COMPOSTS AS POST-FIRE EROSION CONTROL TREATMENTS AND THEIR EFFECT ON RUNOFF WATER QUALITY

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ABSTRACT. *Erosion from fire-damaged wildlands poses a significant water quality concern. Deprived of vegetation, runoff intensifies, which escalates exports of sediments and other pollutants. Used as mulches, composts shield the soil surface and reduce runoff by absorbing water and promoting infiltration. This field study considered three types of compost used as mulches following the controlled burn of coastal sage scrub vegetation. Nine treatments considered a coarse greenwaste compost (>9.5 mm), a fine greenwaste compost (<9.5 mm), and a biosolids co-compost, each of which was surface-applied to 2.5 and 5 cm mulch depths, along with a final treatment of incorporation of 5 cm of material into 8 cm of soil. Results were aggregated from four sequential natural storm events on 2.5:1 steepness replicated plots, with the runoff sampled for sediment, nutrients, and metals. A novel non-parametric Kaplan-Meier approach was adapted to sum metal samples falling below detection limits. Compared to untreated controls, compost use effectively controlled runoff, sediment, nutrient, and metal exports after fire removed the vegetation from the slope. Runoff, total dissolved solids (TDS), total suspended solids (TSS), and total solids (TS) were reduced by averages of 86%, 88%, 80%, and 97%, respectively. Suspended metals were typically reduced by 93% to 95%. Compost use also reduced turbidity and, in most cases, nutrient exports. Mulching and soil incorporation were, in general, equally effective. Applying 5 cm mulches offered no performance advantage over 2.5 cm mulches, and in the case of biosolids compost 5 cm performed less well, contributing more dissolved solids, ammonium-N, and dissolved metals (Cd, Cr, Cu, and Mo) than the 2.5 cm treatment. Greenwaste compost particle size did not significantly affect runoff, and results for the coarse and fine greenwaste composts were similar. Compost mulches appear to be an effective means of reducing pollution from soils following wildfires.*

Keywords. *Composts, Erosion control, Fires, Metals, Mulches, Nutrients, Sediment, Water quality.*

In much of California and similar areas with semi-arid Mediterranean-style environments, where vegetation that develops during wet winters desiccates during dry summers, fire danger is acute (Shakesby, 2011; Westerling and Bryant, 2008). Because surface flows concentrate and travel more quickly over denuded soils, opportunities for infiltration are reduced so that runoff is both accelerated and increased (Smith et al., 2011). In addition, incompletely combusted soil litter and organic matter can condense to form hydrophobic layers within soils, further reducing its infiltration potential (Huffman et al., 2001), although this effect varies spatially, does not always occur, and gradually decreases when present (Larsen et al., 2009; Pierson et al., 2008). Vegetation loss may also increase runoff if transpiration rates are reduced (Shakesby, 2011).

Soil erosion and pollution exports are also increased after fires. Exposed soil particles are more easily dislodged by falling rain, and increased runoff rates encourage both rill and interill erosion (Bradford et al., 1987; Dúran Zuazo et al., 2009). Sediments from this erosion, which may fill downstream reservoirs and damage aquatic habitats, are a principal regulatory concern following wildfires (Debano and Conrad, 1978; Larsen et al., 2009; Minear and Kondolf, 2009). Chemical pollutants are also associated with both sediments and runoff water. Although fires volatilize N from plants and soils, elevated ammonia and nitrate concentrations in runoff are common after burns (Debano and Conrad, 1978; Debano et al., 1998; Meixner and Wohlgemuth, 2004). Following wildfires, plant phosphorus (P) stores are concentrated in ash deposits that can be carried off-site in runoff (Debano et al., 1998; Lane et al., 2008). Both N and P can eutrophy surface waters (Carpenter et al., 1998), while nitrate increasingly pollutes groundwater in much of California (Scott, 2010). Salts, which leach readily, are receiving increasing regulatory attention in California because they are accumulating in groundwater to levels known to depress the yields of sensitive crops (Schoups et al., 2005). There is less information as to the fate of heavy metals following wildfires, although copper (Cu), mercury (Hg), and zinc (Zn) may be mobilized by combustion (Auclair, 1977; Burke et al., 2010).

Pollutant exports can be managed by reducing runoff

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and associated erosion. Mulches immediately protect soil surfaces and improve infiltration to reduce runoff and sediment losses following wildfires, and mulching has proven to be more reliable than broadcast seeding (Bautista et al., 1996; Peppin et al., 2010; Riechers et al., 2008). Most post-fire mulch treatments rely on spread wood chips or straw, but composts applied as mulches can also be used to reduce runoff and control erosion. In studies designed to represent construction activity, compost blankets have been shown to reduce runoff volumes (Persyn et al., 2004), performing as well as or better than mulches (Faucette et al., 2005). Composting also eliminates weed seeds and plant diseases so that they are less likely to be introduced into a recovering landscape (Crohn et al., 2008; Downer et al., 2008; Eghball and Lesoing, 2000). California Department of Transportation erosion control policies recognize a number of alternatives for reducing erosion. Typical costs are included in guidance materials (Caltrans, 2013). For 2:1 slopes, ten alternatives are described, including duff ($98,840 ha^{-1}), mulch ($49,420 ha^{-1}), compost ($37,065 ha^{-1}), straw ($29,652 ha^{-1}), hydroseed ($24,710 ha^{-1}), bonded fiber matrix ($37,065 ha^{-1}), jute mesh ($29,652 ha^{-1}), coir netting ($86,485 ha^{-1}), rolled blankets ($61,775 ha^{-1}), and turf reinforcing mats ($91,427 ha^{-1}). Uncomposted mulches are more costly than compost because more material is recommended to achieve desired results. Composts are therefore economically competitive with other covers, although specific prices will vary according to the availability and proximity of materials and equipment. Compared to more inert options such as straw mats, compost blankets can foster faster development of lasting vegetation by enriching soils with organic matter and associated nutrients (Reinsch et al., 2007).

Although composts contain macro- and micronutrients that benefit plants, these nutrients can act as pollutants if exported off-site. Concentrations of some pollutants, including nutrients and heavy metals, may be increased when compost blankets are used on construction-damaged soils, but when data are adjusted using runoff volume data to reflect the mass export per unit treated area, pollution exports have been shown to decrease, sometimes dramatically (Glanville et al., 2004). Mass export is superior to concentration as an indicator of environmental impact because it specifically quantifies the pollutant load.

This study considered ten different treatments including a control (no compost) and nine permutations representing different combinations of compost feedstocks, particle sizes, and application approaches. Its purpose was to evaluate compost as a post-fire treatment for reducing pollutant exports including heavy metals, nutrients, and sediments. Four natural rain events were included in this study, and aggregate flow-weighted pollution export loads were then determined. For many samples, metals concentrations fell below detection limits. Depressed concentrations did not necessarily suggest low mass export rates because they were often associated with elevated flows, which can dilute elevated mass exports. To control bias in these cases, a non-parametric Kaplan-Meier approach was used to aggregate data when concentrations fell below detection limits.

Because many types of materials are composted and marketed, we included materials derived from greenwaste from one composter and a blend of biosolids and stable bedding residuals from another processor. Characteristics of the composts and soil are presented in tables 1 and 2, respectively. Separate treatments tested fine fractions passing through a 9.5 mm screen ("fines") and coarse material >9.5 mm ("overs") for greenwaste compost. We considered compost mulches applied to depths of 2.5 and 5 cm. Incorporation was also evaluated as an option. Incorporated treatments rototilled 5 cm of compost into 8 cm of soil. The treatments were as follows, and each treatment was replicated three times:

- GC2.5 = 2.5 cm of greenwaste compost overs (>9.5 mm).
- GC5 = 5 cm of greenwaste compost overs.
- GCI = 5 cm of greenwaste compost overs incorporated into 8 cm soil.

Table 1. Compost characteristics and salinity (EC).

Analyte	Biosolids Compost	Greenwaste Compost Coarse	Greenwaste Compost Fine
Total N (%)	2.7	0.82	1.0
Organic N (%)	2.0	0.81	0.96
NH_4^+-N (mg kg^{-1})	7200	100	390
NO_3^--N (mg kg^{-1})	0.11	14	11
Total P (mg kg^{-1})	19000	990	1400
Bulk density (g cm^{-3})	0.34	0.18	0.27
$EC_{1:5}$ (m℧ cm^{-1})	17	6.0	6.3
Organic matter (%)	62.1	79.7	67.9
Organic C (%)	27	40	37
Moisture (%)	36	6.7	41
pH	7.3	6.4	4.4
As (mg kg^{-1})	<1.0	1.6	1.9
Cd (mg kg^{-1})	7.6	<1.0	1.1
Cr (mg kg^{-1})	51	6.4	8.8
Cu (mg kg^{-1})	220	15	18
Pb (mg kg^{-1})	20	13	31
Hg (mg kg^{-1})	1.3	<1.0	<1.0
Mo (mg kg^{-1})	9.8	1.2	2.3
Ni (mg kg^{-1})	16	2.7	8.2
Se (mg kg^{-1})	9.4	<1.0	<1.0
Zn (mg kg^{-1})	410	68	49

Table 2. Soil characteristics and salinity (EC).

Analyte	Value
Total N (%)	0.064
Organic N (%)	0.063
NH_4^+-N (mg kg^{-1})	0.772
NO_3^--N (mg kg^{-1})	0.445
Total dissolved P (mg kg^{-1})	2100
CEC (meq per 100 g m^{-1})	7.8
EC_e (m℧ cm^{-1})	0.35
Organic matter (%)	0.24
Organic C (%)	0.72
Sand (%)	54
Silt (%)	28
Bulk density (g cm^{-3})	1.53
pH	7.1
As (mg kg^{-1})	4.49
Cd (mg kg^{-1})	0.18
Cr (mg kg^{-1})	31.9
Cu (mg kg^{-1})	11.7
Pb (mg kg^{-1})	6.1
Mo (mg kg^{-1})	<0.003
Ni (mg kg^{-1})	7.7
Se (mg kg^{-1})	<0.02
Zn (mg kg^{-1})	40.8

- GF2.5 = 2.5 cm of greenwaste compost fines (≤9.5 mm).
- GF5 = 5 cm of greenwaste compost fines.
- GFI = 5 cm of greenwaste compost fines incorporated into 8 cm soil.
- B2.5 = 2.5 cm of biosolids co-compost (≤9.5 mm).
- B5 = 5 cm of biosolids co-compost.
- BI = 5 cm of biosolids co-compost incorporated into 8 cm soil.
- Control = no mulch applied.

Study Site and Sampling

The study was conducted at the Citrus Research Center and Agricultural Experiment Station at the University of California, Riverside (33° 57′ 46″ N, 117° 20′ 1″ W). The 362 m^2 site was located on a uniform 2.5:1 (horizontal:vertical) slope. Vegetation was a degraded coastal sage scrub, which was thick with dry grasses. The local fire department completed a controlled burn of the area on 8 August 2009.

There is no standard plot size for compost blanket runoff experiments. Boxed experiments are typically small. Faucette et al. (2004) and Xiao et al. (2010) used 1.1 m × 0.9 m boxes and 0.9 m × 0.3 m boxes, respectively. Field plots may be of a similar scale or larger. For example, Persyn et al. (2004) and Glanville et al. (2004) used 1.5 m × 1.2 m plots and 1.8 m × 1.2 m plots, respectively. By contrast, Reinsch et al. (2007) and Bhattarai et al. (2011) used longer 12 m × 3 m plots and 10 m × 1.5 m plots, respectively. In our case, the plot sizes were determined by the available area and water storage capacity. Each treatment was replicated three times on plots installed according to a randomized complete block design. After the burn, 30 4.3 m long × 1.2 m wide plots were laid out in three rows (fig. 1). This plot size is large enough to adequately represent sheet erosion and is likely long enough to indicate the success of the material in controlling incipient rilling. Plastic edging (15 cm) was installed around each plot to a depth of 2.5 cm. At the base of each plot, 6 cm of aluminum flashing was installed to convey water to a slotted 10.2 cm (4 in.) inner diameter PVC collection pipe. Each collection pipe was then connected through a 90° elbow to additional 10.2 cm (4 in.) PVC pipe running downslope to one of 30 covered 113 L (30 gal) plastic bins at the bottom of the slope (fig. 1). The bins were fitted with lids and covered with waterproof tarps. The runoff collection system was inspected and cleaned prior to each rain event. Following each storm, the material in the bins was mixed thoroughly, and a labeled 1 L HDPE sampling bottle was quickly filled from each bin by using the sampling bottle itself to vigorously mix the water so that the water collected within the bottle was representative

Figure 1. View of installed plots.

of the mixture without. Samples were initially stored in an ice chest and were then moved to a laboratory cold room until analysis.

Rainfall data, temperature, wind speed and direction, and other weather data were obtained from the California Irrigation Management Information System (CIMIS) weather station located adjacent to the research site. Because continuous precipitation data were not available, peak intensities were determined as the largest value in the CIMIS hourly rainfall totals. Biosolids compost was spread on 30 September 2009, and greenwaste compost was applied five days later on 5 October. Total runoff was measured and sampled following four storm events:

- Event 1: 14 December 2009, a 56 h, 20.2 mm event with 2.6 mm h^{-1} peak intensity.
- Event 2: 19 January 2010, a 25 h, 29.8 mm event with 14.7 mm h^{-1} peak intensity.
- Event 3: 21 January 2010, a 45 h, 42.5 mm event with 7.6 mm h^{-1} peak intensity.
- Event 4: 23 January 2010, a 33 h, 56.7 mm event with 8.4 mm h^{-1} peak intensity.

Little precipitation and no runoff occurred between the burning of the plots and these four runoff events. The study ended in March 2010, and these four events represented the most substantial 74% of the rainfall during the entire study period and 66% of the entire rainfall during the year starting on 30 September.

Compost and Soil Analysis

Compost analysis for particle size distribution, nutrients, and total metals was conducted using appropriate TMECC standard test methods (Thompson et al., 2001). Composite soil samples were collected from the hillslope and were analyzed for their physical and chemical properties as given in table 2. Soil particle size distribution was analyzed using the hydrometer method described by Gee and Bauder (1986). Soil pH and EC measurements were made on saturated soil paste extracts (Richards, 1954) using a Con 6/TDS 6 pH and EC instrument (Oakton Instruments, Vernon Hills, Ill.). Inorganic N (NO_3^--N and NH_4^+-N) measurements were made on 2 M KCl extracts using the calorimetric technique and a Technicon AutoAnalyzer (Page et al., 1983). Organic matter content was measured with the loss on ignition method. Soil total carbon (C) and nitrogen (N) was measured on ground soil samples (passing through a 100 mesh) using the combustion method on a Thermo Electron NC analyzer. Organic N (%) was calculated by subtracting the inorganic N (%) from the total N. Soil total metals and TDP were measured using nitric acid digestion technique, USEPA Method 3051A (USEPA, 2007). In brief, the ground soil samples were treated with HNO_3 and HCl and subjected to microwave digestion. The digested samples were filtered and analyzed using inductively coupled plasma (ICP) emission spectroscopy. All soil samples were analyzed for their properties in replicates of three.

Runoff Water Analysis

Runoff was measured directly from the water collected in 113 L vessels at the bottom of each experimental plot. Turbidity and pH were measured within 48 h of collection directly on the 1 L subsample. Turbidity was measured using an HF Scientific Micro 100 Laboratory Turbidimeter (USEPA Method 180.1, without dilution). The pH was measured using USEPA Method 150.1. Total sediments (TS) were quantified by settling and evaporation (ASTM D3977-97-A). Total dissolved solids (TDS) and total suspended solids (TSS) were measured after filtering a 100 mL subsample using Millipore 47 mm glass vacuum filter holders fitted with 0.70 µm glass fiber filters. Dissolved solids were those that passed through the filter, while suspended solids were those that were retained.

Adsorbed/suspended phosphorus and trace metals were measured on extracts derived from microwave-assisted nitric acid digestion of suspended sediments using USEPA Method 3051A (USEPA, 2007). In brief, about 0.5 g of air-dried suspended sediment sample was extracted using concentrated nitric and hydrochloric acid by heating in suitable microwave equipment. The sediment sample was initially treated with 30% H_2O_2 for complete oxidation of any traces of organic matter. The treated sediment sample and acids were then transferred into a Teflon vessel, carefully sealed, and placed in a CEM microwave and heated for 10 min to complete digestion. After cooling, the contents of the vessel were filtered and diluted to a known volume. Suspended P and metals (As, Cd, Cr, Cu, Pb, Mo, Ni, Se, Zn) were then quantified (USEPA Method 200.7) using a Perkin Elmer Optima 3000DV ICP-AES (inductively coupled plasma - atomic emission spectrometry) analyzer. Results for each constituent were weighted using corresponding TSS losses for presentation in terms of units of mass per contributing area (mg m^{-2}).

To measure dissolved nutrient constituents, a 10 mL runoff water sample was frozen and later used to determine nitrate (NO_3^--N), ammonium (NH_4^+-N), orthophosphate (OP), and total dissolved P (TDP) using a Technicon AutoAnalyzer II (USEPA Methods 353.2, 350.2, 310.1, and 365.4, respectively). Another 10 mL portion of the filtered sample was stabilized with 0.1 mL nitric acid and stored at 4°C. Trace elements (As, Cd, Cr, Cu, Hg, Pb, Mo, Ni, Se, Zn) were then quantified (USEPA Method 200.7) with a Perkin Elmer Optima 3000DV ICP-AES analyzer. Detection limits (in µg L^{-1}) were As = 10, Cd = 0.4, Cr = 1, Cu = 2, Hg = 1, Pb = 20, Mo = 0.001, Ni = 0.001, Se = 20, and Zn = 3. Mercury analysis was limited to samples collected during the first rain event. All values except turbidity and pH are expressed as unit area values representing the mass of a water quality parameter exported per square meter of land. Measurements of this sort can be added to describe cumulative exports of water and pollutants as well as to compare contributions from different storm events. Cumulative losses from k events, L (mg m^{-2}), can determined as:

$$L = \sum_{n=1}^{k} \frac{C_n V_n}{A} \tag{1}$$

where C_n (mg L^{-1}) is the concentration of the pollutant in

event i runoff, V_n is the event n volume, and A (m^2) is the plot area.

Three samples were lost during event 2 due to flooding in the vicinity of the collection vessels, one each from treatments GC2.5, GFI, and B5. This presents a problem for loss measurement summations because omitting the data forces the exclusion of properly collected measurements from events 1, 3, and 4. To include all of the collected data, when summing losses we assumed that the missing data equaled the average of the other two plots' values measured during event 2. Variability between these other two plots was uniformly small, suggesting that this represents a reasonable approximation of the actual values. For pH and turbidity, which did not involve loss summations, the missing event 2 data were simply excluded from the statistical analysis. Statistically significant differences ($p < 0.05$) were evaluated by univariate ANOVA and Tukey's test with the SPSS v.20 procedure designated for randomized complete block design studies.

Summation with Non-Detects

This study emphasizes the cumulative export of contaminants during four consecutive storms, including heavy metals. However, metal concentrations often fall below detection limits, making direct summations impractical. Summations are further complicated by the flow-weighting used to calculate losses (eq. 1). One alternative for non-detects is to enter the detection limit concentration or some fraction thereof, such as one-half, as a surrogate (Glanville et al., 2004). Unfortunately, because runoff varies for different plots and storm events, the influence of the surrogate is not uniform across the plots and events. The largest events are the least precise. Correlations between runoff volumes and metal exports are exaggerated. The effective detection limit, when considered in $\mu g\ m^{-2}$, varies depending on the runoff volume. Surrogate values can dominate results if high-volume events frequently dilute metals below detection limits. Errors are compounded when non-detects are summed. Helsel (2010) considered an analogous situation in which the value of interest was the summed toxicity of various compounds, which would similarly be negatively affected by non-detects or by indiscriminate use of a surrogate. To remedy this problem, Helsel (2010) applied an innovative Kaplan-Meier approach for summing non-detects while minimizing the influence of detection levels on aggregate toxicity equivalence concentrations. Rather than making arbitrary or conservative assumptions about C_n values, this nonparametric method relies on what is observable by computing the area under a cumulative nonparametric distribution function derived from the data. No specific underlying distribution is assumed. Percentiles are determined only for detected values, but the magnitude of these percentiles is a function of both detected and non-detected data. This approach for summing non-detects is easily applied to the problem of non-detects in runoff water by substituting the runoff volume-weighted masses from equation 1 (V_nA^{-1}) for the toxicity equivalence factors in Helsel's (2010) example. We use this approach to estimate mean and standard error values for the four-event sum of exported metal masses.

Results and Discussion

Both the compost blankets and the incorporated treatments dramatically reduced runoff volumes. Total runoff from the control plots averaged 20.6 ±3.1 mm, compared to an average of 3.54 ±0.3 mm for all treated plots, an 86% reduction (table 3). Total runoff from the treatment plots ranged from 2.06 ±1.01 (B2.5) to 5.06 ±0.67 (GFI), but differences between the treatments were not statistically significant. Reductions in runoff after compost mulch applications have been noted in both field and laboratory studies using simulated rainfall (Faucette et al., 2005; Faucette et al., 2004; Glanville et al., 2004; Xiao and Gomez, 2009). However, such studies typically apply water intensely for an hour or less. Runoff reductions in this more sustained study, which was conducted under naturally occurring rainfall falling at less intense but more sustained rates, were even greater than those previously reported.

Precipitation and runoff occurred according to different patterns throughout this study. Although we did not

Table 3. Cumulative water, flow-weighted solids, and nutrient exports collected from four storms (mean ±1 standard error).[a]

Treatment	Runoff (mm)	TDS ($mg\ m^{-2}$)	TSS ($mg\ m^{-2}$)
Control	20.58 ±1.77 a	1704 ±94 a	1005 ±27.6 a
GC2.5	3.82 ±0.95 b	333 ±72 c	29.9 ±14.2 b
GC5	3.77 ±0.41 b	279 ±56 c	40.1 ±30.1 b
GCI	2.67 ±0.59 b	282 ±73 c	39.1 ±15.7 b
GF2.5	4.12 ±1.10 b	343 ±156 c	37.1 ±25.9 b
GF5	4.06 ±0.13 b	340 ±86 c	55.4 ±23.4 b
GFI	5.27 ±0.46 b	615 ±34 c	76.3 ±44.2 b
B2.5	2.06 ±0.58 b	401 ±83 c	12.9 ±3.1 b
B5	3.90 ±0.58 b	1152 ±75 b	37.2 ±0.2 b
BI	2.90 ±0.43 b	406 ±75 c	42.1 ±7.2 b
Treatment	**TS ($g\ m^{-2}$)**	**TDP ($mg\ m^{-2}$)**	**OP ($mg\ m^{-2}$)**
Control	1024 ±122 a	19.07 ±4.58 a	16.07 ±5.12 b
GC2.5	18.4 ±7.0 b	3.23 ±2.44 bc	2.63 ±1.99 b
GC5	43.1 ±29.1 b	2.40 ±1.16 c	1.87 ±1.00 b
GCI	23.3 ±4.9 b	2.56 ±1.31 c	2.06 ±1.06 b
GF2.5	31.0 ±18.0 b	3.53 ±3.09 bc	2.78 ±2.38 b
GF5	46.7 ±18.1 b	4.87 ±3.44 bc	7.10 ±8.45 ab
GFI	58.1 ±18.9 b	7.78 ±0.65 abc	6.18 ±0.43 ab
B2.5	9.1 ±4.6 b	3.63 ±1.78 bc	1.76 ±0.39 b
B5	13.0 ±2.4 b	14.95 ±10.12 ab	5.03 ±1.73 ab
BI	29.6 ±5.4 b	4.82 ±3.49 bc	3.23 ±2.06 b
Treatment	**SP ($\mu g\ m^{-2}$)**	**NO_3^--N ($mg\ m^{-2}$)**	**NH_4^+-N ($mg\ m^{-2}$)**
Control	620.3 ±112.9 a	9.97 ±5.86 a	5.65 ±2.32 b
GC2.5	31.87 ±10.36 b	2.95 ±0.09 b	3.45 ±3.04 b
GC5	39.51 ±24.75 b	1.63 ±0.29 b	0.99 ±0.56 b
GCI	3.25 ±1.93 b	2.10 ±0.75 b	1.13 ±1.04 b
GF2.5	3.23 ±2.19 b	2.42 ±0.32 b	1.27 ±0.59 b
GF5	5.66 ±2.27 b	2.39 ±2.00 b	1.27 ±0.55 b
GFI	7.00 ±3.94 b	2.56 ±0.89 b	1.43 ±0.63 b
B2.5	1.39 ±0.60 b	3.39 ±1.39 ab	14.52 ±4.95 b
B5	5.03 ±0.34 b	4.63 ±1.34 ab	83.66 ±17.79 a
BI	5.09 ±0.37 b	2.44 ±0.41 b	12.17 ±9.13 b

[a] Means within the same column followed by the same letter are not significantly different ($p < 0.5$, Tukey's test).

characterize precipitation or runoff patterns during the storms, event 1 was smallest and least intense, and the compost at the beginning of this event was relatively dry. With a total rainfall depth 20.2 mm and a peak intensity of 2.6 mm h^{-1}, event 1 contributed 14% of the total precipitation during the four storms but was responsible for only 2% to 6% of total runoff (table 4). Although event 4 was largest in magnitude, with a total rainfall depth of 56.7 mm, the second smallest storm (i.e., event 2, with 29.8 mm of rainfall) had the greatest peak intensity (14.7 mm h^{-1}) and created the most runoff. Event 2 was responsible for 42% of control runoff and an average of 40% of the runoff from the treated plots. Only 25% of control runoff resulted from event 4 (the largest storm), while an average of 39% of treated plot runoff was associated with this storm. By contrast, event 3, with 42.5 mm of rainfall, which included a brief period of hail, produced 31% of control runoff but only 16% of treatment runoff. Treatment runoff reductions ranged from a low of 62% (GF2.5, event 4) to a high of 96% (BS2.5, event 3) compared to controls. Over all four storms, treatments reduced runoff by 75% (GFI) to 90% (B2.5) and by 82.8% ±1.4% overall. However, differences between treatments were not statistically significant, suggesting that 2.5 cm blankets were as effective as both incorporated and unincorporated 5 cm treatments.

pH Values

No statistically significant pH differences were observed among the treatments or controls for individual storms (data not shown), but runoff pH for all plots averaged 6.73 ±0.05, 7.12 ±0.10, 7.23 ±0.04, and 7.19 ±0.09, respectively, for events 1 through 4. The pH of runoff from the first storm was significantly lower (paired t-test, $p < 0.005$) than from the later events. The pH values of runoff from the last three events, which did not significantly differ, were likely less affected by soil conditions, since the precipitation rates were larger relative to the first rain event. Compost use did not affect runoff pH.

Table 4. Percent of water, flow-weighted solids, and nutrient exports occurring during events 1 through 4 (14 Dec|19 Jan|21 Jan|23 Jan).

Treatment	Runoff (% of mm)	TDS (% of mg m^{-2})	TSS (% of mg m^{-2})
Control	2|42|31|25	15|44|26|16	2|48|33|18
GC2.5	6|34|19|42	11|53|16|20	8|45|21|26
GC5	6|23|21|50	10|36|22|31	4|7|20|69
GCI	6|47|18|29	13|57|16|14	5|60|15|19
GF2.5	5|33|14|47	8|51|18|24	5|65|11|20
GF5	4|52|14|29	10|60|12|18	3|66|10|22
GFI	4|55|14|27	16|60|10|13	3|60|18|20
B2.5	6|40|13|41	20|44|16|20	14|35|6|44
B5	4|32|16|48	7|58|19|16	12|56|9|22
BI	2|47|10|40	10|48|13|29	4|75|10|11
Treatment	**TS (% of g m^{-2})**	**TDP (% of mg m^{-2})**	**OP (% of mg m^{-2})**
Control	2|43|33|23	4|63|22|11	2|64|23|12
GC2.5	6|69|12|13	10|59|15|17	10|57|14|18
GC5	3|15|39|43	20|44|17|19	22|40|16|22
GCI	7|69|10|15	13|69|11|7	14|70|10|6
GF2.5	5|53|5|36	12|59|12|17	15|57|11|17
GF5	4|69|10|17	6|74|10|9	54|36|5|5
GFI	4|82|7|7	8|79|7|6	9|78|6|7
B2.5	13|69|7|11	44|28|9|19	11|45|15|29
B5	4|37|24|35	51|24|10|14	7|45|19|29
BI	1|61|6|32	21|53|8|19	4|62|10|24
Treatment	**SP (% of µg m^{-2})**	**NO_3^--N (% of mg m^{-2})**	**NH_4^+-N (% of mg m^{-2})**
Control	2|56|27|15	37|41|14|8	2|73|14|11
GC2.5	10|51|18|21	8|74|13|5	31|56|4|8
GC5	6|12|22|60	24|49|13|13	11|49|8|32
GCI	5|77|6|12	16|75|6|3	4|83|2|11
GF2.5	4|73|8|15	32|61|5|2	7|72|4|16
GF5	3|76|6|15	16|72|5|6	5|77|3|15
GFI	3|64|15|19	14|83|3|1	6|80|4|10
B2.5	14|51|4|31	26|50|15|9	11|60|9|19
B5	11|60|8|21	17|54|18|11	21|50|14|15
BI	3|80|6|10	28|49|10|13	15|53|12|20

Turbidity

Control turbidities for the four storms were 568 ±25, 708 ±10, 848 ±31, and 771 ±42 NTU, respectively. Table 5 reports turbidity values separately for each rain event, since these values are not mass-based. Corresponding treated plots averaged 101 ±19, 249 ±61, 156 ±29, and 44 ±14 NTU. Turbidity was significantly greater ($p < 0.05$) in the control plots than in any of the treatment plots during events 1, 3, and 4. Event 2 was the first large storm to occur following plot installation. Event 2 turbidity from the control plots was significantly greater than from the 5 cm and B2.5 compost blankets, but these were not significantly different from the other treatments. Overall, compost treatments reduced turbidity from the four storms by 82.3% ±3.4%, 64.8% ±8.6%, 81.6% ±3.4%, and 94.3% ±1.7%, respectively. Similar results were reported by Faucette et al. (2007), who observed a 96% reduction in turbidity with compost blankets. All compost treatments substantially improved turbidity during events 1, 2, and 4.

Salinity was assessed indirectly by measuring the electrical conductivity of the runoff water. Salinity was highest following the first storm. Biosolids composts were associated with the highest salinity concentrations, particularly the 5 cm (B5) mulch applications. However, this does not means that the biosolids composts released the most salts, as water losses from the biosolids treatments were relatively low. The mass of salts leaving a site can be calculated by multiplying observed concentrations by runoff volumes. Although flow-weighted adjustments from EC measurements may be inappropriate because no mass concentration value is involved, such adjustments are possible using total dissolved solids (TDS) concentrations.

Table 5. Runoff turbidity values (NTU) following four storms (values are means ±1 standard error).[a]

Treatment	14 Dec. 2009	19 Jan. 2010	21 Jan. 2010	23 Jan. 2010
Control	568 ±25 a	708 ±10 a	848 ±31 a	771 ±42 a
GC2.5	118 ±45 b	536 ±60 ab	143 ±44 b	32 ±4 b
GC5	100 ±33 b	149 ±8 b	78 ±14 b	38 ±3 b
GCI	167 ±35 b	379 ±55 ab	253 ±88 b	103 ±29 b
GF2.5	102 ±65 b	310 ±122 ab	87 ±51 b	142 ±120 b
GF5	127 ±54 b	197 ±83 b	236 ±134 b	31 ±10 b
GFI	172 ±37 b	341 ±134 ab	223 ±61 b	69 ±20 b
B2.5	112 ±54 b	133 ±73 b	65 ±22 b	17 ±3 b
B5	86 ±43 b	211 ±135 b	59 ±23 b	24 ±6 b
BI	130 ±56 b	443 ±19 ab	257 ±117 b	291 ±106 b

[a] Means within the same column followed by the same letter are not significantly different ($p < 0.5$, Tukey's test).

Total Dissolved Solids

We report TDS rather than salinity (EC) measurements because TDS measurements from different storms can be summed using equation 1, a property that conforms to our study design. Table 3 presents export totals across all four events. At 1704 ±163 mg m^{-2}, the controls exported significantly more TDS than any of the compost treatments. B5 releases, at 1093 ±138 mg m^{-2}, exceeded the other treatments. Even though the $EC_{1:5}$ values for the biosolids composts were almost three times those of the greenwaste compost, no incorporated or 2.5 cm mulches differed significantly (table 1). Excluding B5, compost use on average reduced TDS by 78.0% ±2.2% to an average of 372 ±32 mg m^{-2}.

Table 4 apportions these totals between the four events. Each storm removed salts from the plots so that less was available to enter later runoff. The influence of previous events on subsequent losses can be seen by comparing over time the apportioned runoff and TDS values listed in table 3. For example, in the controls, 1.8% of total runoff occurred during event 1, but 15% of total TDS escaped during that event. Corresponding values for event 4 were 25% (runoff) and 16% (TDS). The ratio of TDS to runoff for the control treatment was 8.2 (15%/1.8%) for event 1 compared to 0.62 (16%/25%) for event 4. These proportional loss ratios (PLRs) can be bracketed in the same manner as table 4 to represent runoff-normalized losses for events 1|2|3|4. The PLR bracket for the control is then 8.19|1.04|0.84|0.62. This suggests that TDS export potential from the control plots decreases strongly after the first rain, and that storm sequence is at least as important a factor as storm intensity.

The mean PLR bracket for all compost treatments was 2.43|1.29|1.03|0.53. This PLR bracket indicates that for 14 December, the proportion of total TDS leaving the treatment plots was, on average, 2.43 times greater than the proportion of runoff associated with that storm. By event 4, this proportion had fallen to 0.53, indicating that TDS losses are higher immediately after application. Presumably this is because much of the salt mass deposited in fallen ash and present in applied compost was exported by the end of the study. It seems likely that salt contributions will continue to decrease to background levels as salts are progressively removed to surface and groundwater by recurring storms.

Total Suspended Solids

The physical, chemical, biological, and aesthetic impacts of suspended solids on receiving waters make TSS measurements an important indicator of water quality (Bilotta and Brazier, 2008). Compost use dramatically reduced suspended solids export. TSS losses from the control plots totaled 1004.8 ±27.6 mg m^{-2}. Compost treatment losses were much lower than the controls, ranging from 76.3 ±44.2 mg m^{-2} (GFI) to 12.9 ±3.1 mg m^{-2} (B2.5) and averaging 41.1 ±3.3 mg m^{-2}, for an average reduction in TSS of 95.9% ±0.6%. This was consistent with the data reported by Faucette et al. (2007), wherein use of compost blankets reduced TSS by 94% after rainfall simulations on a sandy clay loam soil with high erodibility factor. No significant differences emerged between the compost treatments. Over the course of this study, compost as a protective measure improved its effectiveness relative to runoff volume. All differences were statistically significant, except for event 1 compared to events 2 and 3. B2.5 and B5 lost 14% and 12% of their total TSS during event 1, while 69% of GC5 losses occurred during event 4 (table 4). In general, losses were greatest during the intense event 2, with 51% of compost treatment losses occurring during this storm. The mean PLRs for the compost were 1.35|1.30|0.86|0.73, suggesting that runoff-normalized TSS losses declined somewhat over time.

The effectiveness of compost treatments to prevent erosion can be attributed to their ability to impede the impact energy of raindrops and their large particle size, which are relatively resistant to runoff transport compared to soil colloids (Faucette et al., 2007). However, our data indicate that TSS was not significantly influenced either by compost particle size or thickness of the compost blankets, suggesting that the effectiveness of these treatments is largely dependent on their ability to reduce raindrop impact energy and promote infiltration.

Our TSS measurements were conducted on subsamples. This approach, although standard, may underrepresent the actual suspended mass if heavier particles begin settling during subsample collection (Chan et al., 2008; Clark and Siu, 2008). Rills were not observed in the compost plots. It is also likely that the control plots, once rills developed, contributed larger particles relative to the compost plots, although we did not check for this. Our control plots were of sufficient size to develop shallow rills in the control plots, and more rills may have developed on longer plots or during more intense rainfall. Rills can, but do not necessarily, contribute larger particles than interrill flow in soils of mixed particle sizes (Alberts et al., 1980; Yang et al., 2006). The TSS losses from the control plots were much greater, and possibly larger and heavier, than from the compost plots, and any proportional underestimate of the actual suspended solids load would therefore be magnified in control plot runoff. Errors associated with the TSS method are therefore conservative with respect to our result and are unlikely to affect our conclusions.

Total Solids

Compost treatments reduced TS even more effectively than TSS. TS were reduced by an average of 97.0% ±0.5% in the compost plots compared to the controls. Control and compost treatment losses averaged 1024 ±122 and 30.3 ±3.2 g m^{-2}, respectively. All treatments performed well, reducing losses by 94.3% (GFI) and 99.1% (B2.5). Faucette et al. (2007) reported a similar 93% reduction in total solids using compost blankets. The PLR was 1.12|1.47|0.90|0.62. The intensity of event 2 led to proportionately greater TS loss:runoff ratios compared to other events. Mean PLRs following this event were improved. Our nine treatments were intended to represent a wide range of application possibilities. Although in theory composts can differ in their ability to reduce erosion (Xiao and Gomez, 2009),

under the conditions of this field study all types controlled TSS losses. This suggests that many compost types and installation techniques can be useful for controlling TSS.

Total Dissolved P

Phosphorus is a common surface water concern. Fire transfers the P from combusted plants onto the soil surface, from where it can be easily moved by runoff to surface waters, where it poses a eutrophication concern (Blake et al., 2010). Manure and biosolids-based composts can also be rich in P; the biosolids compost applied here contained 1.9% P. However, TDP losses were greatest from the control, averaging 19.1 ±4.6 mg m^{-2}. Overall, compost use was effective in reducing TDP losses compared to the control. Compost treatments reduced cumulative TDP losses by 72.2% ±6.9%, with improvements ranging from 21.6% (B5) to 87.4% (GC5). However, controls did not statistically differ from B5 (15.0 ±10.1 mg m^{-2}) or GFI (7.78 ±0.65 mg m^{-2}). Biosolids composts apparently contained P in an easily moved form. The difference in erodibility can be seen by comparing the PLR brackets of the biosolids and greenwaste compost treatments. The mean PLRs for all biosolids composts were 7.38|0.84|0.55|0.43, while for the greenwaste composts the mean PLR bracket was 2.39|1.59|0.76|0.32. Biosolids composts therefore showed a great propensity toward early losses compared to greenwaste composts, but susceptibility to losses decreased for both with each successive storm. Losses during the relatively mild event 1 were high at 1.6 ±1.3, 6.1 ±5.6, 1.0 ±0.8 mg m^{-2} for B2.5, B5, and BI, respectively, while event 1 control losses were only 0.7 ±0.2 mg m^{-2}. Glanville et al. (2004) similarly found increased TDP in runoff from unvegetated plots treated with yard waste and biosolids composts. Losses from the biosolids were highly variable; associated coefficients of variation (CVs) for event 1 alone were 1.45, 1.59, and 1.40 for these treatments, while CVs for other treatments ranged from 0.25 (GFI) to 0.60 (GCI). The variability associated with event 1 biosolids composts was not observed later. By the heavier second storm, these erodible particles were evidently reduced, as biosolids compost TDP losses were only 1.0 ±0.2, 2.9 ±1.1, and 2.5 ±1.0 mg m^{-2} for B2.5, B5, and BI, respectively, compared to a statistically greater 12.0 ±2.4 mg m^{-2} for the control.

Experience with repeated applications of biosolids composts suggests that mass exports should not be elevated after compost use (Spargo et al., 2006), and after event 1 control TDP losses were statistically greater than all compost treatments except for event 2 GFI. Except for the biosolids compost blankets, most TDP was lost during the intense event 2 (table 4). Early but transient losses from the biosolids compost applied here suggest the presence of a small number of unusually mobile P-enriched particles that were quickly lost during the first storm event. It is known that the behavior of P in different biosolids products can vary depending on how they are produced (Penn and Sims, 2002), but the production method for the particular feedstock applied here is unknown.

Orthophosphate

Control and compost plots released 16.07 ±5.12 and 3.6 ±0.7 mg m^{-2} orthophosphate (OP), respectively. Compost use reduced OP losses by 55.8% (GF5) to 89.0% (GC5), averaging 77.4% ±4.1%. OP loss patterns were similar to TDP, except that B2.5 and B5 OP values were not elevated during event 1. On average, compost treatment OP decreased steadily compared to runoff volumes; the mean compost PLRs were 2.21|1.35|0.76|0.44. By comparison, the control PLR was 1.09|1.52|0.73|0.46, and weighted losses were during event 2 when rainfall was most intense. Compost use was effective in reducing OP mass values. This is crucial in reducing the potential for eutrophication in surface waters, as OP is immediately available to algae and plants.

Suspended P

Mulch use was particularly effective in reducing SP. Losses from controls averaged 620.3 ±112.9 μg m^{-2}. After treatment, SP ranged from 39.51 ±24.75 (GC5) to 1.39 ±0.60 (B2.5). On average, compost reduced SP by more than 98%. Of the compost treatments, GC2.5 and GC5 released the most SP, but the differences between compost treatments were not statistically significant. PLR results suggest that losses from the compost treatments were somewhat greater during the first two events and then decreased: 1.37|1.51|0.67|0.59. We did not evaluate the extent to which SP is likely to become available. Combustion imposes a variety of changes to the soil environment that can increase or decrease the availability of sorbed P (Murphy et al., 2006). Although SP exports were less than TDP masses, we did not measure P adsorbed to TS, and much more P would likely be found in these settled solids. Glanville et al. (2004) reported a total adsorbed P:TDP ratio of 26:1. Compost use was quite effective in reducing SP relative to dissolved forms. On average, compost use reduced the SP:TDP in effluent to 2.1:1. Glanville et al. (2004) showed similar reductions (2.6:1) for a biosolids compost.

Nitrate-N

The soils in the study area are known to be relatively rich in NO_3^--N due to atmospheric deposition (Padgett et al., 1999). Control NO_3^--N losses averaged 10.0 ±5.9 mg m^{-2}, while mean losses from the compost treatments were 2.7 ±0.3 mg m^{-2}. Compost use reduced nitrate losses by 53.6% (B5) to 83.7% (GC5), averaging 73.0% ±3.3%. Differences between B2.5 and B5 and the control were large but were not statistically significant due to the high variability of the controls. Faucette et al. (2004) used carefully controlled laboratory conditions, including a single simulated storm event, to control variability to find that a biosolids compost yielded significantly more NO_3^--N than their control or other treatments. The first flush of event 1 was most highly enriched in nitrate. This is because nitrogen in the form of NO_3^--N is highly mobile. For the control plots, 37% of all losses occurred during the first storm and another 41% during the second (table 4). Mean exports for the compost treatments for events 1 and 2 were 20% and 62% of all

losses, respectively. Together, the final two storms accounted for just 22% and 16% of control and compost plot losses, respectively. NO_3^--N losses as a function of runoff declined quickly, suggesting that NO_3^--N losses are likely to be transitory. The mean compost NO_3^--N PLRs were 4.21|1.57|0.62|0.18.

Ammonium-N

Biosolids are enriched in NH_4^+-N, and runoff from the biosolids compost plots reflected this. The NH_4^+-N in the B5 runoff was quite elevated and was statistically greater than losses from all other treatments (table 3). Due to high variability in the control results, no other statistically significant differences emerged. Nevertheless, compared to the controls, values were greater for all biosolids compost plots but less for all greenwaste compost plots. Losses from the biosolids plots exceeded those from the control plots throughout the study, although ammonium-N releases from the compost treatments decreased more quickly than did releases from the controls. PLRs for the control, greenwaste, and biosolids plots were, respectively, 1.34|1.73|0.46|0.42, 3.25|1.36|0.76|0.46, and 2.09|1.71|0.27|0.40. Our results are consistent with Faucette et al. (2004) and Glanville et al. (2004), who reported 12-fold and 8-fold increases, respectively, in NH_4^+-N losses from biosolids blankets relative to controls. The elevated concentrations of NH_4^+-N in this study were likely due to the short time between treatment applications and intense storm events. Long-term studies would provide useful insights into subsequent N transformations and losses over time for different compost treatments.

Dissolved Metals

Dissolved metal export values are presented in table 6. This table lists the percentage of samples that were detected for each treatment under the column labeled "Detect." Brackets indicate the range of possible mean total metal values (in μg m^{-2}) including all four events. Values range from a low that assumes a zero value for non-detects to a high that substitutes the analyte detection limit for non-detects. If all samples associated with a given treatment are non-detects, the first value is 0 μg m^{-2}. If a metal is detected in all samples, the first and last values are identical. Within treatments, summary statistics are presented for each

Table 6. Cumulative dissolved metal exports. Non-detect summary statistics provided only when non-detects are <50%. "Mean ±SE" indicates mean ±1 standard error (mg m^{-2}), "Detect" indicates the percentage of samples in which the metal was detected, and "Range" was determined as [non-detects = 0, non-detects = detection limit]. Mercury (Hg) was assayed only for the first event.[a]

	As		Cd			Cr		
Treatment	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})
Control	0%	[0, 206]	-	0%	[0.0, 8.2]	-	17%	[3.4, 21.0]
GC2.5	8%	[1, 36]	-	17%	[0.3, 1.5]	-	25%	[1.0, 3.8]
GC5	0%	[0, 38]	-	8%	[0.1, 1.5]	-	42%	[2.2, 4.0]
GCI	0%	[0, 27]	-	8%	[0.4, 1.3]	-	17%	[1.0, 3.0]
GF2.5	8%	[1, 42]	-	0%	[0.0, 1.6]	-	17%	[1.4, 4.4]
GF5	17%	[2, 41]	-	25%	[1.0, 2.3]	-	25%	[1.9, 4.5]
GFI	8%	[1, 51]	-	25%	[1.6, 2.8]	-	0%	[0.0, 5.1]
B2.5	8%	[0, 21]	1.02 ±0.20 b	50%	[0.7, 1.2]	2.75 ±0.55 b	58%	[1.7, 3.0]
B5	25%	[3, 37]	2.40 ±0.25 a	83%	[2.1, 2.4]	8.76 ±0.93 a	75%	[7.1, 8.8]
BI	25%	[1, 29]	-	42%	[0.6, 1.4]	3.51 ±0.56 b	58%	[2.2, 3.6]

	Cu			Hg		Mo		
Treatment	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})
Control	216 ±9 b	100%	[216, 216]	0%	[0, 0.36]	-	33%	[4.9, 23.5]
GC2.5	77 ±14 dc	100%	[77, 77]	0%	[0, 0.20]	9.5 ±0.7 c	67%	[8.4, 10.0]
GC5	52 ±5 d	100%	[52, 52]	0%	[0, 0.22]	8.5 ±2.8 c	75%	[7.8, 9.0]
GCI	59 ±13 d	100%	[59, 59]	0%	[0, 0.16]	5.9 ±1.6 c	67%	[5.2, 6.1]
GF2.5	82 ±34 dc	100%	[82, 82]	0%	[0, 0.19]	10.9 ±3.5 c	83%	[10.3, 11.1]
GF5	185 ±83 bc	100%	[185, 185]	0%	[0, 0.17]	8.5 ±2.4 c	83%	[7.9, 8.7]
GFI	118 ±28 bcd	100%	[118, 118]	0%	[0, 0.22]	14.7 ±1.2 bc	75%	[13.7, 14.9]
B2.5	137 ±23 bcd	100%	[137, 137]	0%	[0, 0.12]	32.0 ±5.5 b	100%	[32.0, 32.0]
B5	366 ±39 a	100%	[366, 366]	0%	[0, 0.13]	134.7 ±11.2 a	100%	[134.7, 134.7]
BI	105 ±32 bcd	100%	[105, 105]	0%	[0, 0.06]	21.3 ±10.1 b	100%	[21.3, 21.3]

	Ni			Pb		Se		Zn		
Treatment	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})
Control	30.9 ±5.5 a	75%	[26.9, 32.2]	0%	[0, 411]	0%	[0, 412]	832 ±111 a	100%	[831, 831]
GC2.5	8.1 ±0.8 a	58%	[6.1, 8.2]	0%	[0, 72]	17%	[8, 73]	193 ±43 a	100%	[193, 193]
GC5	6.7 ±0.2 a	67%	[5.2, 7.0]	0%	[0, 75]	0%	[0, 75]	201 ±9 a	100%	[200, 200]
GCI	5.0 ±1.4 a	75%	[4.3, 5.1]	0%	[0, 53]	0%	[0, 53]	200 ±25 a	100%	[199, 199]
GF2.5	7.8 ±3.5 a	67%	[5.8, 8.0]	0%	[0, 82]	0%	[0, 82]	232 ±83 a	100%	[232, 232]
GF5	85.4 ±78.9 a	75%	[84.2, 85.4]	0%	[0, 81]	0%	[0, 81]	1225 ±947 a	100%	[1224, 1224]
GFI	22.7 ±3.9 a	83%	[21.7, 22.7]	0%	[0, 101]	0%	[0, 101]	466 ±14 a	100%	[465, 465]
B2.5	13.3 ±2.9 a	100%	[13.3, 13.3]	0%	[0, 41]	25%	[3, 42]	318 ±123 a	100%	[317, 317]
B5	45.0 ±2.9 a	100%	[45.0, 45.0]	0%	[0, 71]	42%	[26, 79]	501 ±49 a	100%	[501, 501]
BI	10.8 ±3.7 a	92%	[10.4, 10.8]	0%	[0, 57]	25%	[2, 59]	218 ±75 a	100%	[218, 218]

[a] Means within the same column followed by the same letter are not significantly different ($p < 0.5$, Tukey's test).

analyte that was detected in half or more (≥50%) of all samples according to the Helsel (2010) Kaplan-Meier approach. In these cases, statistically significant differences are also indicated.

Although Hg has been observed to accumulate in soils following wildfires (Burke et al., 2010), neither Hg nor Pb were detected during the study (table 6). The As and Hg ranges are therefore solely a function of the runoff rates and analyte detection limits. Because Hg was only assayed after the first storm, Hg values reflect runoff only from event 1. Except for one of the BI plots during event 3, As was only detected during event 1. No As was detected in the controls, and it was found only occasionally in the treated plots. Arsenic (As) was found in 25% of the B5 and BI samples but only in 8% of B2.5 runoff. Se was not detected in control or greenwaste compost runoff but was found in all of the event 1 biosolids composts runoff samples. Detection was sporadic by 19 January, and Se was not found in the last two events.

Cd was not detected in the control, but it was detected in most of the treatments during events 1 and 2. It was found more often in the biosolids treatments, and summary statistics were prepared for B2.5 and B5, where it was present in 50% or more of the samples. At 1.02 ±0.20 μg m^{-2}, Cd losses were statistically significantly less ($p < 0.05$) from B2.5 than from B5, which exported 2.40 ±0.25 μg m^{-2}. Cr followed a similar pattern. Most was exported during events 1 and 2. Losses from the biosolids composts were greatest and, at 8.76 ±0.93 μg m^{-2}, B5 values were larger than losses from either B2.5 or BI (2.75 ±0.55 and 3.51 ±0.56 μg m^{-2}, respectively).

Mo and Ni were detected in most samples. Ni was detected in all event 1 and 2 samples. Incidence of Mo was similar, except that it was not detected in the controls following event 2. Detection for events 3 and 4 was less frequent in the control and greenwaste compost treatments than in the biosolids composts. Although detection of Mo and Ni was somewhat less frequent with each sequential storm, overall incidents of detection exceeded 50% for all but the control Mo. No statistically significant differences between treatments were observed for Ni. Greenwaste compost treatment Mo losses did not differ significantly. B5 Mo losses, at 134.7 ±11.2 μg m^{-2}, were greater than all other treatments. Mo losses from B2.5 and BI (32.0 ±5.5 and 21.3 ±10.1 μg m^{-2}, respectively) were statistically similar to each other and were greater than the greenwaste compost treatments, although this difference was only occasionally significant.

Cu and Zn were detected in all samples. Zn losses were greatest from the control (832 ±111 μg m^{-2}) and GC5 (1225 ±947 μg m^{-2}) treatments and least from the coarse greenwaste compost treatments, which ranged from 193 ±43 (GC2.5) to 201 ±9 (GCI). There were no differences in Zn in runoff from any treatment. Cu losses were greatest from the control (216 ±9 μg m^{-2}) and B5 (366 ±39 μg m^{-2}). Coarse greenwaste values were least again, ranging between 52 ±5 (GC5) and 77 ±14 (GC2.5). B5 losses were greater than any other compost treatment, but no other statistically significant differences emerged.

Suspended Metals

Suspended metal export values are presented in table 7. This table is formatted in the same manner as table 6. No suspended As, Mo, or Se were detected during this study. Neither As nor Se was detected in dissolved form, either, although dissolved Mo was detected in most of the compost treatments and in 33% of the samples from the control plots (table 6). We did not test for suspended Hg. All Pb was detected in suspended forms. The Pb lost from the controls averaged 1,886 ±510 μg m^{-2}. This was significantly greater than the losses from the compost treatments, which ranged from 47.5 ±10.0 μg m^{-2} for B2.5 to 121.4 ±22.0 μg m^{-2} for B5. Although Pb in the biosolids compost material was greater than in the greenwaste composts, and Pb was detected most frequently in the biosolids compost treatments, suspended Pb losses from the different composts were similar.

Suspended Cd was detected in 42% of the control samples. Summary statistics were therefore not estimated, but values ranged from 33.2 to 53.9 μg m^{-2}. Cd losses were significantly greater from biosolids compost treatments B5 and BI than from other treatments. Formal statistical comparisons between the control and GC2.5, GC5, and GF 2.5 were not conducted due to low detection levels. However, the Cd export from the control was much higher than that of the green compost, as the lowest possible mean value for the control plots (33.2 μg m^{-2}) greatly exceeded the highest possible mean values for GC2.5, GC5, and GF 2.5. It should be noted that control Cd losses likely exceeded losses from all other plots, even though it was not always detected and no summary statistics were calculated. Cumulative flows from each of the control plots significantly exceeded those from the treatments even when non-detects from the control and treatment plots were assumed to be zero and the Cd detection limit, respectively. For this reason, table 7 indicates a statistical difference between the control and other treatments.

Measurable concentrations of suspended Cu, Cr, Ni, and Zn were found for all plots. Losses of these elements from the controls were, respectively, 3,136 ±982, 1,930 ±609, 942 ±221, and 12,764 ±2,374 μg m^{-2}. Losses from control plots were in all cases significantly greater than losses from the treatment plots. Copper losses from all compost treatments averaged 209 ±221 μg m^{-2}, a 93% decrease compared with the bare soil plots. Cr, Ni, and Zn mass exports were reduced by an average of 95% compared to the controls. Cr losses were reduced to an average of 104.3 ±80.6 μg m^{-2}. Ni losses were reduced to an average of 50.4 ±37.5 μg m^{-2}. Zn losses were reduced to 634 ±540 μg m^{-2}.

Metals contained in composts would be expected to partition differently from those in the control treatments. For the treatment plots, Mo was only found in dissolved form, while Pb was only measured in suspended form. Differences between the control and treatment plots are attributable to the elevated sediment loads from the control plots. Solution pH values were similar between the treatments and the controls, while the presence of organic matter in the treatment plots would normally be expected to reduce metal availability. Results are consistent with the

Table 7. Cumulative suspended metal exports. Non-detect summary statistics provided only when non-detects are <50%. "Mean ±SE" indicates mean ±1 standard error (μg m^{-2}), "Detect" indicates the percentage of samples in which the metal was detected, and "Range" was determined as [non-detects = 0, non-detects = detection limit].[a]

	As		Cd			Cr		
Treatment	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})
Control	0%	[0, 902]	-	42%	[33.2, 53.9]	1,930 ±609 a	100%	[1930, 1930]
GC2.5	0%	[0, 26]	-	17%	[0.2, 1.2]	49.8 ±24.9 b	100%	[50, 50]
GC5	0%	[0, 36]	-	25%	[0.1, 1.6]	74.3 ±49.5 b	100%	[74, 74]
GCI	0%	[0, 35]	1.45 ±0.64 c	50%	[1.2, 1.6]	106.2 ±54.4 b	100%	[106, 106]
GF2.5	0%	[0, 33]	-	25%	[1.0, 1.5]	60.0 ±39.0 b	100%	[60, 60]
GF5	0%	[0, 50]	3.57 ±1.41 c	83%	[3.4, 3.6]	136.4 ±56.8 b	100%	[136, 136]
GFI	0%	[0, 66]	2.05 ±0.42 c	67%	[1.9, 3.3]	124.4 ±62.5 b	100%	[124, 124]
B2.5	0%	[0, 11]	3.46 ±1.68 c	100%	[3.5, 3.5]	64.0 ±24. 6 b	100%	[64, 64]
B5	0%	[0, 31]	8.21 ±2.23 b	100%	[8.2, 8.2]	153.4 ±52.5 b	100%	[153, 153]
BI	0%	[0, 38]	8.26 ±2.81 b	100%	[8.3, 8.3]	170.0 ±47.8 b	100%	[170, 170]

	Cu			Mo		Ni		
Treatment	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})
Control	3,136 ±982 b	100%	[3136, 3136]	0%	[0, 301]	942 ±221 a	100%	[942, 942]
GC2.5	60 ±33 a	100%	[60, 60]	0%	[0, 8.7]	27.3 ±15.3 b	100%	[27, 27]
GC5	81 ±55 a	100%	[81, 81]	0%	[0, 12.0]	45.2 ±28.5 b	100%	[45, 45]
GCI	110 ±53 a	100%	[110, 110]	0%	[0, 11.6]	56.6 ±28.5 b	100%	[57, 57]
GF2.5	72 ±46 a	100%	[72, 72]	0%	[0, 11.0]	32.5 ±21.9 b	100%	[33, 33]
GF5	160 ±59 a	100%	[160, 160]	0%	[0, 16.5]	62.6 ±24.4 b	100%	[63, 63]
GFI	136 ±56 a	100%	[136, 136]	0%	[0, 22.1]	67.0 ±31.7 b	100%	[67, 67]
B2.5	230 ±95 a	100%	[230, 230]	0%	[0, 3.8]	26.8 ±9.7 b	100%	[27, 27]
B5	536 ±173 a	100%	[536, 536]	0%	[0, 10.3]	62.0 ±19.7 b	100%	[62, 62]
BI	494 ±156 a	100%	[494, 494]	0%	[0, 12.5]	73.8 ±19.5 b	100%	[74, 74]

	Pb			Se		Zn		
Treatment	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Detect (%)	Range (μg m^{-2})	Mean ±SE (μg m^{-2})	Detect (%)	Range (μg m^{-2})
Control	1,886 ±510 a	58%	[1768, 2568]	0%	[0, 2005]	12,764 ±2,374 b	100%	[1276, 1276]
GC2.5	68.8 ±41.1 b	50%	[53, 82]	0%	[0, 58]	292 ±102 a	100%	[292, 292]
GC5	85.7 ±56.2 b	67%	[21, 89]	0%	[0, 80]	465 ±300 a	100%	[465, 465]
GCI	113.1 ±55.6 b	67%	[97, 116]	0%	[0, 78]	516 ±264 a	100%	[516, 516]
GF2.5	-	42%	[11, 76]	0%	[0, 74]	291 ±180 a	100%	[291, 291]
GF5	85.3 ±36.5 b	58%	[119, 141]	0%	[0, 110]	607 ±219 a	100%	[607, 607]
GFI	-	42%	[95, 168]	0%	[0, 148]	545 ±253 a	100%	[545, 545]
B2.5	47.5 ±10.0 b	100%	[47, 47]	0%	[0, 25]	510 ±235 a	100%	[510, 510]
B5	121.4 ±22.0 b	100%	[108, 121]	0%	[0, 68]	1,196 ±452 a	100%	[1196, 1196]
BI	117.9 ±48.2 b	58%	[127, 139]	0%	[0, 84]	1,275 ±418 a	100%	[1275, 1275]

[a] Means within the same column followed by the same letter are not significantly different ($p < 0.5$, Tukey's test).

known solubilities of these metals in soil, which can be ordered as Mo >> Cd > Cu >Zn > As > Cr > Ni > Se >> Pb (Sauvé et al., 2000).

Conclusions

Compost blankets effectively controlled runoff, sediment, nutrient, and metal exports after fire removed coastal sage scrub from a 2.5:1 slope. Compared to the control plots, runoff was reduced by an average of 86%. TDS, TSS, and TS were reduced by averages of 88%, 80%, and 97%, respectively. Surface mulching and soil incorporation were, in general, equally effective. The 5 cm mulches offered no performance advantages over 2.5 cm for any of the treatments.

With one exception (B5, NH_4^+-N), compared to the controls no compost treatment demonstrated a statistically significant increase in the mass exports of any of the pollutants we considered. Compost particle size did not affect runoff significantly, and results for greenwaste compost "overs" (>9.5 mm) and "fines" (<9.5 mm) were similar. Significant decreases for all coarse greenwaste compost treatments were shown in TDS, TSS, TS, TDP, OP, NH_4^+-N, and NO_3-N. Fine greenwaste compost treatments performed similarly although, compared to the controls, improvements in TDP and OP, while substantial, were not always statistically significant.

Biosolids/stable bedding compost mulches also reduced TDS, TSS, and TS. In the case of biosolids compost mulches, applying to a depth of 2.5 cm offered superior performance compared to 5 cm applications. Biosolids compost applied to 5 cm depth (B5) exported more TDS and NH_4^+-N than did the 2.5 cm application, as well as more Cd, Cr, Cu, and Mo. With the possible exception of NH_4^+-N, releases from B2.5 and BI were similar to all other treatments, and biosolids composts and biosolids composts applied at these rates were suitable for erosion control. Metal losses were observed more frequently in the runoff from the biosolids compost treatments than from the greenwaste compost plots, but values were frequently below detection limits.

More metals were dissolved than were associated with suspended sediments. We used a modified Kaplan-Meier

approach to aggregate values when some observations fell below detection limits. Statistically significant differences were assessed based on statistics generated when no more than 50% of values fell below the detection limit. Research is needed to determine the influence that increasing numbers of non-detects have on the accuracy of this nonparametric approach. All compost treatments reduced losses of suspended Cd, Cu, Cr, Pb, Ni, and Zn compared to the controls. Dissolved metal losses were generally similar to the controls, although many control values were diluted below detection thresholds. Elevated flow rates in the controls often diluted metal concentrations below detection limits, but observations of metals where they were detected suggest that untreated soil metal exports were comparable to or higher than those with composts applied. Treatments with 2.5 cm of applied mulch appeared to export the least metals, but differences were generally not statistically significant due to variability.

Pollutant losses were partially a function of storm runoff intensities. Losses were generally greater during the second storm (29.8 mm over 25 h with a maximum intensity of 14.7 mm h^{-1}), since the first storm was relatively light (20.2 mm over 56 h with a maximum intensity of 2.6 mm h^{-1}). To evaluate loss patterns over time, losses were normalized to create proportional loss ratio (PLR) brackets. PLR patterns showed that runoff-normalized pollutant exports declined after the first two storms. Studies limited to one or two storm events may therefore exaggerate pollutant losses from mulched plots. Further study is needed to determine how quickly and to what extent PLRs stabilize over time. It is likely that treatment performance has further improved since the end of this study, as the plots were informally observed to naturally revegetate by the end of March 2010. Vegetation further improves the stability of slopes, although long-term research is needed to compare the performance of specific treatments after revegetation (Hansen et al., 2012). Some further losses may have occurred later in the season, but this study bracketed 73% of the precipitation falling prior to the end of March and 66% of the precipitation falling all year. Results are likely generally representative of the most significant erosion occurring during the year of study, but losses from late-season rill development in the control plots and the overall influences of plant establishment on erosion were not included in this study.

Short-term pollutant losses are controlled when composts are used as mulches because runoff is greatly reduced. Specific results will vary in different years as natural rainfall patterns vary, and results may differ for very intense events. For example, although compost mulches were retained on the slope during this study, losses would occur if surface flows were sufficient to float the material off of the slope. There is a need to characterize the specific conditions under which this becomes a concern. Further research is needed to mathematically describe the factors involved in using compost to reduce runoff.

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