Runoff, Sediment Transport, and Effects of Feral Pig (*Sus scrofa*) Exclusion in a Forested Hawaiian Watershed[1]

Dashiell O. Dunkell,[2] Gregory L. Bruland,[2,3] Carl I. Evensen,[2] and Creighton M. Litton[2]

Abstract: Browsing and trampling by nonnative feral pigs (*Sus scrofa*) negatively impact native flora and fauna in forested ecosystems and cause soil compaction. However, their impact on runoff and erosion is largely unknown. This study addressed this knowledge gap by investigating effects of feral pigs on runoff volume and total suspended solids (TSS) in runoff from the upper forested area of a Hawaiian watershed. Correlations between TSS, runoff, and other environmental variables were also examined. Runoff was collected monthly after 11 individual storm events from June 2008 to April 2009 at seven sites in the Mānoa watershed on the island of O'ahu. Each site consisted of paired runoff plots (5.04 m^2) with one plot located inside a fenced pig exclosure (exclosures 1 yr old at study initiation) and the other located in an adjacent area open to feral pigs. Forest composition and structure (stem density, stand basal area, and seedling/sapling counts) were quantified at each site. Soil moisture, throughfall, runoff volume, and TSS in runoff were sampled for each storm event. The seven sites varied considerably in terms of forest structure, with stem densities ranging from 1,500 to 9,000 stems ha^{-1} and basal areas ranging from 20 to 132 m^2 ha^{-1}. Vegetation at all sites was dominated by nonnative species. Runoff volumes from fenced and unfenced plots were highly variable, ranging from <1 to >128 liters. TSS levels in runoff ranged from <0.01 to 7.05 g $liter^{-1}$. TSS levels were generally higher in wet-season months, but this pattern was not consistent across all sites. TSS in runoff was significantly correlated with throughfall, soil moisture, and coarse woody debris cover. Although pig exclusion did not reduce TSS, significant reductions in runoff volume from pig exclusion plots were observed at one site, and two other sites showed a similar trend. Longer-term studies may reveal stronger or more consistent impacts of feral pigs. Using paired fenced versus unfenced runoff plots to study erosion impacts of feral pigs is a novel approach, and results from this study will help forest managers better understand and manage runoff and erosion dynamics.

Little is known about the impact of nonnative ungulates on soil system function, but the few studies conducted to date indicate that feral pigs (*Sus scrofa*), in particular, can have detrimental impacts on ground cover, compaction, and erosion (Kotanen 1995, Spear and Chown 2009). Some authors have speculated that a single feral pig can potentially disturb up to 200 m^2 of Hawaiian rain forest soil surface in a single day (Anderson et al. 2007), which has prompted concerns about pig impacts in these ecosystems. Feral pigs frequently feed on plants and soil invertebrates, and a large amount of rooting is required for pigs to access these food sources (Nogueira-Filho et al. 2009). There is also

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[2] Natural Resource and Environmental Management Department, University of Hawai'i at Mānoa, 1910 East-West Road, Sherman Laboratory 101, Honolulu, Hawai'i 96822.

[3] Corresponding author (phone: 808-956-8901; fax 808-956-6539; e-mail: bruland@hawaii.edu).

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speculation that pig rooting can also accelerate nutrient leaching and increase runoff and erosion (Campbell and Long 2009), but few studies have actually documented these types of effects. Browsing, trampling, and foraging by pigs can also decrease native plant biodiversity and result in the introduction and spread of exotic species. For example, in Hawai‘i feral pigs disperse *Psidium cattleianum* (strawberry guava) seeds through their feces, and their rooting behavior is believed to cause disturbances that may further enhance its spread (Huenneke and Vitousek 1990). Feral pigs also negatively affect native seedling regeneration by trampling and feeding on plants (Diong 1982).

Feral pigs have also been linked with watercourses and associated floodplains in Australia (Cowled et al. 2008). Pig foraging activities have caused changes in those areas to aquatic macrophyte communities and to the proportional amounts of open water and bare ground (Doupe et al. 2010). Specifically, the degradation of macrophyte communities and disturbance of wetland sediments by pigs increased surface water turbidity and caused prolonged anoxia and pH imbalances (Doupe et al. 2010). Habitat damage by feral pigs is also particularly problematic in other wet-forest ecosystems where plant communities and soils tend to be more sensitive to disturbance (West et al. 2009).

A number of studies of ungulate effects on runoff and stream health in continental systems have focused on large-scale agroecosystems with high densities of ungulates confined to typically small areas (Kauffman and Krueger 1984, Trimble 1994, Sarr 2002, Mieszkin et al. 2009). In those systems, swine wastes can cause water pollution and fecal contamination (Mieszkin et al. 2009). Nitrogen from pig waste was found in groundwater wells in Japan, and more heavily polluted wells tended to receive higher contributions of nitrate originating from animal wastes (Sugimoto et al. 2009). Grazing by livestock such as sheep and cattle often increases runoff and sediment and bacterial loads in streams, degrades streambanks, increases soil compaction, and alters plant and animal communities (Kauffman and Krueger 1984, Sarr 2002). Exclusion of livestock typically improves riparian-zone health and prevents further water quality degradation (Miller et al. 2010). For example, cattle caused about six times as much streambank erosion compared with that in a fenced control area, with most of the difference being from physical disturbance (Trimble 1994). These studies from agricultural watersheds are not necessarily what would be expected in tropical forested watersheds of Pacific islands where terrain is heterogeneous and feral pigs have larger ranges and occur in lower densities. In these tropical island watersheds, feral pigs are thought to be especially detrimental because they typically inhabit erosion-prone areas of steep terrain where the remaining forests are commonly found (Hone and Stone 1989). However, very few studies on the effects of feral pigs on runoff and erosion have been conducted in tropical and subtropical island forest ecosystems.

Indigenous forests on islands such as New Zealand and Hawai‘i evolved in relative isolation from major landmasses and in the absence of large mammalian herbivores. As a result, indigenous flora of these islands exhibits a high degree of endemism and is often particularly vulnerable to damage from introduced mammalian herbivory (Sweetapple and Nugent 2004). Early Polynesian settlers first introduced the Polynesian pig (likely the Asiatic form of *Sus scrofa*) to the Hawaiian Islands as a food source (Katahira et al. 1993, Noguiera et al. 2007). Later, Captain James Cook introduced the European pig during his first voyage to Hawai‘i. Many other introductions followed, and pigs became feral and dispersed throughout the Hawaiian Islands.

Currently, uncontrolled pig populations in Hawaiian rain forests are capable of doubling every four months (Katahira et al. 1993). Except for hunting pressures, malnutrition, disease, or cold weather at the highest elevations, there are no known natural factors limiting pig populations currently in Hawai‘i (Katahira et al. 1993). Although impacts of feral pigs on forest soil cover, erosion, and downslope sedimentation are thought to be severe, they have not been well studied in Hawai‘i or other tropical islands. Similarly, there is a dearth of information related to the effects of feral pigs on runoff and water quality in forested ecosystems globally.

The objectives of this study were to: (1) investigate correlations among environmental variables (slope, infiltration rate, soil moisture, stem density, etc.), runoff volume, and total suspended solids (TSS); and (2) determine if feral pig exclusion influences runoff volume and TSS. The specific hypotheses tested were (1) of all quantified environmental predictors, bare soil cover and soil moisture will have the strongest correlations with runoff volume and TSS in runoff, because ground cover influences soil erosion processes and infiltration is lower under wetter soil conditions; (2) presence of feral pigs will result in higher TSS levels in runoff compared with areas where pigs have been removed; (3) higher throughfall inputs will lead to larger runoff volumes and higher TSS levels in runoff during the wet season (November–April) than in the dry season (May–October). These hypotheses were addressed with the use of seven paired fenced/unfenced runoff plots located throughout the forested areas of the Mānoa watershed on the island of Oʻahu, Hawaiʻi. Runoff plots have been used extensively in agricultural settings (El-Swaify 1989, Mutchler and Murphree 1994), but this study is one of the first where paired runoff plots have been used in forested watershed areas to investigate the impacts of feral pigs on soil runoff and TSS. Furthermore, even though feral pig impacts have received attention from land managers, and anecdotal evidence of their impacts is widespread, there have been few attempts to quantify their impact on runoff and erosion, particularly in tropical island watersheds. Because feral pigs are invaders in ecosystems throughout the world, this research will help land managers better understand and manage for pig impacts on runoff and erosion.

MATERIALS AND METHODS

Study Area

This study was conducted in the Mānoa watershed on the island of Oʻahu, Hawaiʻi. The watershed encompasses approximately 2,528 ha, with both steeply sloping mountainous and flat coastal lands within a relatively small area (Figure 1). Runoff from the Mānoa watershed is carried by two main streams, Mānoa and Pālolo, which join before emptying into the Ala Wai Canal. Although highly developed in the lower and middle reaches, the upper reaches of the watershed are primarily forested and uninhabited by people. Feral pigs are the only large ungulates that occur in the study area, unlike many other Hawaiian watersheds where introduced deer (*Axis axis*, *Odocoileus hemionus*), cattle (*Bos taurus*), and goats (*Capra aegagrus*) can occupy the same habitat as pigs.

In spring 2007, seven forested sites were selected to investigate runoff processes and the effects of feral pigs. The sites were chosen from the upper forested areas of the watershed, based on similarities in slope, accessibility, and vegetation. Following Mutchler and Murphree (1994), a slope of ≤9% is standard in most agricultural studies that use runoff plots. However, areas in the upper forested regions of Pacific island watersheds with <9% slope are not common (the average slope of the Mānoa watershed is 47%). Using a Geographic Information System (GIS), a map of slope characteristics throughout the watershed was created, and areas with slopes between 5% and 30% were identified as potential sites (Figure 1). Assessment of slopes steeper than 30% was not attempted because runoff plots are problematic on these steeper slopes. Final site selection was based on accessibility (e.g., proximity to existing trail networks) as well as homogeneity of slope/vegetation at each site (Table 1).

Site Layout

Each forested site was approximately 10 by 10 m, with two paired 10 by 5 m plots at each site: one plot surrounded by fencing to exclude pigs and the other unfenced (Figure 2). Pig exclosures were built over a 2-month period from June to July 2007. Fences were constructed of 14-gauge (2 mm diam.) utility fencing 0.91 m tall and held in place with metal posts. Barbed wire was strung along the bottom edge of the fences to further prevent

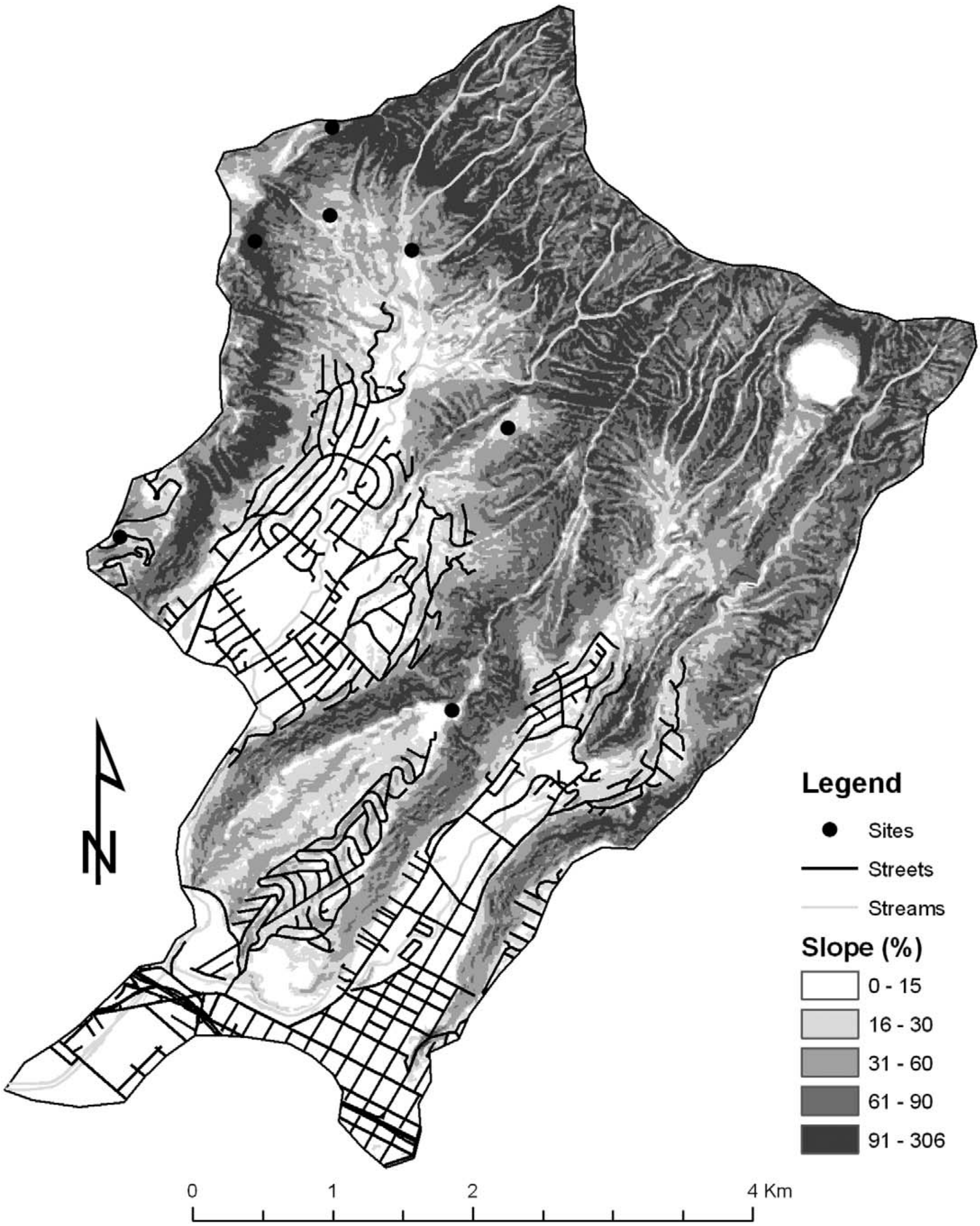

Figure 1. A GIS map of the variation in slope across the Mānoa watershed with the locations of all seven study sites.

TABLE 1

Characteristics of the Seven Study Sites in the Mānoa Watershed, Oʻahu, Hawaiʻi

Site	Elevation (m)	Slope (%)	Soil Series
Lyon (LY)	215	15.5	Lolekaa
Mānoa Cliffs (MC)	450	8.0	Rough Mountainous Land
Mānoa Falls (MF)	171	17.0	Lolekaa
Pauoa Flats (PF)	538	6.0	Rough Mountainous Land
Puʻu Pia (PP)	209	26.0	Lolekaa
Round Top (RT)	340	25.5	Tantalus
Waʻahila Ridge (WR)	340	14.0	Manana

Note: Elevation and soil series were obtained via a GIS analysis, and slopes were recorded on site using a handheld clinometer.

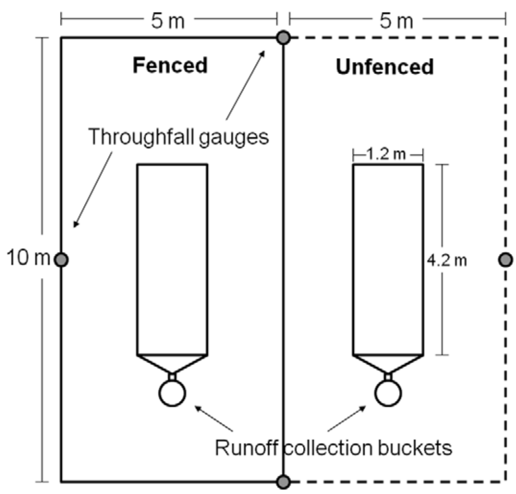

FIGURE 2. Plot layout at each of the seven sites in the Mānoa watershed. Each site was oriented so that both fenced and unfenced plots had similar slope and vegetation.

pig ingress. During initial construction of runoff plots, throughfall gauges were affixed to posts directly centered between the paired plots. In December 2008, four additional throughfall gauges were added to each site (Figure 2).

Following fence installation, runoff plots were constructed within the unfenced and fenced areas at each site. Plots were oriented downslope to effectively capture the natural path of overland flow at each site. All runoff plots were 4.2 m long by 1.2 m wide (5.04 m^2), oriented downslope. To prevent additional runoff entering from outside the plot, 15 cm tall plastic dividers were buried 7.5 cm into the soil along the upslope and outer edges of each runoff plot. These plastic pieces formed the framework that channeled runoff to a collector (El-Swaify 1989) (Figure 2). The collector was located at the downslope end of the plot, consisting of a triangular collector that funneled all runoff into a 50 cm^2 opening (Figure 2). A feed tray connected the collector to an 18.9 liter bucket for runoff storage. Watertight lids were affixed to the top of each bucket to prevent throughfall from entering directly into the bucket.

Activation Periods and Runoff Sampling

Runoff samples were collected from June 2008 to April 2009. At the beginning of each month, a 2-day dry period was identified to activate all plots and initiate runoff collection. Activation included emptying throughfall gauges, collecting soil samples for moisture analysis, and emptying and cleaning runoff collection buckets. Collection times were determined by observing weather forecasts, checking daily conditions, and monitoring the online U.S. Geological Survey (USGS) rain gauge located in Mānoa Valley. When the rain gauge recorded a >2 cm precipitation event following the activation period, collection was initiated. Collection involved measurement of total runoff and acquisition of runoff subsamples to be analyzed for TSS. During all activation/collection activities, no one ever walked inside any of the runoff plots.

Soil samples were taken within sites, but outside runoff plots so as not to disturb the vegetation, litter, and soil in the runoff plots.

Estimation of Other Environmental Variables

Other environmental parameters measured included slope (%), soil series, forest canopy and understory species composition, stem density, tree basal area, seedling/sapling counts, ground cover, soil water content, throughfall, and infiltration rate. Slopes were measured using a handheld clinometer, and no site had greater than a 2% difference between fenced and unfenced plots. Ultimately, all seven sites represented a range of different slope types (Table 1). Soil series for each site were obtained from digital Natural Resources Conservation Service soil maps and global positioning system (GPS) coordinates for each site recorded in the field. These two data layers were overlaid to identify the soil series of each site based on the latest soil survey of O'ahu (U.S. Department of Agriculture 1972). The seven sites represented four different soil series: Tantalus (Hapludands) (RT), Lolekaa (Palehumults) (LY, MF, PP), Manana (Palehumults) (WR), and Rough Mountainous Land (unclassified) (MC, PA, PF) (Table 1).

To quantify differences in forest structure and species composition across sites, a 20 by 20 m plot was centered on each pair of runoff plots, and all trees >2 cm diameter at breast height (DBH) were identified to species and measured for DBH. Stem density (number of stems ha^{-1}) was calculated by counting the total number of stems by species in the 400 m^2 plot and scaling to a hectare basis. Basal area (m^2 ha^{-1}) of each tree was calculated from individual tree DBH measurements and summed to give total basal area in the 400 m^2 plot, which was scaled to a hectare basis. Seedlings and saplings were measured in two 1 m^2 plots, at the upper and lower outside corner, within each fenced and unfenced plot. Seedlings were defined as any plant <15 cm height. Saplings were defined as plants >15 cm height and <2 cm DBH.

Ground cover at all sites was recorded for both the runoff plots and the larger fenced/unfenced area. A measuring tape was used to establish a transect line, and a visual assessment of ground cover was made at predetermined distances along the transect. For the whole plot, measurements were taken every 25 cm along two separate 10 m transects. Ground-cover measurements in the runoff plot were taken every 3 cm along three 1.2 m transects equally spaced along the runoff plot. Visual determination of ground cover was made from a top-down view above the transect. Ground cover was divided into the following categories: live plant, standing dead plant, coarse woody debris, litter, bare soil, rock, and root. Coarse woody debris was defined as woody debris >2 cm diameter. Litter included detritus, leaves, and woody debris <2 cm diameter.

Gravimetric soil moisture content was determined during the activation phase each month from both the fenced and unfenced areas. A 2 cm diameter soil corer was used to collect three separate randomly located individual samples from each area (outside runoff plots but inside the fence for fenced plots) from the upper 5 cm of the soil profile. Individual samples were composited for each treatment in each site. A standard all-weather rain gauge (Productive Alternatives, Fergus Falls, Minnesota) was used to measure throughfall (millimeters). As part of the activation process, each throughfall gauge was emptied and a thin layer of mineral oil was added to prevent evaporation before collection. Infiltration rate and the coefficient of saturation (Ksat) were determined at each plot in May 2009 using an 8 cm Tension Infiltrometer (Soil Measurement Systems, Tucson, Arizona). Equilibrium infiltration slopes were determined graphically. The slope was then multiplied by the area of the infiltrometer's base to determine the volume of water infiltrated per time. Ksat was calculated as:

$$\alpha = \frac{\text{LN(infiltration rate X/infiltration rate Y)}}{(-\text{tension X} - \text{tension Y})}$$

$$\text{Ksat} = \frac{\text{infiltration rate X}}{\text{area of base } (\alpha * \text{tension X}) * (1 + 4/\text{area of base} * \alpha)}$$

where X = tension 1, and Y = tension 2. Infiltration rate X and Y were calculated from the slope at equilibrium infiltration. Infiltration rates >4 m hr^{-1} were reported as 4.0 because values above that rate are not likely for Hawaiian soils.

Runoff Collection and Analysis

Runoff volume was measured for each rainfall event. Before any samples were removed from the collection bucket, the depth of runoff in the collection bucket was recorded and used to calculate total runoff volume. After recording the depth, the contents of the bucket were thoroughly mixed with a meterstick, and a runoff water sample was collected in an acid-washed bottle. Samples were taken to the laboratory on ice and refrigerated before analysis for TSS. TSS was measured by vacuum filtration of 100 ml of sample (Environmental Protection Agency 1971). Three blank filters were dried along with samples as controls, and the mean weight of control filters was used to correct the sample weights.

Statistical Analyses

All statistical analyses were performed using SAS version 9.1 (SAS Institute, Cary, North Carolina). A repeated measures analysis of variance (ANOVA) (Proc MIXED with a restricted/residual maximum likelihood [REML] estimation method) was used to distinguish differences over time (e.g., across months), among sites, and between fenced and unfenced treatments. Proc MIXED uses the "containment method," and, depending on the homogeneity of variances, denominator degrees of freedom may vary from one model to another. Post hoc comparisons of means were conducted with the least squares method. A linear model GLM ANOVA was used to distinguish differences between treatments and among sites for environmental variables that were only measured once. In these cases, a one-way ANOVA was used and post hoc comparisons of means were carried out using the Duncan's multiple range test. A Spearman correlation was used to evaluate associations among TSS and all environmental variables. A multiple stepwise regression (MSR) was also used to determine the best predictor(s) of TSS in runoff. Due to high variability inherent in the experimental setting, results were considered significant at $\alpha = .1$.

RESULTS

Forest Structure Characterization

Across sites, stem density ranged from <1,500 to >9,000 stems ha^{-1} (Table 2). Basal area ranged from ~20 to >132 m^2 ha^{-1}. The vast majority of woody plants found in all sites were nonnative, and the only two native tree species recorded (*Pisonia umbellifera*, *Hibiscus arnottianus*) were found at a single site (MC). A total of 14 different canopy tree species were observed across the seven sites, with each site containing different species composition. Only two sites had the same dominant canopy tree species (PP, RT [Table 2]). The exotic *Psidium cattleianum* tended to form dense monocultures of small trees and was present at a majority of sites. Two other nonnative trees, *Schefflera actinophylla* and *Cinnamomum burmannii*, also formed dense stands at several sites (PF, PP, RT).

Five woody plant species were found in the midstory canopy across all sites. *Ardisia elliptica* was the most common midstory species across sites, though typically as individuals <2.0 cm DBH. Common midstory species, especially *A. elliptica*, appeared in high numbers as saplings and seedlings across all sites. Two sites contained high seedling counts of *Cinnamomum burmannii* (MC and PF), and two other sites contained no seedlings or saplings (WR and RT). There were almost twice as many *P. cattleianum* seedlings and saplings in the unfenced plots as in the fenced plots.

Ground Cover

The following results apply only to the actual runoff plots, unless otherwise specified. Mean litter cover was 81.2% (SE = 5.1) in fenced plots and 77.9% (SE = 4.1) in unfenced plots. Mean bare soil cover was 2.9% (SE = 1.0) in fenced plots and 8.2% (SE = 4.5) in unfenced plots. Mean live plant cover was 11.0% (SE = 5.2) in fenced plots and 9.0% (SE = 3.7)

TABLE 2

Forest Structure and Ground Cover Characterization of the Study Sites in the Mānoa Watershed

Site	Stem Density (stems ha^{-1})	Basal Area (m^2 ha^{-1})	Dominant Species	Bare Soil Cover (%)	Litter Cover (%)	Rock Cover (%)	Live Plant Ground Cover (%)	Coarse Woody Debris Cover (%)
Lyon Arboretum	1,900	41.1	*Elaeocarpus grandis*	2.1	74	5.9	7.2	12
Mānoa Cliffs	3,375	20.0	*Hibiscus arnottianus*	2.6	92	0	6.0	0
Mānoa Falls	5,175	74.0	*Psidium cattleianum*	2.1	70	0	28	0
Pauoa Flats	1,475	37.7	*Elaeocarpus grandis*	1.3	72	0	24	3.4
Pu'u Pia	9,300	132.7	*Schefflera actinophylla*	6.4	79	0	3.8	11
Round Top	2,400	93.8	*Schefflera actinophylla*	22.9	75	0	1.0	1.3
Wa'ahila Ridge	4,625	47.6	*Causuarina glauca*	3.4	96	0	0.4	0.4

Note: Forest structure data were generated from 20 by 20 m plots at each site. Percentage ground cover data were generated from sampling the 5.04 m^2 runoff plots. Because no treatment differences existed, cover values are averages of the fenced and unfenced plots at each site in August 2009.

in unfenced plots. No significant differences in litter ($P = .25$), bare soil ($P = .29$), or live plant cover ($P = .83$) were observed between fenced and unfenced plots. Coarse woody debris was present at five of seven sites, and no root or standing dead cover occurred in any plots (Table 2). At one of the sites (WR), litter cover in the fenced plot was 100% due to large inputs from overstory *Causuarina glauca* trees. The unfenced runoff plot, in contrast, had <92% litter and ~7% bare soil cover, despite having the same overstory canopy as the fenced site, as a result of pig rooting to depths >5 cm.

Motion sensor cameras were deployed at four of the sites (MC, MF, RT, WR). Cameras at two sites (MC, RT) captured images of pigs in the unfenced runoff plot on multiple occasions. One site (RT) appeared to have the highest feral pig and other animal (i.e., chicken) activity, possibly associated with abundant *Persea americana* (avocado) and *Diospyros discolor* (velvet apple) trees nearby.

Soil Moisture, Throughfall, and Infiltration

Gravimetric soil moisture differed significantly across months ($P < .01$) and sites ($P < .01$) (Figure 3). However, there was also a significant site-by-month interaction ($P < .01$), indicating that differences among sites were not consistent over time. In general, soil moisture did not vary as much over time (45%–53%) as it did across sites (33%–64%). Soil moisture increased slightly as the wet season progressed. There were no significant differences in soil moisture between fenced and unfenced treatments ($P = .83$).

Throughfall amounts differed significantly with time ($F = 17.2$; df = 10, 59; $P < .001$) and across sites ($F = 4.73$; df = 6, 59; $P < .001$). The site-by-month interaction could not be tested because of a lack of replication of throughfall measurements across all sites for all events. Throughfall, measured for a single rain event each month, varied from a mean of 11.6 mm (SE = 1.3) in July to 122.3 mm (SE = 18.7) in December (Figure 4). Mean throughfall for the December event was significantly greater ($P < .01$) than that for all other months. March exhibited the next highest throughfall (89.6 mm, SE = 10.2), which was also significantly different ($P < .01$) than that of all other months. February had the second lowest mean throughfall, with 17.2 mm (SE = 3.5). Mean throughfall per event increased steadily from July through December and then decreased, with the exception of March.

Infiltration rates also varied considerably across sites, with a mean value of 1.95 m hr^{-1} (SE = 0.50). The highest values were observed for unfenced plots at three sites (MF, PP, RT) and fenced plots at three sites (LY, RT, WR).

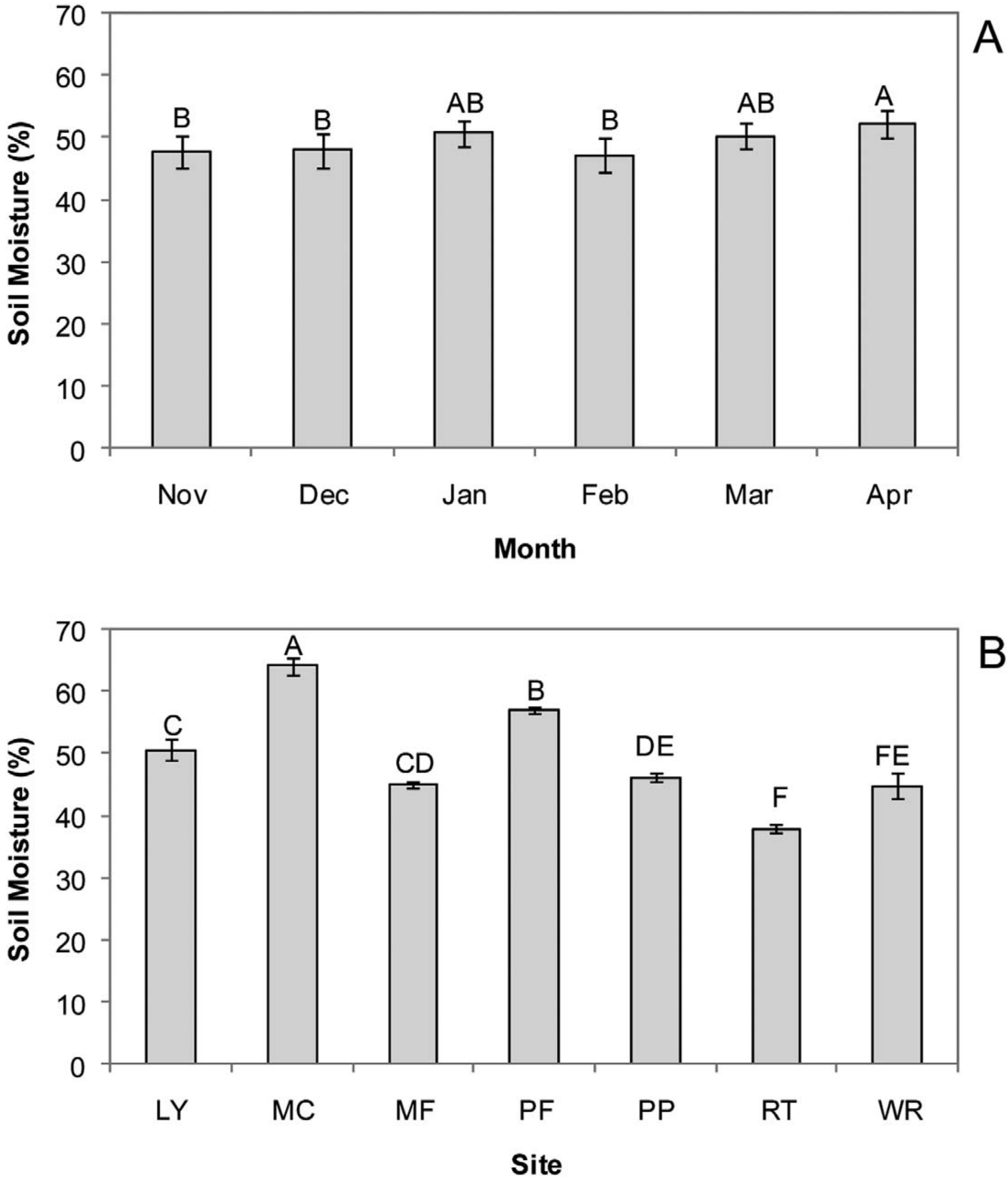

FIGURE 3. (*A*) Mean (±1 SE) gravimetric soil moisture per month before rain events in the Mānoa watershed. (*B*) Mean gravimetric soil moisture per site before rain events. Letters represent significant differences after a post hoc comparison with least squared means. Table 1 provides a key to individual sites on the *x*-axis.

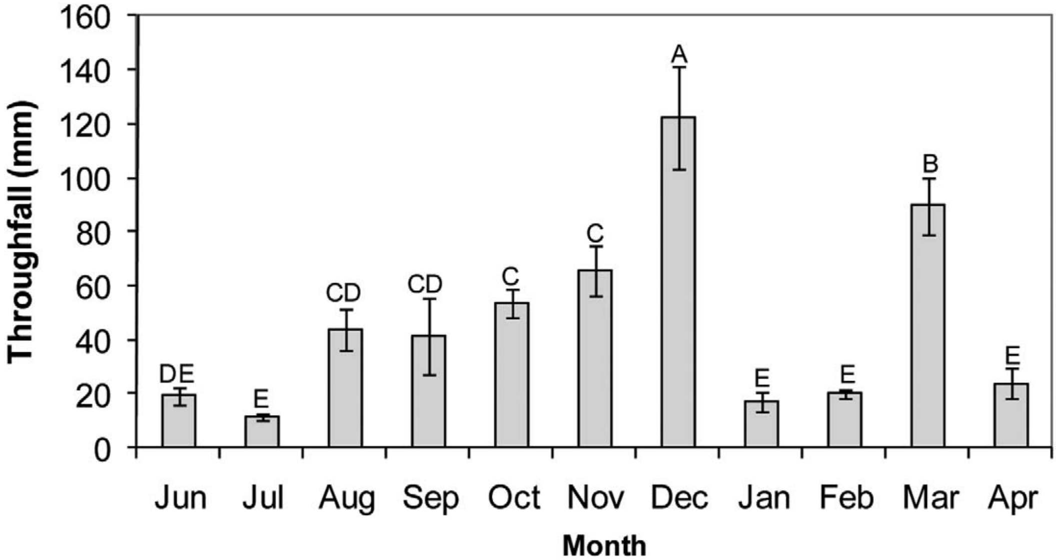

Figure 4. Mean (±1 SE) throughfall of rain events per month in the Mānoa watershed. Letters represent significant differences after a post hoc comparison with least squared means.

There were also large differences among infiltration rates between paired plots at several sites (LY, MF, WR) although fenced versus unfenced differences were not consistent.

Runoff Volume

Across the study period, runoff volumes for individual rain events ranged from <0.1 to >128 liters. Runoff volume on several occasions exceeded the bucket capacity of 13.25 liters at three of the sites even with collection feeder trays that diverted 50% of the runoff. Overflow bucket results were analyzed separately because they were installed at only two of the most accessible sites (LY, MF), though buckets in many of the other sites also overflowed in the wetter months, and during the larger rain events (i.e., December 2008) almost every site exceeded runoff bucket capacity. Thus, runoff volumes presented here are conservative estimates. During the December and March events, the site with highest average throughfall (LY) generated more than the 128 liters combined capacity of collection and overflow buckets. This was an unexpectedly high runoff volume from a 5.04 m^2 area, representing approximately 12–19% of the total throughfall volume from the December and March events. Overall, total runoff volume from individual sites varied from <1% to 16% of total throughfall volume over time. Mean runoff to throughfall ratios ranged from 2.9% to 7.8% (SE = 0.79) across sites.

The overall mean runoff volume across all sites and months was 11.0 liters (SE = 0.8). Mean values for individual sites ranged from a low of 3.8 liters to a high of 24.5 liters (Figure 5). Month ($P < .01$), site ($P < .01$), treatment ($P = .06$), the site-by-month interaction ($P < .01$), and the site-by-treatment interaction ($P = .03$) all accounted for a significant proportion of the variance in runoff volume (Table 3). The monthly pattern for runoff volume generally followed a similar trend to throughfall, with some differences. December and March (the two months with highest throughfall) had the highest runoff volume (Figure 5*A*). July (the month with the lowest throughfall) had the lowest mean runoff volume, which was also less than half the volume of that of any other month. Lyon and MC generally had higher runoff volumes than all the other sites (Figure 5*B*), although due to

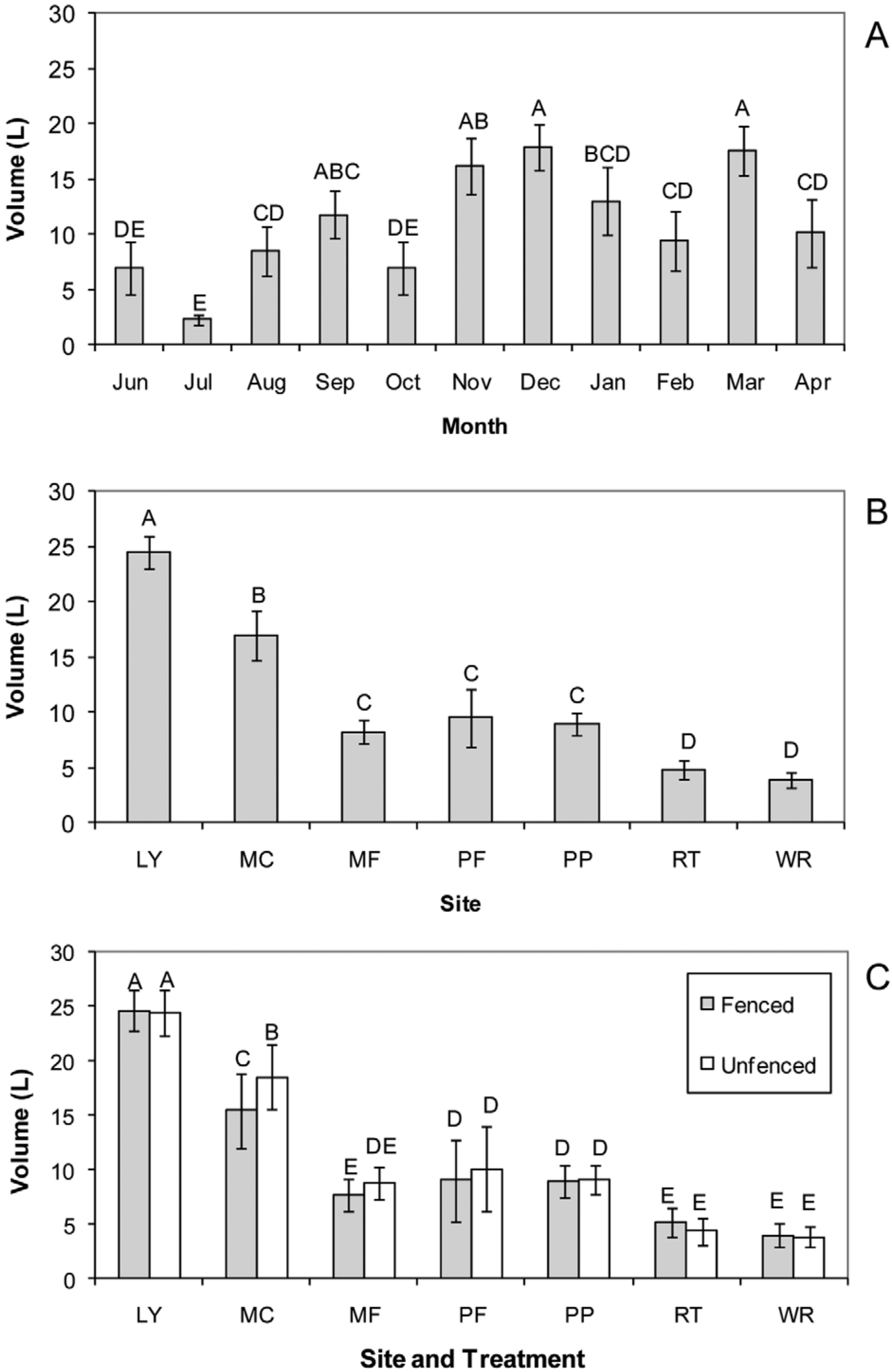

Figure 5. (*A*) Mean (±1 SE) runoff volume in liters per rain event, among months in the Mānoa watershed. (*B*) Mean (±1 SE) runoff volume across sites in liters per rain event. (*C*) Mean (±1 SE) runoff volume for the fenced (shaded bars) and unfenced (open bars) plots across sites in liters per rain event. Letters represent significant differences after a post hoc comparison with least squared means. Table 1 provides a key to individual sites on the *x*-axis.

TABLE 3

Repeated Measures ANOVA Results for Runoff Volume per Monthly Rain Event at the Seven Sites in the Mānoa Watershed (Each Site Had Both Fenced and Unfenced Treatments)

Source	Num. df	Den. df	*F*-statistic	*P*-value
Month	10	60	90.67	<.001
Site	6	60	325.75	<.001
Treatment	1	60	3.76	.057
Month × Site	60	60	20.00	<.001
Month × Treatment	10	60	0.83	.59
Site × Treatment	6	60	2.57	.03

TABLE 4

Repeated Measures ANOVA Results for Total Suspended Solids (TSS) in Runoff from Monthly Storm Events at the Seven Sites in the Mānoa Watershed (Each Site Received Both Fenced and Unfenced Treatments)

Source	Num. df	Den. df	*F*-statistic	*P*-value
Month	10	60	59.91	<.001
Site	6	1	32.46	.13
Treatment	1	1	1.40	.45
Site × Month	60	60	10.39	<.001
Treatment × Month	10	60	0.99	.46
Site × Treatment	6	1	1.44	.56

the site-by-month interaction that was not consistent across all months. For the site-by-treatment interaction (Figure 5*C*), the MC site has significantly lower mean runoff volume in the fenced plot than in the unfenced plot. Two other sites (MF and PF) showed a similar trend, although these differences were not significant. Mean runoff volumes for three sites (LY, PP, WR) were virtually identical in the fenced and unfenced plots. Values were slightly higher for the fenced plot at RT, but this difference was not significant.

TSS in Runoff

Across the study period, the mean level of TSS in runoff was 0.59 g liter^{-1} (SE = 0.09), and levels ranged from <0.01 to 7.05 g liter^{-1}. TSS in runoff differed significantly over time ($P < .001$) but not across sites or between treatments (Table 4). TSS levels were highly variable across sites, as indicated by a significant site-by-month interaction. December had the highest mean TSS in runoff (Figure 6*A*), more than double the amount of any other month. The wet season months of November, December, January, and March had higher mean TSS in runoff than any of the dry-season months. June, July, and August had the lowest mean TSS in runoff. For variability among sites, LY, MC, PF, and PP generally had higher TSS than MF, RT, and WR, although due to the month-by-site interaction, this pattern was not consistent across all months.

Correlation of TSS with Environmental Variables

A correlation analysis (Table 5) demonstrated that TSS was positively correlated with throughfall, soil moisture, and coarse woody debris cover. Coarse woody debris had the highest correlation with TSS ($r = 0.73$, $P = .01$). Runoff volume, in turn, was significantly and positively correlated with throughfall and rock cover (Table 5). Overall, the most influential variable in the correlation matrix was throughfall, because it was significantly correlated with seven other variables

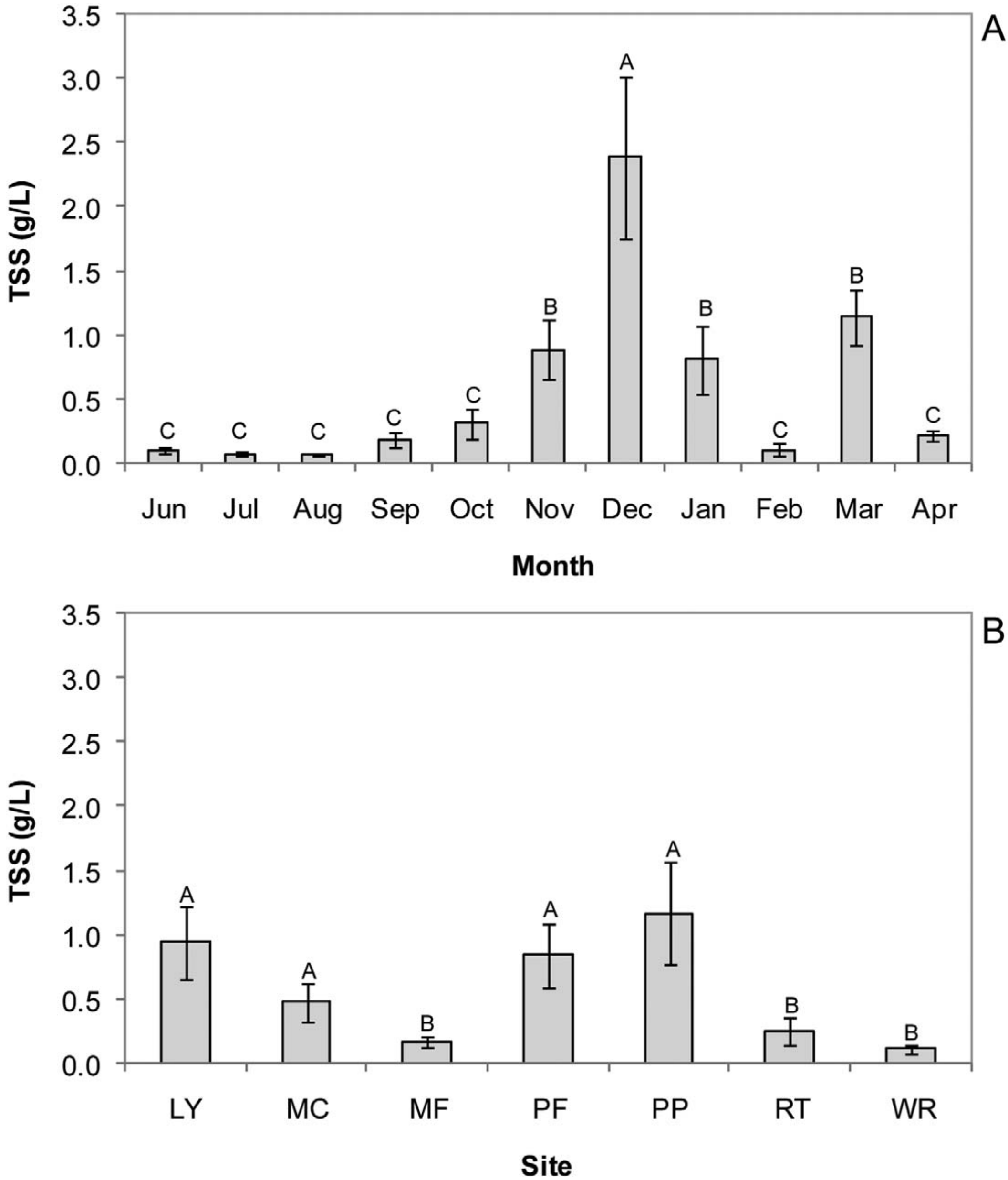

Figure 6. (*A*) Mean (±1 SE) total suspended solids (TSS) in runoff per month averaged across all sites in the Mānoa watershed. (*B*) Mean (±1 SE) TSS in runoff per site averaged across all months. Letters represent significant differences after a post hoc comparison with least squared means. Table 1 provides a key to individual sites on the *x*-axis.

TABLE 5

Spearman Correlation Matrix for Environmental Variables and Total Suspended Solids (TSS) in Runoff for the Mānoa Watershed

	TSS	Thrufall	SMoist	Litter	Bare	Rock	Live	CWD	ROVol	Stem	BA	Slope	Ksat
TSS		**0.58**	**0.58**	–0.38	0.05	0.30	0.41	**0.73**	0.42	–0.43	–0.21	–0.16	–0.24
Thrufall			**0.55**	–0.41	–0.43	**0.61**	**0.82**	0.36	**0.59**	**–0.50**	**–0.46**	–0.32	–0.17
SMoist				0.18	**–0.50**	0.20	0.43	0.13	0.38	–0.35	**–0.79**	**–0.67**	–0.38
Litter					–0.26	–0.10	**–0.51**	–0.24	–0.11	0.17	–0.29	–0.27	–0.22
Bare						–0.22	–0.42	0.15	–0.12	0.28	**0.59**	**0.62**	0.00
Rock							0.15	**0.52**	**0.61**	–0.40	–0.20	0.00	–0.04
Live								0.14	0.38	–0.21	–0.37	–0.33	–0.15
CWD									0.45	–0.17	0.21	0.24	–0.13
ROVol										0.07	–0.08	0.24	–0.05
Stem											**0.61**	**0.61**	0.17
BA												**0.93**	0.42
Slope													0.44
Ksat													

Note: Significant results shown in **boldface** ($\alpha = .10$). Abbreviations are as follows: TSS, total suspended solids; Thrufall, throughfall; SMoist, soil moisture; CWD, coarse woody debris cover; ROVol, runoff volume; Stem, stem density; BA, basal area; Ksat, saturated hydraulic conductivity. Litter refers to plant litter ground cover, Bare refers to bare soil ground cover, Rock refers to rock ground cover, Live refers to live-plant ground cover.

(TSS, soil moisture, rock cover, live plant cover, runoff volume, stem density, and basal area [see Table 5]). Runoff to throughfall ratio was not found to be correlated with slope or any of the vegetation measures. Understory live plant cover had the highest correlation with throughfall of all environmental variables. Slope and soil moisture were the next most influential variables in the matrix. Slope was negatively correlated with soil moisture and positively correlated with bare soil cover, stem density, and basal area. Soil moisture was positively correlated with TSS and throughfall and negatively correlated with bare ground cover, basal area, and slope. Variables found to be significantly correlated with TSS (throughfall, soil moisture, and coarse woody debris) were used in a multiple-stepwise regression. The analysis indicated that all three variables were significant predictors of TSS (TSS = –0.63 + 0.1(throughfall) + 1.68(soil moisture) + 10.50(coarse woody debris); $r^2 = 0.58$; $P = .05$).

DISCUSSION

The first objective of this study was to investigate correlations between environmental variables and runoff volume and TSS in runoff. Hypothesis 1 predicted that bare soil cover and soil moisture would have the highest correlations with runoff volume and TSS in runoff. This hypothesis was partially supported by the data for TSS but not for runoff volume. TSS was significantly correlated with throughfall, soil moisture, and coarse woody debris. The correlation with throughfall indicates that more throughfall provides more opportunity for physical detachment of soil particles. Because overland flow is blocked from entering the runoff plots, the process of soil erosion from the plots is most likely caused by detachment of soil particles from the physical impact of falling droplets inside the plots. Because measured rainfall events were typically 1- to 2-day episodes, it is likely that the larger rainfall events involved higher rainfall intensities. In field experiments, Grace (2008) found that total precipitation, average rainfall intensity, and maximum 30-min rainfall intensity were the most influential storm characteristics causing soil erosion. Mean throughfall per site correlated well with total rainfall in the Mānoa watershed. Only one rain event per month was captured in the throughfall gauges in this study, yet monthly rainfall data from the USGS Mānoa Rain Gauge (USGS 211747157485601 711.6, Kānewai Field, Honolulu, O‘ahu, Hawai‘i) exhibited a pattern similar to that of throughfall. December had the highest rainfall and the highest throughfall, followed by March and November. The rainfall data also demonstrated that the typically wetter months of January and February had less rainfall than the typically drier month of October, demonstrating that although seasonal patterns in rainfall do exist, heavy precipitation events are possible at any time during the year.

We hypothesized that bare soil cover would correlate with TSS in runoff but found no significant correlation. This may be in part because the highest levels of bare soil occurred at one of the driest sites that had overall low TSS in runoff (RT). Bare soil across all sites was surprisingly low, considering the high quantity of sediment coming off the plots during rain events. Litter cover levels were generally high, with 100% litter cover recorded in the fenced plot at one site (WR). Thus it was somewhat unexpected that TSS levels in runoff were so high given the fact that litter cover was generally high across all sites and litter protects the soil surface from raindrop impact, slows overland flow, and reduces erosion. Browning (2008) conducted similar research at the same sites used in this study and found that TSS was correlated with bare soil cover. However, this earlier estimation of bare soil cover involved a much coarser visual estimation compared with the method employed here, and Browning's cover estimates were made in the wet season and cover estimates in our study were made in the dry season when litter dynamics are considerably different. In this study, coarse woody debris ground cover showed a strong positive correlation with TSS. It may be that coarse woody debris is positively correlated with the presence of larger tree species (i.e., *E. grandis*) with taller canopies that result in throughfall with greater velocity when striking the soil surface, causing greater soil particle detachment.

We also hypothesized that soil moisture would correlate with TSS in runoff, because soil moisture is an important factor in determining runoff generation (Lau and Mink 2006). TSS in runoff was correlated with soil moisture in this study and in previous research at these study sites (Browning 2008). The lower correlation between TSS and soil moisture compared with TSS and throughfall in this study might also be explained by the fact that soil samples were taken in the first dry period of each month during the activation process. Because rain events are hard to predict, there was often a lag time between site activation and rainfall events of 2–20 days, such that soil moisture during sampled rain events might have differed from that measured during site activation.

Most Hawaiian soils are known to absorb water readily (Lau and Mink 2006), with infiltration rates ranging from 0.043 to 0.51 m hr^{-1} depending on soil type (U.S. Department of Agriculture 1972). Ksat values between 0.0125 and 0.05 m hr^{-1} have been reported from forested sites near Lyon Arboretum (Lau and Mink 2006). Antecedent soil conditions, however, affect infiltration, with an approximately 50% reduction during wet conditions in Hawai'i (Lau and Mink 2006). Data from our study documented large differences in Ksat across sites and even between treatments at the same site. Some of the calculated values from our sites are in the 0.013–0.05 m hr^{-1} range found by Lau and Mink (2006), but many were considerably larger. There could be several reasons why the Ksat data were so variable. First, the instrument used to estimate Ksat is typically used on level ground, and the sites in this study had slopes ranging from 6% to 26%. Second, it was difficult to find a homogenous soil surface without roots, rocks, or other debris, and this heterogeneity can interfere with infiltration measurements.

Though basal area and stem density were not correlated with TSS, they did highlight the high degree of heterogeneity in the Mānoa watershed in terms of physical and biological differences across sites. Seedling and sapling data demonstrated large *Psidium cattleianum* recruitment, which was of particular interest for this study because it is a well-documented food source for feral pigs (Huenneke et al. 1990, Nogueira et al. 2007), and its seeds are readily spread in pig feces (Diong 1982). Pigs also can increase soil nutrient availability (Spear and Chown 2009), which may facilitate nonnative plant establishment at the cost of native Hawaiian plants that are largely adapted to more resource-limited environments (Ostertag et al. 2009). There were twice as many *P. cattleianum* saplings and seedlings in the unfenced versus the fenced plots, indicating that pigs may be promoting further plant invasions. A study in Big Thicket National Preserve, Texas, found that exotic *Sapium sebiferum* (Chinese tallow tree) was twice as abundant in the presence of feral pigs (Siemann et al. 2009).

Runoff volume was significantly correlated with throughfall and rock cover. The positive correlation of throughfall and runoff volume was expected because higher amounts of throughfall in a rain event generally result in higher runoff. Rock cover was positively correlated with runoff volume, likely because higher rock cover represents more impermeable surfaces that result in less infiltration and higher runoff volumes.

Our second study objective involved determining if feral pigs increase both runoff volume and TSS in runoff. Hypothesis 2 predicted that TSS would be higher in unfenced plots than in fenced plots. The results from this study did not support this hypothesis. Differences in runoff volume between treatments were significant, but there was a significant treatment-by-month interaction. At one site (MC) mean runoff volume was significantly lower in the fenced plot as hypothesized. At two other sites (MF and PF) mean runoff volumes were slightly lower in the fenced plots. At three other sites (LY, PP, WR) runoff volumes were similar in fenced and unfenced plots, and at one site (RT) mean runoff volumes were slightly higher in the fenced plot. Many variables are involved in runoff generation, and the Mānoa watershed is characterized by extremely heterogeneous physical and biological properties, which made this objective particularly challenging to address. This research did, however, demonstrate that feral pigs are present and actively disturbing soils in the unfenced plots. Effects of feral pigs on soils (and therefore runoff) may take longer to appear than the 2 years that the runoff plots and pig exclusion fencing have been in place. The effect of feral chickens (*Gallus gallus domesticus*), mongoose (*Herpestes javanicus*), and rats (*Rattus* sp.) was another potentially confounding variable that was not quantified in this study. Unlike pigs, those species can most likely bypass exclusion fencing to access unfenced plots.

Hypothesis 3 predicted that TSS would be higher in the wet season than in the dry season, and the results of this study generally supported this hypothesis. Runoff volumes were significantly higher in many of the wet-season months (November, December, and March) compared with the dry-season months, but that was not the case across all months. For example, the dry-season month of September had higher mean runoff than the wet-season months of February and April. This is most likely a result of differences in the size and timing of individual rain events as well as differences in antecedent conditions before those events.

Runoff volumes were also different across sites, but this was confounded by a significant site-by-month interaction. During many of the smaller dry-season events, at the outset of this study, runoff volume rarely exceeded collection bucket capacity. However, during the larger wet-season events (December and March) runoff regularly exceeded collection bucket capacity. Lau and Mink (2006) demonstrated that runoff initiation times vary considerably in Hawaiian soils, from just a few minutes to >60–80 min, depending on antecedent saturation deficit and soil type. An important finding from this study was the sheer amount of runoff generated from very small plots. Despite the fact that runoff plots drained an area of only 5.04 m^2, the amounts of runoff were extremely large, ranging from 7.5 to >128 liters in December. In a study of agricultural runoff plots in Hawai‘i, Ryder and Fares (2008) found that runoff volume varied from 0.37 to 12.23 liters, though it appears that runoff may have exceeded bucket capacity during the largest event in their study as well. The slopes of the runoff plots ranged from 8% to 12% in the Ryder and Fares (2008) study, which was less than the average slope of 16% in this study.

TSS levels in runoff in this watershed were unexpectedly high for forested ecosystems. The unfenced runoff plot at one site (PP) yielded 7.05 g $liter^{-1}$ during the largest rain event, which was comparable with agricultural studies that measured TSS in runoff from cultivated fields at >10 g $liter^{-1}$ (Borina et al. 2005). Ryder and Fares (2008) found that TSS levels ranged from 41.3 g $liter^{-1}$ in a fallow field plot after an extremely large rain event of 406 mm, to 0.04 g $liter^{-1}$ after a small rain event of 17 mm in a plot in a nearby oat field. De Carlo et al. (2007) found that TSS levels (0.0008–0.019 g $liter^{-1}$) at the outlet of three streams into Kāne‘ohe Bay, O‘ahu, were often several orders of magnitude less than levels found from the runoff plots in our study. A study of TSS in streams and channels in Georgia found that levels ranged from 0.02 to 0.35 g $liter^{-1}$ (Shelby et al. 2005); a study of TSS in streams from 62 catchments in the Midwest (Johnson et al. 1997) ranged from 0.01 to 0.126 g $liter^{-1}$.

CONCLUSIONS

Results from this study demonstrated that runoff and sediment export from the upper forested areas of the Mānoa watershed are highly variable. Runoff volumes and TSS levels in runoff from our small runoff plots can be quite high, especially for the wet-season months, suggesting that this may be the time of year for managers to think about pig control programs to minimize runoff and erosion. In terms of identifying predictor variables for TSS in runoff, soil moisture, throughfall, and coarse woody debris ground cover were all found to be significantly correlated with TSS. Feral pigs have the potential to directly influence two of these three factors (soil moisture and coarse woody debris cover). However, many other factors influence runoff generation, and the heterogeneity of the upper forested areas of this watershed made it difficult to determine what effects, if any, feral pigs had on TSS in runoff over the course of the study. Because the plots had been in place for only 1 yr before the start of this study, it may simply have been too early for differences among plots to be detected. We did document that feral pigs visited three of the four sites where game cameras were deployed, and rooting evidence suggested that at least two more sites were subject to pig disturbance during the course of this study. Several sites with documented pig activity had higher levels of TSS in the unfenced plots, suggesting that pigs may increase TSS levels in runoff. Comparative studies of fencing to exclude feral pigs from forested watersheds are few (Campbell and Long 2009), and the runoff plots established with this study provide an excellent opportunity to study feral pig impacts on water quality, soils, and vegetation. Using paired runoff plots in a forested setting is a novel approach, because we were unable to find any other such studies globally. In addition to elucidating site and seasonal differences in runoff, this study provides important baseline conditions for future research at these sites.

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