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Effect of pasture buffer length and pasture type on runoff water quality following prescribed burning in the Wivenhoe Catchment

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Abstract. Burning of pastures is a management practice adopted by graziers worldwide. When rain falls on burnt pastures, it can lead to increased pollutant transport in runoff. However, this transport can be modified by vegetative buffers which intercept the runoff downslope of burnt areas. This study examines the effects of different pasture buffer lengths (0, 2, and 5 m) on sediment and chemical transport from two pasture sites near Wivenhoe Dam, the main water reservoir for Brisbane City. Simulated rainfall (100 mm/h) was applied to 18 plots on pasture sites after they were burnt, and insoluble and soluble components were measured in the runoff. Most eroded sediment/organic debris accumulated against the first row of the grass buffer strips or was deposited in the upslope backwater region. Buffer length had little impact on the runoff concentrations of NO_3^- and NO_2^- (NO_x), total Kjeldahl nitrogen, and total nitrogen from the 5-m-length upslope plots but was significant for sediment loss rate, filterable reactive phosphate, ammonium, and total and dissolved organic carbon. Pasture type was significant for NO_x, ammonium, sediment loss rate, and total organic carbon only. Burning increased enrichment ratios of nutrients and carbon in the runoff compared with unburnt plots, but a 2-m buffer strip subsequently reduced the enrichment ratio values by >30%. Buffers strips of unburnt pasture grass may provide an effective tool for post-fire erosion control following prescribed burning; however, further work including scaling to larger plot sizes and catchment level is required.

Additional keywords: buffer strip, nutrient loss, pasture burning, post-fire erosion control, sediment loss.

Introduction

Burning of pasture is commonly practised in many parts of the world for its perceived ability to improve pasture quality and palatability, reduce woody shrubs, control weeds, and increase nutrient supply (see for example Svejcar 1989; Orr and Paton 1997) However, burning is known to affect soil erodibility and runoff quality (DeBano *et al*. 1998; Johansen *et al*. 2001; Burke *et al*. 2005; Llovet *et al*. 2008). Water reservoirs in many areas are surrounded by pastures, so when burning is followed by rainfall, this causes runoff pollutants such as sediment, nutrients, and carbon to enter the water and affect water quality. The intensity of the burning affects the type and amount of ash produced as well as the properties of the underlying soil (Dragovich and Morris 2002; Coelho *et al*. 2004; Llovet *et al*. 2008), both of which have implications for the amount of runoff and sediment loss (Robichaud and Waldrop 1994; Benavides-Solorio *et al*. 2004).

Vegetative buffer strips have been widely employed in catchments of water-supply reservoirs to reduce fluxes of pollutants which may affect water quality. The effectiveness of vegetative buffer strips in mitigating pollutant transport in overland flow has been the subject of extensive research (e.g. Meyer *et al*. 1995; Ghadiri *et al*. 2001; Rose *et al*. 2003; Blanco-Canqui *et al*. 2004; Hussein *et al*. 2007, 2008). There are potentially several variables that can influence sediment-trapping efficiency of buffers, including the type of vegetation used, the length of the buffer, sediment type, hydraulic load, flow rate, and topography. Several studies have shown that stiff grass buffers (hedges) as narrow as 20 cm are very effective in trapping sediment (Dabney *et al*. 1995). Shorter, less rigid vegetation, such as pasture grasses, is less effective in trapping sediment (Meyer *et al*. 1995), and longer buffer widths may be required to reduce fluxes of pollutants (Gharabaghi *et al*. 2002). However, it has also been reported that large increases in the length of grass barriers do not always result in substantial increases in sediment-trapping efficiency of the barriers (Ghadiri *et al*. 2001; Blanco-Canqui *et al*. 2004).

While there are some data on runoff from Australian landscapes following burning, most of these data relate to sediment loss from forested areas (e.g. Prosser and Williams 1998; Dragovich and Morris 2002; Lane *et al*. 2006). There is less information on losses of sediment, nutrients, and carbon from burned pastures or on the enrichment of sediment-sorbed contaminants in runoff following burning (Johansen *et al*. 2001). With the used of prescribed burning, there is opportunity for specific areas to be left unburnt downslope of the burnt areas, to reduce runoff and contaminant transfer. However, there is also limited information worldwide on the use of vegetative buffers as a post-fire erosion control measure. Some research has been done on post-fire application of mulches, biosolids, or logs (e.g. Wohlgemuth 2003) but little quantitative work on buffer

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strips. Without such work, it is difficult to establish guidelines on the optimum practices for pasture burning near reservoirs or to introduce effective post-fire erosion control methods such as buffer strips.

Simulated rainfall experiments in the field were therefore used to investigate the effectiveness of buffer strips of ungrazed, natural pasture grass in reducing the overland transport of sediments and associated pollutants in the rainfall runoff following burning of pastures near Wivenhoe Dam. Wivenhoe is the major water supply reservoir for Brisbane City. Pasture buffer strips were pre-wet by simulated rainfall and then subject to runoff from plots upslope of the pasture strips which had been previously subject to burning. The study was designed to compare the amounts of sediment and dissolved or total nutrients and carbon in the runoff that exited through pasture buffers of length 5, 2, and 0 m (i.e. no buffer), in the downslope direction of flow, thus evaluating the effect of buffer length on pollutant transport. The runoff water quality was also compared between two types of pasture in the site area: pastures seldom inundated by the dam water and pastures frequently inundated.

Methods and materials

Site details

Field rainfall simulations were carried out in the Wivenhoe Dam catchment, ~70 km north-west of Brisbane. The broad catchment area of the dam is 5730 km^2, of which ~41% is used for pasture production (Healthy Waterways 2009). When at full capacity, the dam can hold 1.16×10^6 ML. The catchment is subtropical and has a mean annual rainfall of 911 mm. The site area was in Mojoo Bay (27°20′28″S, 152°34′56″E), which has slopes of 4–10% under native pasture. The geology of the site consists of flood plain alluviums, and soil profiles show little sign of pedological development. Two pasture types were evaluated. Seldom-inundated pastures (SIP) are above the full-capacity waterline of the reservoir and are currently leased as grazing land for cattle. Frequently inundated pastures (FIP) are below the maximum water mark and are grazed under licence. The SIP plots were on soils with higher sand and lower clay contents (sandy clay loams) than FIP plots (clay loams). Mean contents of total organic carbon (TOC), total nitrogen (TN), and total Kjeldahl phosphorus (TKP) in the topsoils (0–0.10 m) before burning were 14.9, 1.1, and 0.4 mg/g for SIP and 11.3, 0.5, and 0.1 mg/g for FIP plots.

Plant type and percentage cover were different for the two pasture types. Grasses on the SIP plots were sparse and close to maturity (thus dry and easy to burn). They comprised mostly purple wire grass (*Aristida ramosa var. speerosa*) and weeping lovegrass (*Eragrostris* spp). Grass cover on the FIP plots was denser, greener, and shorter, thus harder to burn; the most common grasses were black spear grass (*Heteropogon contortus*), chichory (*Cichorium intybus*), and couch (*Cynodon dactylon*). Both pasture types were under the same grazing management before being fenced off to exclude grazing. Rainfall simulation experiments, in the fenced area, were carried out 1 year after fencing, thus allowing 1 year of uninterrupted pasture regrowth before testing. Regular grazing of SIP before fencing of the experimental site, and the lower moisture in the soil profile (relative to FIP), may have both contributed to reduced prevalence of the densely growing and more palatable grass species in favour of those less palatable and more sparsely growing (Tran and Gilroy 2006). Bare patches of soil were frequently seen between basal stumps of pasture on SIP plots.

Pasture burning

Burning of pastures was conducted the day before the start of rainfall simulation experiments. Due to varying plant type and amount of ground cover, there was a difference in the intensity of the burning between SIP and FIP plots. The fire was intense and fast-moving across SIP plots, whereas burning on FIP plots was slow and needed encouragement to spread, resulting in less exposed area of soil surface on the burnt plots of FIP.

Provision of buffer strips

Grass buffer strips employed on land or on the banks of waterways, with the intention of protecting water quality, are usually 10–50 m long. Providing such a strip length for rainfall simulation studies was difficult as the rain had to fall over the buffer strip as well as the experimental plot for the results to be realistic. The longest rainfall simulator that we could assemble was 12 m long, providing 10 m of uniform rain coverage. Because of this limitation, and based on our experiences in grass strip investigation under simulated rainfall (Ghadiri *et al*. 2001; Rose *et al*. 2003; Hussein *et al*. 2008), we decided to investigate the sediment trapping effectiveness of 2- and 5-m-long grass buffer strips (i.e. strips of unburnt pasture). Results obtained from burnt plots with buffer strips were compared with those from plots without such strips (i.e. buffer = 0 m). Figure 1 shows the rainfall simulator set-up on a burnt plot with no buffer. Rainfall simulation experiments were therefore carried out in two groups: one group of nine on SIP and another nine on FIP (Table 1) with three replications for each buffer length. Plots were 1.5 m wide and 5, 7, or 10 m long (Table 1). Plot slopes averaged $4.8 \pm 0.6\%$.

Rainfall simulation experiments

A 12-m-long, oscillating, spray-type simulator was used, consisting of four 3-m-long modules, equipped with measuring devices for rain and runoff rate (as described by Loch *et al*. 2001). Rainfall could thus be applied over part, or all, of the plot area. This simulator uses Veejet 80100 nozzles, which provide a high uniformity of rainfall coverage (with <5% variation across the length and width of the plots. Water for the simulator was sourced from a nearby spring. The concentration of soluble chemicals in this water was low [filterable reactive phosphate (FRP), 0.01 mg/L; NO_3^- and NO_2^- (NO_x) + ammonium (NH_4^+), 0.6 mg/L; and dissolved organic carbon (DOC), 8.7 mg/L]. A rainfall rate of 100 mm/h ($\pm 6\%$) was used, and the duration of experiments was ~30 min for no-buffer plots (Fig. 1) and ~60 min for plots with buffers, based on the speed of runoff initiation following the start of the simulated rainfall. This rainfall rate simulates a 1-in-30-year rainfall event for this area.

Rainfall simulation experiments on all plots with a 2- and 5-m buffer length were carried out for 60 min as follows. During the

Fig. 1. Rainfall simulation equipment on a burnt seldom-inundated pasture (SIP) plot with no buffer.

Table 1. Lengths of experimental plots under rainfall simulation

Burnt plot length (m)	Buffer plot length (m)	Total plot length (m)	Replicates
	Seldom-inundated pasture		
5	0	5	3
5	2	7	3
5	5	10	3
	Frequently inundated pasture		
5	0	5	3
5	2	7	3
5	5	10	3

first 30 min, only the simulator unit spaying water on the buffer strip was turned on. This served the two purposes of wetting up the buffer zone and also producing base-line information for sediment loss from unburnt plots. Sampling was done at 5-min intervals. In the second 30 min, the simulator modules producing rainfall on the upslope burnt plot were also turned on and runoff sampling from the downslope end of the buffer strips continued at the same 5-min interval for another 30 min.

Two 0.5-L samples were collected every 5–8 min during the run, one for the determination of sediment concentration and another for chemical analysis. Sampling time was recorded for all samples and used for calculating flow rate and its variation during runs. Samples were immediately stored at <4°C and were then transported to the laboratory at the Nathan campus of Griffith University for subsequent physical and chemical analyses. Composite soil samples were also taken before and after each simulation from the plots at 0–0.10 m for baseline determination of nutrient contents. While soil erosion may be expected to have more effect on the top few centimetres of soil, the 0.10 m depth ensured greater sampling reproducibility given the large variations in micro-topography/organic debris in the plots. Soils were stored, transported to the laboratory, and analysed in the same way as the sediment samples.

Laboratory analyses

The volume of the collected sediment concentration samples was first determined and then the weight of sediment (total suspended solids, TSS) was measured by oven-drying to constant weight at 105°C. Sediment concentration (mg/L) was then calculated using sample volume and sediment dry weight. Runoff rates were determined from volumes of runoff collected within the measurement times. The product of sediment concentration and runoff rate gave the sediment loss rate (SLR, kg/ha.h).

The second group of refrigerated runoff samples (chemical samples) once in the laboratory were immediately filtered. The filtrates were analysed for soluble nitrogen (NO_3^-, NO_2^-, and NH_4^+), FRP, and DOC using flow injection analysis (FIA). The sediments collected on the filter paper were analysed for total Kjeldahl nitrogen (TKN) and TKP by digestion and FIA analysis (Diamond 1996). TKN is the sum of organic N and NH_4^+-N. The TOC and total nitrogen (TN) in the sediments were analysed by an Elemental Analyser using Dumas combustion. Chemical analyses similar to those carried out on the sediments were also carried out on the soil samples collected before and after each experiment.

Statistical analyses

Data were analysed by two-factor analysis of variance using SAS (version 9.2, SAS Institute Inc., Cary, NC). Factors were pasture type (SIP, FIP) and buffer length (0, 2, or 5 m). Variables were SLR, and soluble and total nutrients/carbon concentrations in the runoff. Relationships between variables were analysed by regression analysis.

Results

Sediment transport in runoff

During the rainfall simulations, runoff from the burnt plots was observed to carry more sediment and floating debris than runoff from unburnt plots. Subsequent analysis of the sediment samples for organic/inorganic matter contents showed that up to 38% of the mass of sediment collected in the runoff samples was organic matter. The proportion of organic matter in the sediment samples varied between the two pastures and with buffer length treatments (see section below: *TKP*, *TN*, *TKN and TOC in the runoff.*) Both organic and inorganic solids can contribute to dam water pollution; therefore, they are collectively called 'sediment' in this section.

In the burnt plots, the scorched plant residue on the soil surface affected flow pattern over the plot and its ability to transport sediments. Generally, a large number of small ponds were created by floating material trapped behind grass stumps or surface irregularities. This condition prevented the development of rills, which are usually a major contributor to soil erosion by sediment entrainment on agricultural or some pasture lands, or on burnt areas (Wohlgemuth 2003). The ponded water in these small ponds also protected the soil from raindrop impact, thus reducing soil particle detachment by rain.

When buffer strips were present at the exit from test runoff plots, most of the sediment or floating material was carried towards the low slope end of the plots, and this accumulated against the first row of the unburnt plants of the buffer strip. The accumulation at the start of the buffer strips added to the efficiency of the buffer in slowing down the surface runoff, thus causing further deposition of suspended sediment in this ponded 'backwater' region (Ghadiri *et al*. 2001; Rose *et al*. 2002; Hussein *et al*. 2007). Although most sediment deposition took place in the backwater region, there was also some deposition observed inside the grass strip area, especially on the SIP plots.

Measured sediment concentrations in the runoff were not high, generally <1 g/L, and as is commonly observed, the concentration decreased with time in an approximately negative exponential form, with this decline becoming quite limited after ~30 min of rainfall. Whether soil erosion is dominated by rainfall or overland flow, the physical reasons for such decline to steady state are well understood (e. g. Sander *et al*. 1996; Rose *et al*. 2007). In contrast, and again for well-understood physical reasons (Marshall *et al*. 1996), infiltration rate under the constant-rate simulated rainfall was observed to decline with time, resulting in an increase in runoff coefficient with time, which became approximately constant after ~30 min under rainfall. The decreased sediment concentration was thus offset by the increased runoff rate so that the product of the two, SLR (kg/ha.h), was therefore approximately constant with time for most experiments. This is illustrated in Fig. 2 for the FIP 0 and 5 m buffer plots and for the SIP 5 m buffer plot. However, occasionally, there was some variation, as shown by the SIP 0-m buffer plot in Fig. 2, possibly due to the large amount of sediment loss. Since the main focus of this study is the effect of width of grass buffer strips and pasture type on sediment and chemical loss, comparisons are made in terms of the mean SLR (over the time range) for each plot.

Mean SLR from SIP burnt plots ranged from 62 to 2281 kg/ha.h, and for FIP burnt plots from 45 to 432 kg/ha.h. By example in Fig. 2, mean SLR values were 1300, 100, 240 and 170 kg/ha.h for SIP 0 and 5 m and FIP 0 and 5 m (plots 3, 8, 17 and 13) respectively. The high SLR values from SIP 0 m (up to 2070 kg/ha.h at ~7 minutes) are probably due to the lack of soil cover in this plot following burning. Mean SLR values from unburnt plots were 104 ± 44 and 161 ± 88 kg/ha.h from the SIP and FIP plots, respectively. Burning increased SLR relative to the mean SLR on unburnt plots by factors of up to 20 times on SIP plots and up to 8 times on FIP plots. Increases in sediment yield following burning have likewise been noted by many researchers (e.g. Lane *et al*. 2006; Ferreira *et al*. 2008). Johansen *et al*. (2001) measured sediment increases of up to 6 times in burned pastures compared with unburned pastures under 60 mm/h rainfall simulation on plots 3 by 10.7 m in New

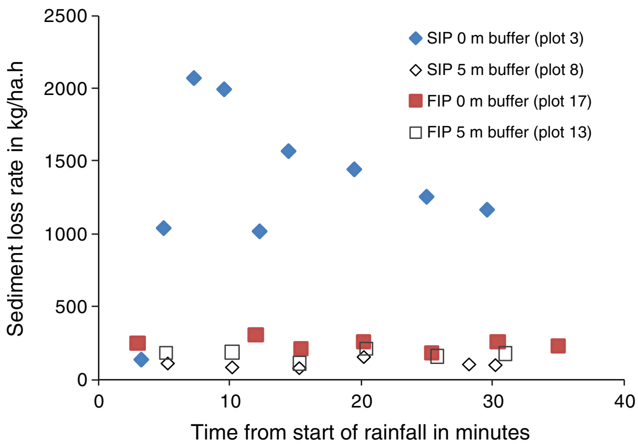

Fig. 2. Changes in sediment loss rate (SLR) with time for seldom-inundated pasture (SIP) and frequently inundated pasture (FIP) simulations for no buffer (0 m buffer) and 5 m buffer plots.

Mexico, USA. Increasing buffer length reduced SLR (Fig. 3), particularly for SIP plots, whose cover was much more open than FIP plots. Analysis of variance for SLR indicated that there were significant effects due to buffer length ($P = 0.015$) and pasture type ($P = 0.03$), and the burning method × pasture type interaction was also significant ($P = 0.039$).

Soluble chemicals in the runoff

The concentrations of the soluble chemicals, FRP, NO_x, and NH_4^+ in the runoff were generally constant during the simulation period, as illustrated in Fig. 4 for one of the SIP plots with no buffer. In contrast, DOC concentration usually decreased with time and then reached equilibrium (approximately equal to the rainfall water concentration—see **Methods**: *Rainfall simulation experiments*) at ~30 min into the simulation experiments (Fig. 4). The change with time may be due to differing solubilities of the complex carbon components remaining after burning (DeBano *et al*. 1998).

To analyse the effect of buffer length on water quality of the runoff, the mean concentrations of soluble chemicals during the 30 min of rainfall have been used. Mean FRP concentrations in the runoff from burnt plots varied between 0.25 and 2.04 mg/L and were generally higher than from unburnt SIP and FIP plots (0.31 ± 0.07 and 0.51 ± 0.18 mg/L, respectively), although three plots had slightly lower FRP concentrations, most likely due to plot variability. Miller *et al*. (2006) found that burning of forest sites likewise significantly increased levels of phosphate-P in runoff, with runoff concentrations as high as 5 mg/L. The increased FRP concentration in the runoff is due to the large amounts of highly available P that can be found in ash immediately after fires (DeBano *et al*. 1998). Buffer length had a significant effect on FRP levels ($P = 0.0001$), with most reduction within the 2-m buffer length (Fig. 5*a*). This is presumably due to adsorption of FRP to soil particles and exposed plant roots on contact during the passage of runoff through the buffer strips. Further extension of the buffer length to 5 m gave varied results, with the mean FRP concentration decreasing for the SIP and increasing for the FIP plots, possibly due to plot variability. There were no significant effects due to pasture type and no significant interactions.

Soluble nitrogen in the runoff primarily comprised NO_x (range 2.7–4.8 mg/L), as NH_4^+ concentrations were low (<0.29 mg/L). Examination of soluble NO_x components showed that nitrite (NO_2^-) levels were negligible (generally <0.01 mg/L), and the NO_x thus essentially comprised NO_3^-. The NO_x values from burnt plots were somewhat similar to those of unburnt plots (3.72 ± 0.58 and 2.82 ± 0.03 mg/L for SIP and FIP, respectively). It therefore appears that burning had little effect on NO_x levels. DeBano *et al*. (1998) reported that large amounts of ammonia-N are formed during and after burning of vegetation but that the burning does not produce nitrates. Nitrates are instead formed after burning, through nitrification of the ammonia by the soil microbial population. Any NO_x in the runoff from our study was therefore most likely from pre-existing soil/pasture nitrates. Simulations were carried out within a day of burning, which probably precluded the time for full nitrification to occur from ammonia. Buffers had no significant effect on NO_x ($P = 0.17$), and

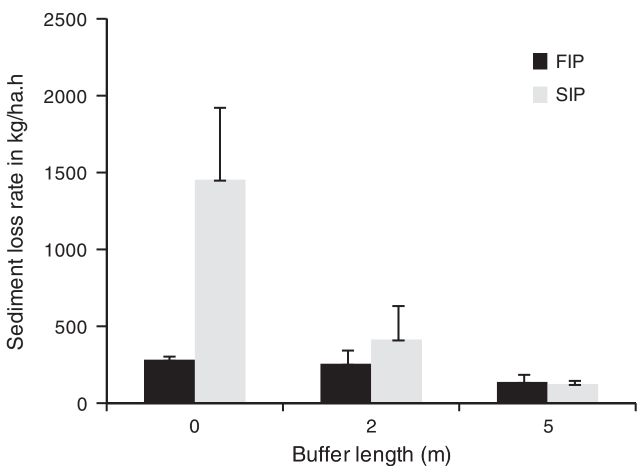

Fig. 3. Time-averaged sediment loss rate (SLR) as affected by buffer length for seldom-inundated pasture (SIP) and frequently inundated pasture (FIP). Bars indicate the standard error of the mean.

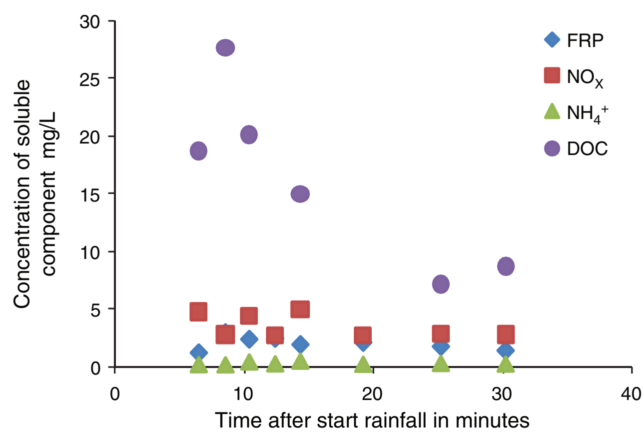

Fig. 4. Changes in concentration of filterable reactive phosphate (FRP), NO_3^- and NO_2^- (NO_x), ammonium (NH_4^+), and dissolved organic carbon (DOC) with time after start of rainfall for a seldom-inundated pasture (SIP) plot with no buffer (0 m buffer).

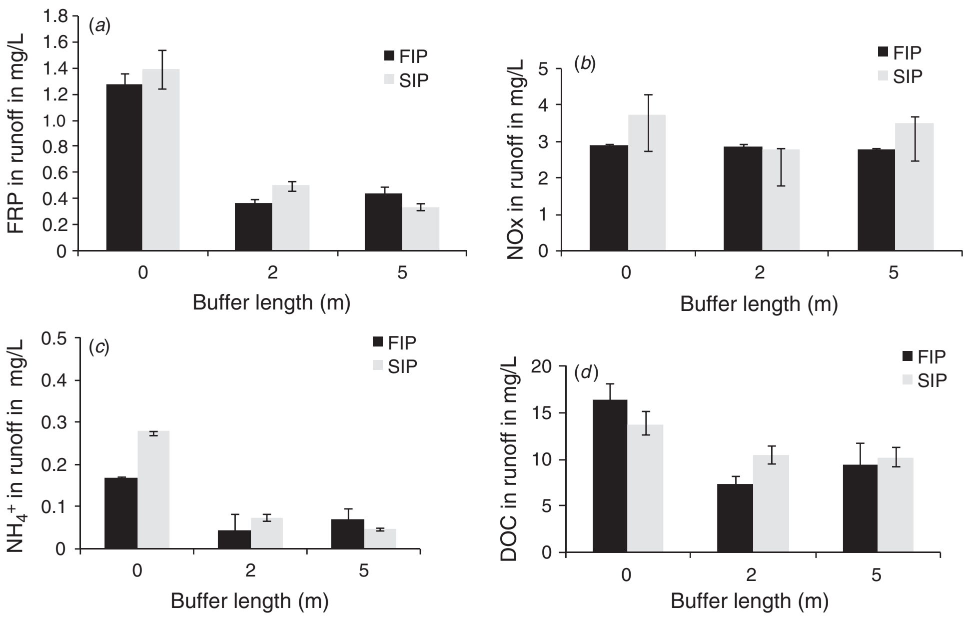

Fig. 5. Concentration of (*a*) filterable reactive phosphate (FRP), (*b*) NO_3^- and NO_2^- (NO_x), (*c*) ammonium (NH_4^+), and (*d*) dissolved organic carbon (DOC) in the runoff as affected by buffer length for seldom-inundated pasture (SIP) and frequently inundated pasture (FIP) plots. Bars indicate the standard error of the mean.

there was no clear trend in concentration as buffer length increased (Fig. 5*b*). Pasture type was significant ($P = 0.033$) but the interaction between buffer length and pasture was not.

Soluble NH_4^+ values from burnt plots ranged from 0.01 to 0.29 mg/L compared with means of 0.05 ± 0.01 and 0.09 ± 0.04 mg/L for unburnt SIP and FIP plots, respectively. In contrast to NO_x, soluble NH_4^+ concentrations in the runoff usually increased with burning. This is most likely due to increased concentrations of ammonia-N in the ash, immediately post-fire, as reported by DeBano *et al.* (1998) and Wan *et al.* (2001). The NH_4^+ concentration generally decreased with increasing buffer length (Fig. 5*c*). Buffer length and pasture type were significant for NH_4^+ ($P < 0.0001$ and $P = 0.015$), and their interaction was also significant ($P = 0.006$).

The DOC in runoff from burnt plots varied from 5.1 to 19.5 mg/L, with mean DOC values of 9.6 ± 4.8 and 10.4 ± 4.3 mg/L in unburnt SIP and FIP plots. Buffer length was significant ($P = 0.0096$); pasture type and interactions were not significant. DOC changes with buffer length were somewhat varied (Fig. 5*d*), decreasing for the 2-m buffers but increasing between the 2- and 5-m buffer lengths for the FIP plots. This may be due to gradual addition of DOC from pasture/soil in the FIP buffer zone.

TKP, TN, TKN, and TOC in the runoff

The effect of different widths of buffer strip on the runoff transport of total nutrients and carbon (TKP, TKN, TN, and TOC) was examined. The predominant component in the runoff was TOC, with concentrations ranging from 25 to 218 mg/g. Mean TOC values in runoff from unburnt SIP and FIP plots were 87 ± 30 and 58 ± 42 mg/g, respectively, thus showing a large variation between replicated plots as was found for DOC above. This was also the case for the burnt plots; nevertheless, there were significant buffer effects ($P = 0.022$) and pasture type effects ($P = 0.019$) for TOC, but the interaction was not significant. Figure 6 indicates that TOC decreased significantly for the 2-m buffer, followed by some increases for the 5-m buffer, possibly through secondary removal and transport of floating organic matter within the buffer zone.

Mean TKP in the burnt plot runoff varied between 0.09 and 1.07 mg/g, in contrast to the unburnt plots (0.35 ± 0.11 and 0.40 ± 0.39 mg/g for SIP and FIP, respectively). TKP decreased with increasing buffer length; buffer length, pasture type, and their interaction were not significant. TKN concentrations in burnt plot runoff ranged from 0.17 to 3.79 mg/g (SIP unburnt, 1.3 ± 0.83; FIP unburnt, 1.32 ± 1.57 mg/g); buffer length,

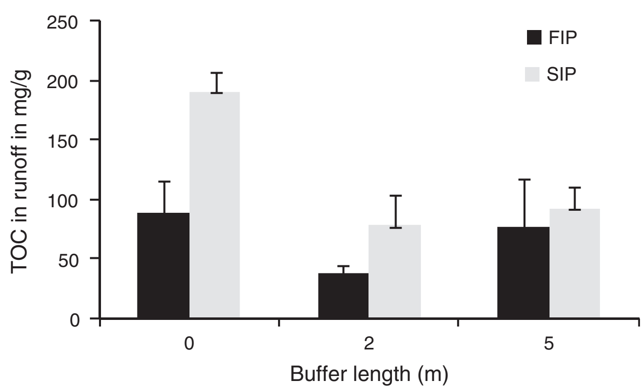

Fig. 6. Concentration of total organic carbon (TOC) in the runoff as affected by buffer length for seldom-inundated pasture (SIP) and frequently inundated pasture (FIP) plots. Bars indicate the standard error of the mean.

pasture type, and their interaction were not significant. The TN concentrations in runoff from burnt plots varied from 1.9 to 11.31 mg/g (SIP unburnt, 5.21 ± 1.52; FIP unburnt, 3.73 ± 2.55 mg/g). Again, buffer length, pasture type, and their interaction were not significant.

The total amount of these chemicals in the runoff can amount to large values when there is high sediment loss. Thomas *et al.* (1999) found an increase in the loss of total phosphorus by 3–4 orders of magnitude following forest burning. In our study, the highest SLR occurred from the hot burn SIP plot (plot 4) with no buffer (SLR = 2281 kg/ha.h). This would equate to losses of 2.1, 7.7, 18.1, and 369 kg/ha.h, respectively, of TKP, TKN, TN, and TOC under this intense rainfall, at the respective chemical concentrations for this plot. While some of this pollutant load would be reduced by further deposition downslope from the burnt plots, these high loads in the runoff from burnt pasture could still have the potential to contribute pollutant loads into the reservoir.

Effect of buffer length on reduction of SLR and soluble runoff loads

To quantify the effect of buffer length in reducing some runoff pollutants that are important for water quality, data were assessed in more detail. Figure 7 illustrates the reduction in runoff loads with respect to buffer length for SLR and some soluble parameters (means ± standard errors from the six plots at each respective buffer length are used for simplicity). The parameters show a decrease in concentration between the 0- and 2-m buffer lengths, but only SLR shows a further marked decrease from the 2- to 5-m buffer length.

Exponential curves were fitted to all data relating buffer length to concentration of the different parameters (see legends for Fig. 7*a*, *b*). The curves for SLR and soluble components (FRP, NH_4^+, and DOC) had moderate to low R^2 values ($R^2 = 0.52$, 0.59, 0.66, and 0.24, respectively) and significant F values ($P = 0.007$, 0.0002, <0.0001, and 0.04, respectively), while the curves for NO_x had non-significant F and low R^2 values. Those curves with significant F values could be used to predict runoff concentrations under the same rainfall conditions for buffers slightly longer than 5 m (see **Discussion**). However, changes in slope, soil type, or pasture vegetation could affect the runoff predictions; therefore, the equations should be used with caution.

Effect of grass buffer strips on the enrichment of sorbed chemical in runoff

Preferential transport of finer particles in runoff causes the particles and their sorbed nutrients and chemicals to be enriched in the eroded sediment compared with the original soil from which they originate (Ghadiri and Rose 1991*a*, 1991*b*; Johansen *et al.* 2001). However, such concentrations are dependent on the availability of these chemicals in the original soil. Buffer strips may be quite effective in removing the bulk of the sediment from runoff, but the fine sediment fraction, which passes through the buffer, is often richer in chemicals than the eroding soil, thus rendering the strip less effective in preventing surface water pollution than would appear. Thus, to further evaluate the effectiveness of buffer

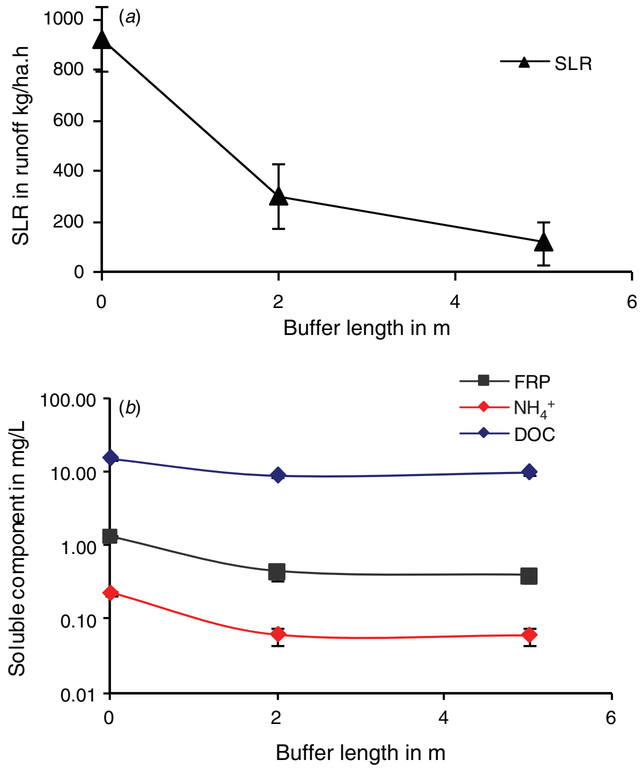

Fig. 7. (*a*) Effect of buffer length on sediment loss rate (SLR), using mean data ± standard error. Exponential regression for all data: $y = 626.4e^{-0.385x}$ where y is SLR and x is buffer length (m). (*b*) Effect of buffer length on filterable reactive phosphate (FRP), ammonium (NH_4^+), and dissolved organic carbon (DOC), using mean data ± standard error. Exponential regression for all FRP data: $y = 1.01^{-0.233x}$ where y is FRP and x is buffer length (m). Exponential regression for all NH_4^+ data: $y = 0.18^{-0.254x}$ where y is NH_4^+ and x is buffer length (m). Exponential regression for all DOC data: $y = 12.90^{-0.082x}$ where y is DOC and x is buffer length (m).

strips in reducing such pollutants, the effects of the presence and length of grass buffer strips on the enrichment of total nutrient and carbon were assessed using enrichment ratios (ER). The ER values are calculated by dividing the concentration of the parameter in the eroded sediment by the concentration of the same parameter in the original soil (see **Methods**: *Site details*). The ER values (Table 2) ranged from 3.1 to 10.7 from burnt plots without buffers. The provision of a 2-m grass buffer strip cut ER by ~30–50%, while increasing the buffer strip to 5 m further reduced ER values by only a small amount. There were significant differences for ER (t-test, $P < 0.05$) between the burnt and unburnt plots for TKP and TN, and significant

Table 2. Effect of buffer strip width on the enrichment ratio of total nutrients (Kjeldahl phosphorus, TKP; nitrogen, TN) and organic carbon (TOC) in the runoff sediment

Pasture condition	Buffer length (m)	Enrichment ratio TKP	TN	TOC
Unburnt	–	1.5	5.6	5.6
Burnt	0	3.1	9.2	10.7
Burnt	2	2.0	4.4	4.4
Burnt	5	1.7	4.2	4.3

differences between ER values from 0- and 2-m buffers for TKP, TOC, and TN, indicating that the 2-m buffer could help to reduce enrichment. Johansen *et al*. (2001) noted ER values of 2.3 and 1.2 for sorbed-caesium transport from burnt pastures in the USA but did not find significant differences between burnt and unburnt plots.

To estimate the approximate width of grass buffer capable of reducing the total nutrient and carbon to levels similar to those of unburnt soils, ER values were plotted against the buffer lengths and fitted with exponential curves. The regressions for TKP and TN were significant ($P < 0.05$), and so equations for these curves were then used to estimate the buffer length needed to lower the ER values to those of the unburnt pastures. The calculated buffer strip widths were: for TKP, 5.6 m; for TN, 2.3 m. Therefore, under these experimental conditions, the minimum grass strip required to lower all ER values to those from unburnt pastures is 6 m. Such estimates need to be treated with caution as they assume the same slope throughout and intact buffer strips, free of the effect of grazing, cattle tracks, etc.

Discussion

The burning of the pastures in this study led to enhanced sediment, nutrient, and carbon losses compared with unburnt pastures when high intensity rainfall was applied shortly after burning. The only parameters that remained relatively unaffected by burning were the components of NO_x. As outlined previously, burning of vegetation produces ammonia-N (DeBano *et al.* 1998) rather than nitrates. The nitrates may be formed later by nitrification of the ammonia, but nitrification may take from several days to several weeks depending on post-fire conditions of moisture, temperature, and ammonium concentration (e.g. Malhi and McGill 1982; Carreira *et al.* 1994; Stange and Neue 2009), with temperatures of ~30°C and moisture contents near field capacity generally being considered optimum for tropical soils. It is unlikely that nitrification occurred within one day of burning under the dry post-fire condition of this study. The NO_x in the runoff from our study was therefore most likely from pre-existing soil/pasture nitrates.

The use of bounded plots and rainfall simulation to measure post-fire erosion losses allows control over several variables affecting these losses and provides a detailed spatial analysis of the runoff processes. However, Ferreira *et al*. (2008) note that these types of experiments have some disadvantages, such as being dependent on vegetation type or hillslope position or because they are based on closed systems within the plot boundaries. Those authors suggest that nested approaches may offer more insight into the erosion processes. For this study, the location of plots on two pasture types may help to address one of the important limitations identified by Ferreira *et al*. (2008). Other research has found that plot size affects soil loss (Hudson 1993; Cerdia *et al*. 2009), with small plots often having larger and more variable soil loss than bigger plots as they exclude deposition points downslope. The effect of the buffer in modifying soil loss and runoff will therefore depend on the size of the upslope burnt area as well as other factors such as convergence of runoff. The plot size in this study was, however, constrained by the size of the rainfall simulator. Future research may employ other techniques such as use of larger plot sizes or monitored catchment measurements to enhance our understanding of the processes involved at larger scale, but in the meantime, this research provides a treatment comparison of two pasture types and buffer strip lengths on erosion losses from 5-m upslope plots.

Several researchers have noted that the intensity of burning produced differences in soil properties, vegetation cover, water repellency, erodibility, and runoff (Robichaud and Waldrop 1994; Benavides-Solorio *et al*. 2004; Coelho *et al*. 2004). DeBano *et al*. (1998) noted that the effects of burning are related to temperatures reached, length of burning, and initial soil conditions among other things. The intensity of burning was observed to be higher for the SIP than FIP plots due to the drier vegetation but was not quantified in these experiments. This higher intensity could have lead to greater ashing of vegetation, and the rapid dissolution of this ash in the runoff water could help to explain the higher concentrations of many soluble and total nutrients from the SIP plots.

Burning resulted in large increases in sediment losses in the runoff. The buffers were clearly effective in removing sediment from the runoff in both SIP and FIP plots (Fig. 3), with the most dramatic reduction occurring with a 2-m buffer for the SIP plots. Most of the coarse sediment or floating material carried from the burnt plots accumulated against the first row of the buffer strips or deposited in the backwater region. This implies that the buffer strips behaved more like ‘barrier strips’ than ‘filter strips’ for all but very fine particles, as suggested by Ghadiri *et al*. (2001) and Rose *et al*. (2002).

Concentrations of chemicals in the runoff exceeded the Australia and New Zealand Environmental Conservation Council (ANZECC) trigger values for South East Australian lakes/reservoirs, even from the unburnt plots. According to ANZECC guidelines (ANZECC, ARMCANZ 2000), trigger values are 0.01 mg N/L for NO_x, 0.01 mg N/L for NH_4^+, 0.005 mg P/L for FRP, 0.35 mg N/L for TN, and 0.01 mg P/L for TP. Runoff from burnt plots without grass buffers often exceeded these trigger values by >100 times. Ferreira *et al*. (2008) note that typically plot runoff concentrations exceed catchment concentrations by 2–9 times.

The levels of the pollutants in the runoff clearly have the potential for downslope pollution so that some management action is required. Buffer strips up to 5 m in length were not effective in reducing NO_x but appear to have some effect in removing FRP and NH_4^+ from runoff. SLR also showed a significant exponential decline pattern with strip length (Fig. 7*a*). A first-order rate equation was developed to provide an approximate prediction of the effectiveness of buffer strips longer than those investigated for removing sediment from runoff. However, such extrapolations from 2- and 5-m buffer strips to longer strips must be treated with caution, as the results are based on runoff generated from small plots, which may differ substantially in behaviour from runoff generated on large catchments of several hectares and under natural rain (Ferreira *et al*. 2008).

The concentrations of FRP and NH_4^+ also follow an exponential decline with length of buffer strip, which can be used cautiously to estimate the effectiveness of longer grass strips in removing these chemicals from runoff (Fig. 7*b*). This exercise suggests that grass buffers of ~23 and 11 m length may be capable of lowering the concentrations of FRP and NH_4^+, respectively, in the runoff from burnt pasture to a level that may not cause ecosystem stress in the dam reservoir.

Organic materials that form DOC in the runoff are responsible for degraded taste, odour, and colour of water and the formation of carcinogenic disinfection by-products during water treatment. However, there are no current trigger values for DOC. Levels of DOC in the runoff were slightly enhanced by burning and buffer strips generally reduced DOC levels.

Burning increased the concentrations of TKP, TKN, TN, and TOC in the runoff sediment. The predominant component among these was TOC. The grass buffers reduced concentrations of all these parameters, with most reduction occurring within the first 2 m of buffer, particularly for TOC and TN. Further reductions within the 5-m buffer were only achieved with the TKN and TKP. This relative independence of buffer length is because a large proportion of eroded sediment, consisting mainly of floating organic residue, accumulated against the first row of grass in the buffer strip. As shown in detailed studies by Ghadiri *et al*. (2001) and Rose *et al*. (2002, 2003), this is also due to the reduction in flow velocity (favouring net deposition) in a backwater region caused by the buffer strip.

When comparing the absolute losses of nutrients and carbon, the losses of total components in the sediment were generally greater than losses of the soluble forms. For example, using data from the plot with the highest sediment loss (SIP plot 4, no buffer), average sediment concentration was 5.12 g/L. For a hypothetical runoff of 200 L (about half of the total applied in the rainfall), this would amount to a total sediment loss of 1024 g. With an average TKP concentration of 0.94 mg/g, a loss of 962.6 mg/200 L would occur in comparison to the total FRP amount of 408 mg/200 L (average FRP concentration of runoff 2.04 g/L $\times$ 200 L). Likewise, for a low sediment loss plot (SIP plot 9, 5-m buffer), a TKP loss of 7.2 mg would occur in contrast to 5.4 mg FRP/200 L. Thus, in this study, the sediment trapping efficiency of the buffer is likely to have a greater overall effect on the pollutant transport rather than reduced transport of soluble nutrients.

Conclusion

Under high-intensity simulated rainfall, pasture buffer strips were very effective in forcing the deposition of influent sediments, including floating organic debris, from runoff following burning. ANOVA indicated that buffer length was significant for SLR, FRP, NH_4^+, DOP, and TOC. A 2-m buffer strip was effective in removing much of the suspended sediments and soluble components except for NO_x. Increasing the buffer length to 5 m caused moderate decreases in SLR, but results were variable for the other parameters. Likewise, the ER values were decreased by 30–50% of their value within the first 2 m of the buffer, and a further increase in buffer length only reduced ER marginally. The type of pasture (SIP *v.* FIP) was significant for four of the runoff parameters (SLR, NO_x, NH_4^+, TOC).

Data from this type of plot experiment provide a very useful comparison of burning treatment effects but they should be combined with spatial and temporal data at a larger catchment level to provide a more comprehensive understanding of contaminant transport following burning.

Within these limitations, our study suggests that strips of unburnt pasture grass offer a potential tool for post-fire erosion and pollutant transport control following prescribed burning. The length and location of such pasture strips would depend on the main aim of the buffer utilisation. If the primary aim is to reduce the loss of sediment (including organic carbon, nitrogen, and phosphorus), buffer conditions such as grass thickness and height might therefore be more important than infiltration characteristics, which would affect loss of soluble components from the runoff.

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