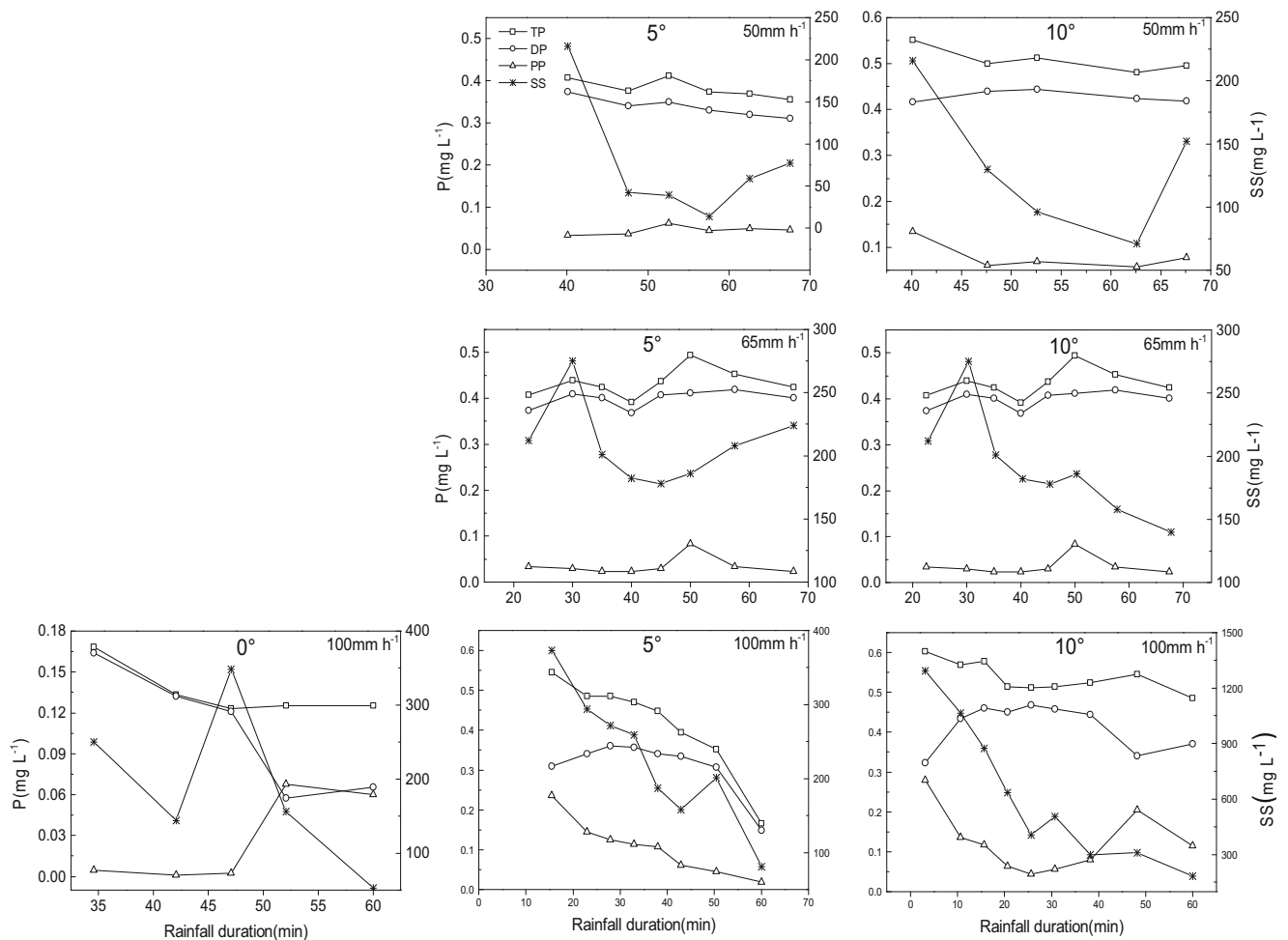


Fig. 3 Concentrations of SS, TP, DP, and PP in runoff from vegetated surfaces over time for different slopes and rainfall intensities. See Fig.2 for abbreviations. Note different axis scales

In our study, PP accounted for 83.7% of the TP collected. DP concentrations were consistently very low. Concentrations of DP in soil pore water are often low due to the high propensity of orthophosphate to react with iron- or aluminum-rich minerals in acidic soils and with calcium-rich minerals in alkaline soils to be taken up by plants or to be immobilized by the soil microbial biomass (Zhu et al. 2018).

In the case of the vegetated surfaces, no runoff at all was observed when the slope was 0°, except when a rainfall intensity of 100 mm h⁻¹ was applied. There was also an increase in the time required to initiate runoff when *F. arundinacea* was present, compared to bare soil. The additional delay varied between 14 and 38 min (Fig. 3). This was principally due to differences in the soil surface conditions. Grass roots tend to promote better soil structure (i.e., the development of aggregates) which promotes better infiltration (Daynes et al. 2013). Plant stems also generate higher surface roughness compared with bare soil surfaces which reduces overland flow velocities and associated shear stress and hence reduces the propensity for particle entrainment (Pearce et al. 1998; Wu et al. 2011). The initial (peak) SS and P concentrations observed in runoff at 65 mm h⁻¹ were lower than those

generated at 100 mm h⁻¹ and were similar to those generated at 50 mm h⁻¹. However, the effect of an increase in slope from 5° to 10° was less pronounced particularly at lower intensities (Fig. 3). At 100 mm h⁻¹, all concentrations were much higher at 10° than at 5° and for TP and DP at 10°, they remained high over the whole duration of the event, whereas at 5°, they fell away steeply. The SS concentrations were significantly lower in runoff from the vegetated surfaces compared to equivalent conditions under bare soil. However, DP was a more important component of TP in runoff in the present of vegetation. This was due to the effectiveness of *F. arundinacea* in reducing erosion with consequently lower SS concentrations (and associated PP) in runoff.

SS and P losses with rainfall runoff

SS and P losses in bare land and grassland runoff

Calculated loss rates of SS, TP, and PP losses in bare land and grassland runoff under different rainfall intensities and slopes are shown in Table 1. SS and P losses increased significantly with increases in rainfall intensity and slope for both the bare soil and

Table 1 Losses of SS and P losses in runoff under different rainfall intensities and different slopes

Slope	Underlying	Bare land				Tall fescue grass			
		30	50	65	100	30	50	65	100
	Rainfall intensity (mm h ⁻¹)								
0°	SS (g m ⁻² h ⁻¹)	0.03	1	1.19	5.37	0	0	0	0.11
	TP (mg m ⁻² h ⁻¹)	0.1	0.66	0.87	2.30	0	0	0	0.60
	PP (mg m ⁻² h ⁻¹)	0.06	0.10	0.09	0.22	0	0	0	0.01
	PP/TP (%)	60	15.15	10.34	9.57	–	–	–	2.10
5°	SS (g m ⁻² h ⁻¹)	10.75	24.41	96.99	166.68	0.04	0.66	5.64	15.44
	TP (mg m ⁻² h ⁻¹)	6.05	11.91	33.08	64.59	0.54	3.34	9.47	19.87
	PP (mg m ⁻² h ⁻¹)	5.39	9.97	32.71	60.82	0.07	0.15	1.04	4.58
	PP/TP (%)	89.09	83.71	98.88	94.16	12.44	4.62	10.95	23.06
10°	SS (g m ⁻² h ⁻¹)	20.65	81.84	108.59	159.32	0.72	0.54	10.00	22.25
	TP (mg m ⁻² h ⁻¹)	8.94	26.79	25.37	68.1	1.05	5.23	10.34	30.69
	PP (mg m ⁻² h ⁻¹)	8.03	24.72	24.13	64.69	0.16	0.86	1.37	5.20
	PP/TP (%)	89.82	92.27	95.11	94.99	15.63	16.36	13.26	16.93

SS suspended solids, TP total phosphorus, PP particulate phosphorus, DP dissolved phosphorus

vegetated treatments. For example, SS losses in runoff from bare soil at 5° increased from 10.8 g m⁻² h⁻¹ at an intensity of 30 mm h⁻¹ to 167 g m⁻² h⁻¹ at 100 mm h⁻¹. Analogous losses of TP and PP increased from 6.05 and 5.39 g m⁻² h⁻¹, respectively, at 30 mm h⁻¹ to 64.6 and 60.8 mg m⁻² h⁻¹, respectively, at 100 mm h⁻¹. Other workers (e.g., Wang et al. 2014a) have also reported that sediment and available phosphorus losses tend to

increase with rainfall intensity and slope with total nutrient losses dominated by the sediment-associated fraction.

The data in Table 1 confirm the qualitative observations from Figs. 2 and 3 that SS and P losses were consistently lower from the vegetable surface than from bare soil (*cet.par.*). They suggest that *F. arundinacea* is a very effective control for erodibility (Cerdà and Doerr 2010) and associated P losses in

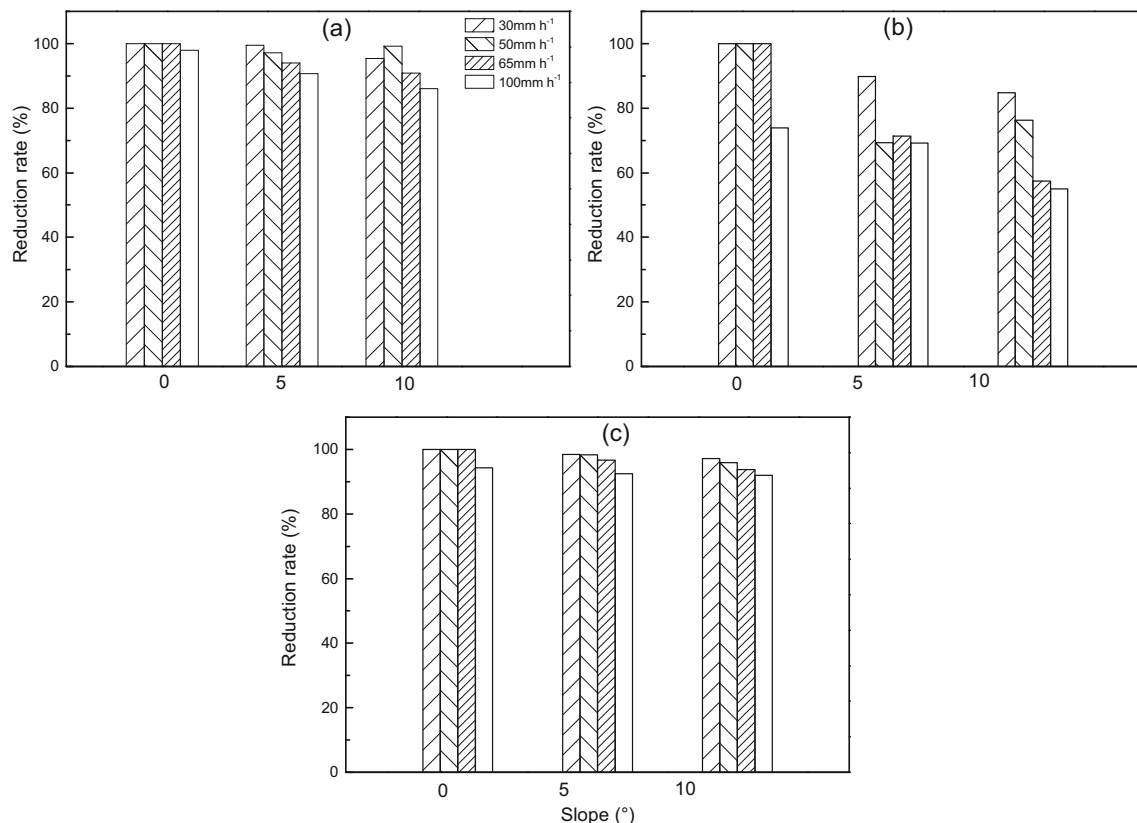


Fig. 4 Reduction rate of SS, TP and PP on vegetated surfaces compared with bare soil. **a** SS. **b** TP. **c** PP. See Fig. 2 for abbreviations

surface runoff. In the bare soil treatments PP accounted for the most (> 80%) of the TP at 5° and 10°, although at zero slope, PP losses were low. In the vegetated trial, PP represented between 2.1 and 23.1% of TP losses with DP dominating TP loss. This is consistent with the explanation of Blanco-Canqui et al. (2004) that reductions in PP flux are often due to sediment deposition promoted by vegetation-induced infiltration and enhanced by P adsorption by colloidal (clay) particles.

Reduction rate of SS and P by tall fescue grassland

Vegetation cover has long been recognized as an effective measure to improve infiltration and soil erosion (Smith and Wischmeier 1957; Renard et al. 1991) and associated nutrient loss. However, on permanent grassland, P concentrations can be especially high at the soil surface due to the absence of tillage, which in turn increases the potential for DP losses, either in overland flow or via leaching (Sharpley 2003). This is particularly so if P fertilizers have been applied historically in excess of plant demand, as has commonly been case in many parts of the world (e.g., Withers et al. 2001; Massey et al. 2013).

Reduction rates of SS and P losses by *F. arundinacea* were calculated based on the difference between the bare soil and vegetated treatments and are shown in Fig. 4. The reduction rates range from 86 to 99.5% for SS, 92 to 98.5% for PP, and 54.9 to 89.8% for TP at 5° and 10°. For the zero slope treatments, there was no loss of SS or P from the vegetated surfaces, except at a rainfall intensity of 100 mm h⁻¹. Overall, the reduction rates of SS and P losses decreased as rainfall intensity and slope increased, but the relationship between reduction rate and these explanatory variables is complex.

Correlation analysis between SS and TP, TP and PP losses

As well as changes in load, the form of P transferred of the experimental slopes changed over time. Most PP load was transferred in the early part of each storm event, while DP concentrations tended to be maintained and, hence, to make a more important relative contribution to TP transfers towards the end of each event. Although many studies have been conducted on the relationships between SS, TP, and PP, there are relatively few which have described these relationships quantitatively.

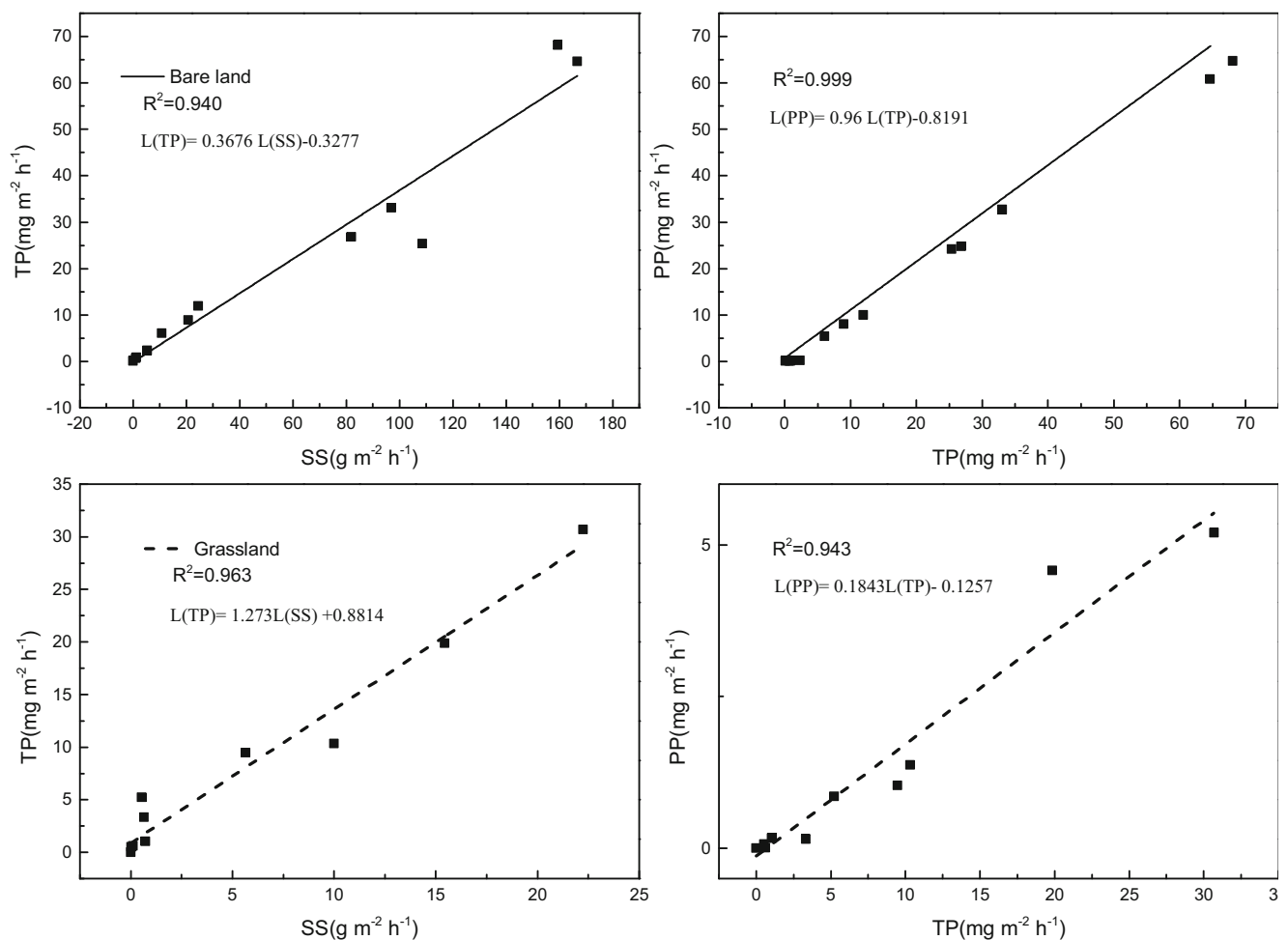


Fig. 5 Relationships between SS and TP, TP, and PP losses in surface runoff from bare soil and vegetated surfaces. See Fig. 2 for abbreviations. Note different x-axis scales

Regression equations and coefficients of determination for the linear relationships between SS and TP losses and between TP and PP losses for both bare soil and vegetated treatments are shown in Fig. 5. All the relationships were strong and highly significant confirming the qualitative observations above about the importance of soil erosion for transporting P, particularly from bare soil. The gradient of the best-fit line between TP and SS loads for bare soil was much lower than unity, suggesting that TP losses are also dependent on other factors. Similarly, the slope of the line between PP and TP loads for the vegetated slopes was less than unity, reflecting the fact that DP losses play a much bigger role in TP losses in the presence of *F. arundinacea*.

Influence of rainfall intensity and land slope on SS and P losses

Influence of rainfall intensity on SS and P losses

For the zero gradient trials, the vegetated soil only produced runoff with a rainfall intensity of 100 mm h^{-1} , so SS and P losses were not considered for these treatments. For the other

treatments, the results of regression analysis between rainfall intensity and the loads of SS and TP are shown in Fig. 6 for both bare soil (top panels) and vegetated slopes (bottom panels). Values of R^2 for all relationships between SS loads and intensity and between TP loads and intensity were high (> 0.85), but the relationships were stronger (line gradients higher) for treatments with higher slope. For bare soil, line gradients were low when the slope was 0° (0.076 and 0.031 for SS and TP, respectively) reflecting low surface runoff to the outlet and associated total losses. When the slope was higher, the line gradients were higher and similar to one another for angles of 5° and 10° (suggesting that intensity was a more important control than topographic gradient in these treatments). In treatments with vegetated surfaces, the effect of an increase in slope from 5° to 10° on SS and TP losses was more pronounced, resulting in a steeper line gradient in the relationship between intensity and pollutant flux.

Influence of land slope on SS and P losses

Relationships between SS and TP losses and slope under different rainfall intensities are shown in Fig. 7. For the most

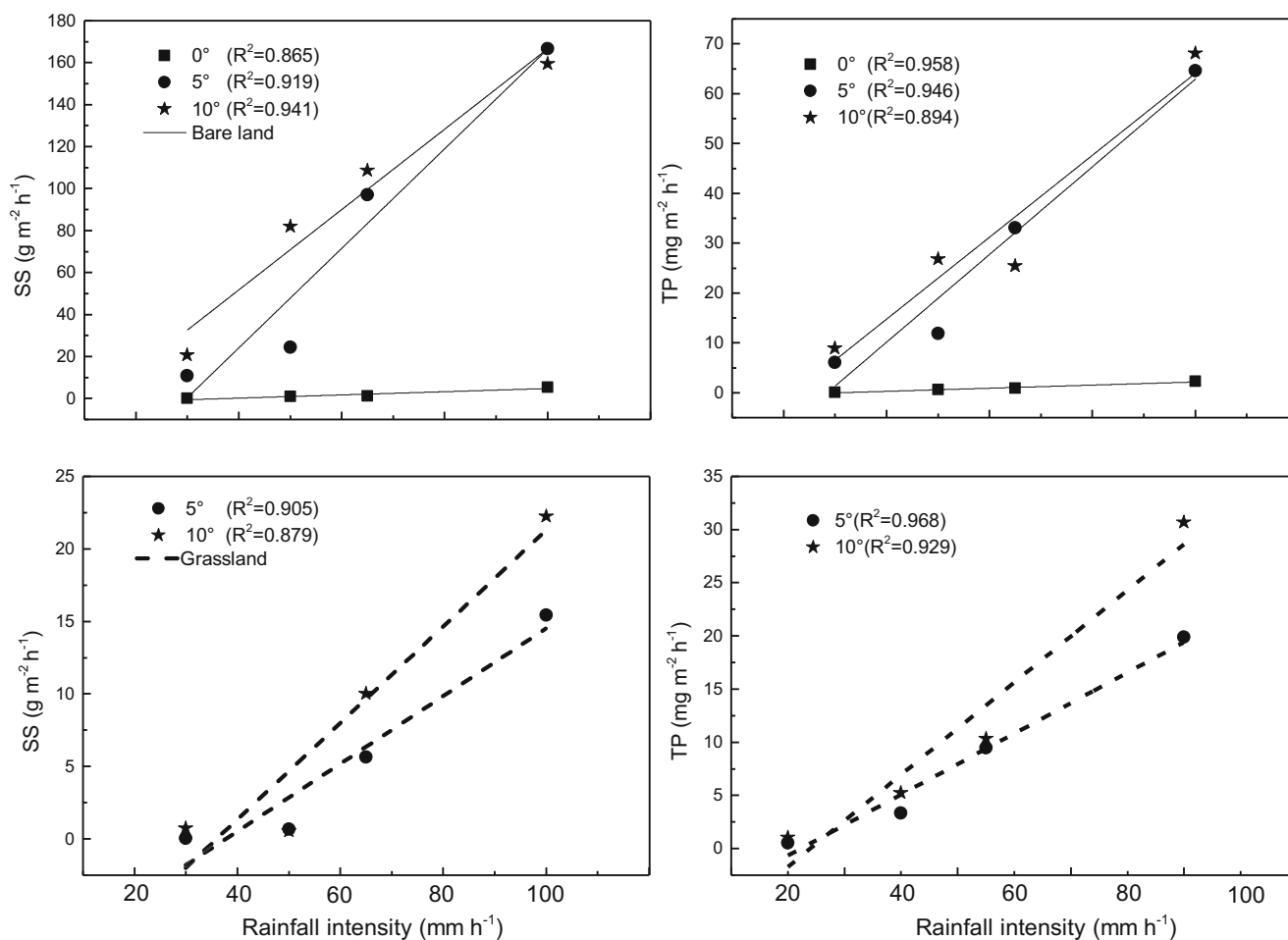


Fig. 6 Losses of SS and TP as a function of rainfall intensity for different slopes with bare soil and vegetated surfaces. See Fig. 2 for abbreviation. Note different x-axis scales

part R^2 values are high here too, with a general increase in loss rate for both SS and TP in both the bare soil and vegetated surfaces. The linear gradients of the relationships also increase with rainfall intensity, reflecting the combined influence of slope and rainfall. These data confirm the findings of Silburn et al. (2011) who reported that SS concentration and loads increased linearly with slope in the range from 4 to 8% on plots with low cover. The effect of slope on erosion is (i) to increase the downslope travel distance of particles released as a consequence of impact by rain drops (splash; see Kinnell 2005) and (ii) to generally increase the flow velocity (for a given roughness and overland flow discharge) and hence increase the shear stress induced by the flow (when supply is not limited sediment flux is proportional to the discharge to a power and slope to a power; Kirkby et al. 2008). It should be noted that the extent to which particles are mobilized or not in an event with particular intensity, duration, and kinetic energy or by overland flow with a given discharge and velocity will also depend on the nature of the soil (often called the erodibility which varies with particle size distribution and, in particular, its cohesion). At lower rainfall intensities, erosion may be “detachment-limited” (where splash and wash fail to

overcome cohesive forces) and at higher intensities, it can be “transport-limited” (Zhao et al. 2015). The precise relationship between these phases and the controlling factors will undoubtedly be soil-specific.

In general, the linear relationships between rainfall intensity and the losses of SS and TP were stronger than those with slope as the explanatory factory. Moreover, the relationships between slope and SS and between slope and TP were stronger at rainfall intensities of 30 and 50 mm h⁻¹ than they were at 65 and 100 mm h⁻¹. This suggests that the effect of slope is diluted at high intensity (i.e., intensity and associated overland flow discharge become the dominant controls regardless of slope; Ahuja et al. 1981; Sharpley 1985; Zhao et al. 2015). This is consistent with the finding that the exponent for discharge in generic relationships for erosion rate is often larger than that for slope (Kirkby et al. 2008).

Combined influence of rainfall intensity and slope on SS and P losses

It is now well recognized that soil erosion and associated nutrient losses tend to increase with rainfall intensity and the

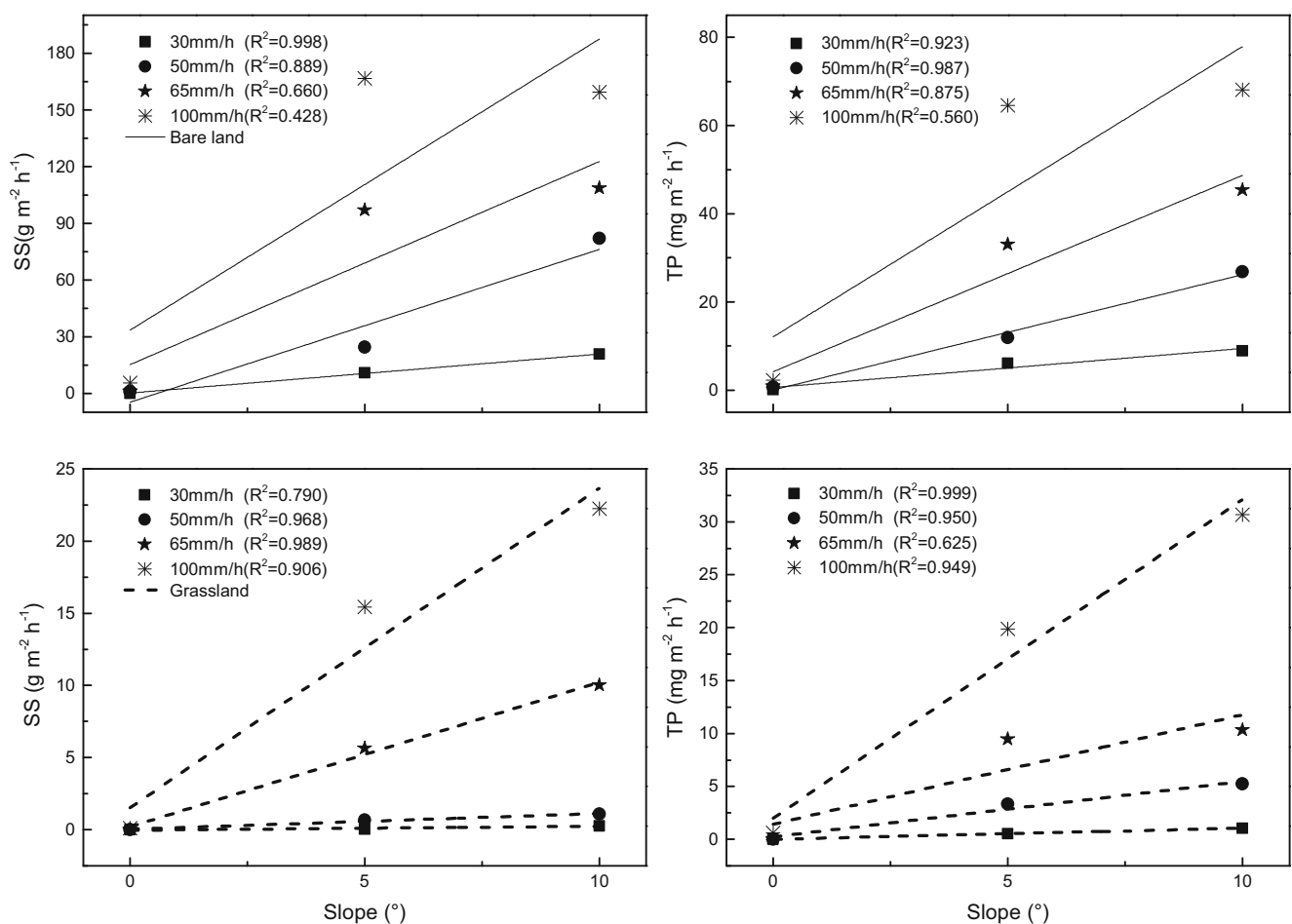


Fig. 7 SS and TP losses as a function of slope under different rainfall intensities with bare soil and vegetated surfaces. See Fig. 2 for abbreviations

local topographic gradient (Quansah 2010; Martínez-Murillo et al. 2013; Wang et al., 2014b; Chen et al. 2017). These factors are incorporated into empirical soil erosion models (e.g., the universal soil loss equation and its revisions; Renard et al. 1991; Bischetti et al. 1997) and into process-based conceptual models (e.g., Kirkby et al. 2008). The extent to which particle losses also control losses of P will depend on other factors, including the concentration of total P in the near-surface soil and the distribution of this P into unavailable (fixed) inorganic fractions (such as weathered rock minerals), exchangeable P, organic P, and dissolved inorganic P. The loss of DP will depend on the temporal pattern of DP concentration in the near surface soil pore water, which will be a function of the sizes of the other P pools and the history of the soil (e.g., cropping history, season and fertilizer application history; see Zhu et al. 2018). The DP concentration in overland flow could also be a function of the contact time between the overlying water and the underlying pool of DP (DP will move to the overlying water via a combination of diffusion and advection) and of the desorption rate (Wu et al., 2012b). Work by Ziadat and Taimeh (2013) has suggested that rainfall intensity is more important than slope as a controlling factor. To evaluate the combined effects of these factors on SS, TP and PP losses a multiple linear regression analysis was performed. The results are summarized in Table 2. Note that data collected for zero slope trials was not included in the regression analysis because surface runoff generation was highly variable. In general, the two explanatory variables (rainfall intensity and slope) could explain a very high proportion of the variance in SS and P losses ($p < 0.05$, $R^2 > 0.86$). Values of R^2 were higher in the bare soil treatments than in the vegetated ones (Table 2), suggesting that other factors could also be responsible for material losses in the presence of a grass sward.

Finally, there are simple linear regression relationships between the cumulative runoff volume collected and the losses of SS, TP, and PP (Table 3). These relationships were all highly significant ($p < 0.01$, $R^2 > 0.911$) for both the bare

Table 2 Integrated relationships between pollutant losses, rainfall intensity, and land slope

Underlying	Pollutants	Regression equation	R^2
Bare land	SS ($\text{g m}^{-2} \text{h}^{-1}$)	$L_{(\text{SS})} = 2.144q + 3.578S - 74.478$	0.942
	TP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{TP})} = 0.851q + 0.678S - 26.609$	0.947
	PP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{PP})} = 0.815q + 0.634S - 25.848$	0.945
Grassland	SS ($\text{g m}^{-2} \text{h}^{-1}$)	$L_{(\text{SS})} = 0.283q + 0.587S - 14.817$	0.862
	TP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{TP})} = 0.360q + 0.705S - 17.290$	0.894
	PP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{PP})} = 0.071q + 0.088S - 3.298$	0.867

SS suspended solids, TP total phosphorus, PP particulate phosphorus, L the losses of pollutants, q rainfall intensity, $30 \text{ mm h}^{-1} \leq q \leq 100 \text{ mm h}^{-1}$, S land slope, $5^\circ < S \leq 10^\circ$

Table 3 Relationships between pollutant losses and collected runoff volume (V)

Underlying	Pollutants	Regression equation	R^2
Bare land	SS ($\text{g m}^{-2} \text{h}^{-1}$)	$L_{(\text{SS})} = 0.0054V - 13.265$	0.911
	TP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{TP})} = 0.0022V - 7.9231$	0.935
R	PP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{PP})} = 0.0021V - 8.0441$	0.933
Grassland	SS ($\text{g m}^{-2} \text{h}^{-1}$)	$L_{(\text{SS})} = 0.0013V - 1.2596$	0.966
	TP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{TP})} = 0.0016V - 0.2547$	0.941
	PP ($\text{mg m}^{-2} \text{h}^{-1}$)	$L_{(\text{PP})} = 0.0003V - 0.3206$	0.958

SS suspended solids, TP total phosphorus, PP particulate phosphorus, L the losses of pollutants, V runoff volume (L)

soil and vegetated treatments; although, in part, this is due to the fact that runoff volume is used to compute the pollutant loads (Eq. 1).

Conclusions

This research shows (not unexpectedly) that SS and P losses from a sandy loam soil in surface runoff increase with increasing rainfall intensity (30 to 100 mm h^{-1}) and topographic gradient (0° to 10°). However, it is important to recognize that, although the soil texture used in the experiments we describe is very common (especially in this part of China), it is particularly prone to erosion because coarse particles are generally weakly cohesive. Rainfall intensity appeared to have greater influence on SS and P losses than slope. Particulate P was the dominant fraction (> 80%) of total P loss when the soil was bare, but dissolved P was dominant in the presence of a grass (*F. arundinacea*) cover.

The relationships between SS and TP losses and between TP and PP losses, in both the bare soil and vegetated slope runoffs, were very significant ($p < 0.01$). Linear equations could also explain a large fraction (> 86%) of the variance in SS and TP on the basis of rainfall intensity and slope in both cover types ($p < 0.01$). These equations can be used as the basis for predicting non-point source pollutant losses under different rainfall intensities and slope in the field, although it is likely that other factors (e.g., topographic controls on soil moisture) will come into play at the landscape level. It is also important to note that although the use of a full height (12 m) rainfall simulator ensured that soils were exposed to realistic rainfall kinetic energies, the use of air-dried, sieved, and reconstituted soil blocks may have had disruptive effects on soil structure. Undisturbed soils may behave differently in the field.

Overall, the experiments demonstrate that *F. arundinacea* can be highly effective at reducing SS and P transfers and would, therefore, be an excellent cover choice for spatially targeted erosion control areas (e.g., in-field grass swales) and near stream-vegetated filter strips.

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