Environ Monit Assess (2019) 191: 676
https://doi.org/10.1007/s10661-019-7778-4

Stream flow composition and sediment yield comparison between partially urbanized and undisturbed coastal watersheds—case study: St. John, US Virgin Islands

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Received: 28 September 2018 / Accepted: 27 August 2019 / Published online: 25 October 2019

Abstract In steep dry-tropical islands, rural and urban development can lead to accelerated soil erosion and the delivery of land-based materials into marine ecosystems. The objective of this paper was to compare stream water composition, clay minerology, and sediment yield between a partially urbanized (Coral Bay) and an undisturbed (Lameshur) coastal watersheds in St. John, US Virgin Islands (USVI). The saturation index of streamflow water samples was calculated using "The Geochemist's Workbench" software and most likely precipitated minerals from observed storm events was then compared with X-ray diffraction on soil clay mineralogy. The spatial distribution on both annual mean (2010) erosion rates and storm event–wise (Hurricane Otto) sediment yield among the two study watersheds were modeled using the revised and modified universal soil loss equations (RUSLE; MUSLE), respectively. Cations concentration in stream flow water samples and sediment yield were higher for the partially urbanized (Coral Bay) compared to the undisturbed (Lameshur) watershed. Our findings suggest that rural/urban development may increase stream water cations concentration and inputs of sediment to downstream ecosystems. Future studies evaluating the effect of management practices such as pavement or other stabilization of dirt roads and their impact on stream water quality and quantity and sediment yield are crucial for the proper sediment management in the study watersheds and potentially in other rural-urbanizing tropical watersheds.

Keywords Soil erosion · Sediment yield · Stream flow composition · Clay mineralogy · Rural/urban watersheds

Electronic supplementary material The online version of this article (https://doi.org/10.1007/s10661-019-7778-4) contains supplementary material, which is available to authorized users.

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Introduction

A soil layer covers most of the Earth's land surface. This layer is commonly known as the pedosphere and is the product of a complex interaction between the geosphere, hydrosphere, atmosphere, and biosphere. This interaction provides the conditions in which humankind grows all of its crops (Erns 2000). Soil biota also supply many forest ecosystem services essential to the environment such as primary production through organic matter and nutrient cycling, which determines the chemical and physical composition of the pedosphere, the climate control at global scale through the regulation of C and N fluxes (Brevik et al. 2015). Particularly in dry-tropical watersheds, soil conservation and sediment management activities could help to regulate the temporal delivery and magnitude of sediment yield to coastal waters (Ramos-Scharron and LaFevor 2016).

Soil develops permanently under the impact of fluxes of matter (Chadwick and Chorover 2001) and water thru the system. Among these fluxes, water fluxes are of particular concern as water is the weathering reactive agent as well as the medium for sediment transport in concentrated flows. The land degradation caused by water erosion and human-induced land use change (Hontoria et al. 1999; Garcia-Ruiz 2010; Henry et al. 2013) is a severe threat to the conservation of water and soil around the globe (Trimble and Crosson 2000; Harden 2001; McHugh 2007; Martinez-Mena et al. 2008). Soil erosion and delivery of land-derived sediments into coastal waters are natural processes, but the rates have increased by human activities such as agriculture, deforestation, urbanization (Rawlins et al. 1998), and the building of unpaved roads (Ramos-Scharron 2012; Wemple et al. 2017).

Increased sediment loads into fringing coral reefs have become a menace throughout the insular Caribbean because of the high economic dependency on marine resources and the vulnerability of these ecosystems to terrigenous sediment inputs (Rogers 1990; Gardner et al. 2003). Terrigenous sediments are harmful to coral reefs and can negatively affect coral health, growth, and abundance as well as fecundity and reproduction (Fabricius 2005). Corals in semi-enclosed bays adjacent to steep watersheds such as those in St. John, US Virgin Islands (USVI) may be especially vulnerable to excessive sediment inputs from watershed erosion (Rogers and Teytaud 1988). Particularly on St. John, USVI the watershed erosion in developed watersheds has been linked to an increase in soil erosion (Hubbard et al. 1987; MacDonald et al. 1997; Ramos-Scharron and MacDonald 2007a) and the delivery of land-derived (terrigenous) sediments to coastal bays with fringing coral reefs (Anderson and MacDonald 1998; Gray et al. 2012; Hubbard et al. 1987; Brooks et al. 2007; Ramos-Scharron and MacDonald 2007b).

The aim of this paper is to compare stream water samples composition, sediment geochemistry and erosion rates between partially urbanized and undisturbed watersheds in St. John, *USVI* testing the hypothesis that the delivery of land-based materials to coastal waters is accelerated by the level of rural/urban development among the study watersheds. Results of this investigation indicate that rural/urban development may increase the delivery of dissolved and suspended land-based materials into downstream ecosystems, especially fringing coral reefs.

Material and methods

Study area

St. John, with a total extension of 50 km^2, is an ideal location to test the impact of rural-urban development on the delivery of land-based materials to coral reefs environments. In this study location, there is a clear delineation between partially urbanized and undeveloped watersheds due to the Virgin Islands National Park (Gray et al. 2008, 2012). St. John is located ~ 80 km East of Puerto Rico, and it is the smallest of the three major USVIs (Fig. 1). The area is topographically complex, with steep slopes, as more than 80% of the island has slopes greater than 30% (Anderson and MacDonald 1998), encouraging the vulnerability to soil erosion. Vegetation is dominated by dry evergreen forests, shrubland, and moist tropical forests (Woodbury and Weaver 1987). The climate is subtropical dry with mean annual precipitation usually exceeding 100 cm year^{-1}. Coral Bay (CB), and Great and Little Lameshur Bay (LB) are comprised predominantly of water island formation which is a phenocryst-poor keratophyre, a silicic albite-rich extrusive with varying percentages of quartz and plagioclase (Rankin 2002). CB is a 13.3 km^2 bay with mangroves, sea-grass beds, and fringing coral reefs. The watersheds that drain into CB bay have steep slopes (averaging 18% with large areas over 35%), high erodible soils, with high runoff volumes associated with average rain events (Ramos-Scharron and LaFevor 2016). Rural and urban development in the hillslopes and the construction of many steep dirt roads increased significantly runoff generation (Ramos-Scharron and MacDonald 2007a) and sediment loads to coastal waters (Anderson and MacDonald 1998; MacDonald et al. 1997; Brooks et al. 2007; Ramos-Scharron and MacDonald 2007b). Previous studies have also demonstrated significantly higher rates of terrigenous sediment accumulation below the developed area (CB) compared to the undeveloped area (LB) (Gray et al. 2008). The highest rates of terrigenous sediment accumulation since 2007 occurred following Hurricane Otto in October 2010 (Gray et al. 2012).

Field collection

Soil samples

Soil samples were collected manually at specific sites in the upper watersheds (Fig. 1). These samples were subjected to laboratory analysis including textural analysis, clay minerology, and scanning electronic microscope (SEM) imagery at the CICESE-UNAM laboratories in Ensenada, Baja California, Mexico.

Stream water samples

A total of 33 stream water samples were collected manually from the stream flow at specific sites using 125 ml Nalgen bottles during periods of ephemeral runoff following storms. This sampling strategy was followed in both study areas, and dissolved composition was compared between three different storm events in 2010: October 5–8 (Hurricane Otto), July 20, and September 19. Figure 2 shows the 2010 precipitation data with emphasis on Hurricane Otto (start) when 288 mm precipitated in only 3 days, representing 30% of the annual mean rainfall.

Sediment accumulation in traps

Marine sediment traps were installed at the receiving bays (watershed outlets) and were used to record sediment accumulation rates of terrigenous materials over 26-day periods. See Gray et al. (2012) for a full description of methods. The sediment trap data were compared to the sediment yield modeled for Hurricane Otto.

Laboratory analysis

Soil samples

Organic matter content in the soil samples was determined by loss on ignition (LOI), following standard procedures. A laser particle sorter (Beckman-Coulter LS200) was used to determine grain size distribution for the fractions between 0.4 μm and 2 mm. Mineralogical identification of 5 sediment samples (Fig. 1) was determined using X-ray diffraction (XRD) techniques, and scanning electron microscope (SEM) imagery were obtained at the CICESE-UNAM laboratories in Ensenada, Mexico. XRD analyses were performed using a PANanalytical X'Pert PRO automated powder diffractometer with Cu Kα radiation. This method involved the analysis of three different preparations of each sample that had been previously subjected to physical and chemical treatments: air-dried, heat, and glycol drying as described by Moore and Reynolds (1996). Also, 20 soil samples, 10 samples from LB, and 10 samples from CB watersheds were analyzed by SEM equipped with a back-scattered electron detector, a secondary electron detector, and a Princeton Gamma-Tech (PGT) X-ray energy dispersive system (EDS).

Stream water samples

Major element chemistry of the stream water samples was determined by inductively coupled plasma-atomic emission spectrometry (Varian Liberty 100 ICP-OES). The major cations such as sodium (Na), calcium (Ca), silica (Si), magnesium (Mg), and potassium (K) were analyzed using an induced coupled plasma (ICP) equipment to quantify the streamflow composition and evaluate the bulk dissolved load among the two study watersheds. Additionally, an ion chromatograph (Dionex 2000) equipped with a separation column for anions was used to measure the chloride content, as well as the other major anions, to apply the sea salt correction input by aerosols as described by Holland (1978).

The saturation index (SI) is a method that indicates whether natural waters will tend to dissolve or precipitate a particular mineral based on its chemical composition. The SI value is negative for minerals that had dissolved, positive when it precipitated, and zero when the water and minerals are at chemical equilibrium. Here, this technique will be used to identify certain elements that can be used as tracers of urbanization impacts on streamflow composition. The SI was calculated by comparing the chemical activity of the dissolved ions of the mineral (Ion Activity Product, IAP) with their solubility product (Ksp) as described by Bethke (1996). In equation form,

$$SI = \log\left(IAP/K_{sp}\right) \quad (1)$$

"The Geochemist's Workbench" (GWB) software was used to calculate the saturation index of the stream water samples. The results of this analysis will be compared with the results of the X-ray diffraction analysis on the soil samples, to validate the most likely minerals being precipitated as clay minerals and the minerals

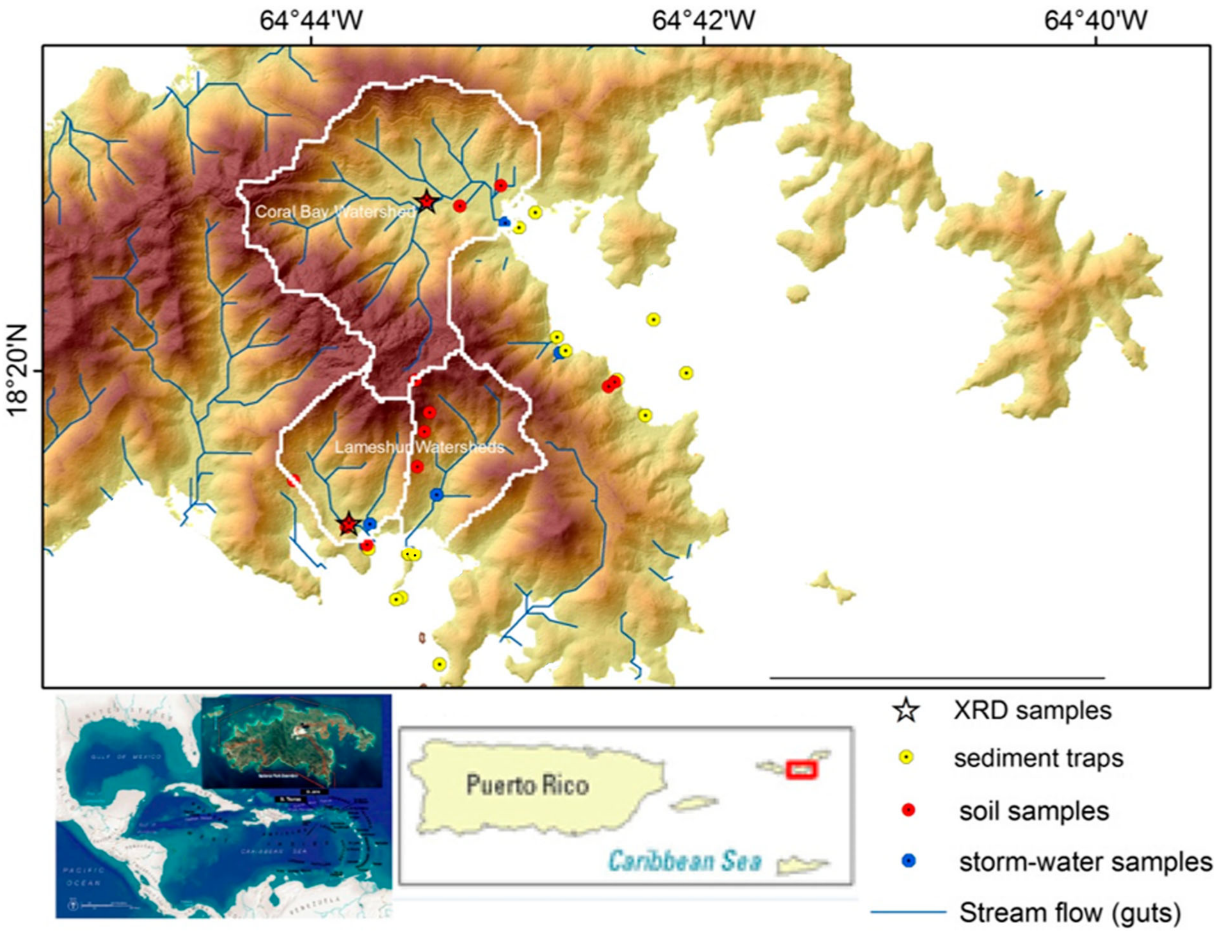

Fig. 1 Geographic location of the study watersheds, soil, and stream water samples

most likely to be dissolved will be compared with the bedrock mineralogy, with the aim of assessing the spatial variability of stream flow composition and clay mineralogy among the two study watersheds.

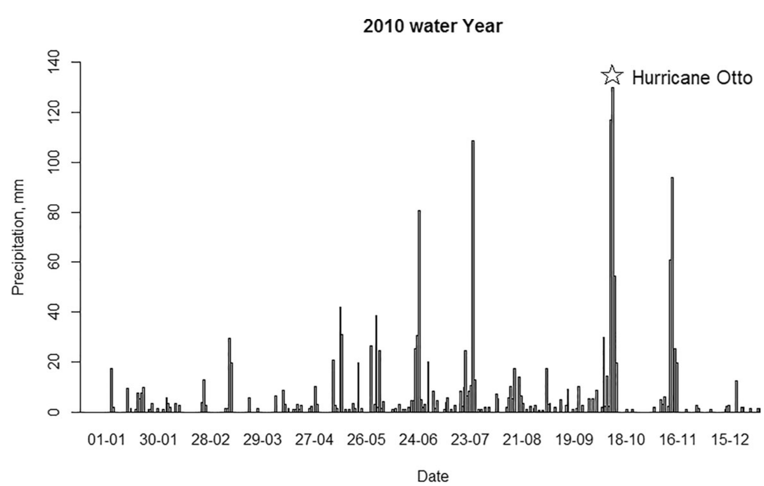

Fig. 2 Daily rainfall time series recorded at St. John, US Virgin Islands from January 1, to December 31, 2010

Soil erosion modeling

Mean annual soil erosion rates, from sheet and rill erosion, were modeled for the two study watersheds using the revised universal soil loss equation (RUSLE, Renard et al. 1996), a revision of the USLE (Wischmeier and Smith 1978). RUSLE considers the erosive energy of precipitation and runoff, on certain soil erodibility, local topography, and land cover conditions to estimate long-term soil erosion rates. RUSLE is a modeling tool developed for soil conservationists (Wischmeier and Smith 1978) and widely applied in conservation management activities (Rennard and Freimund 1994). It is relatively easy to perform in many environmental settings, including tropical watersheds (Millward and Mersey 1999; Chen et al. 2011), and due to the available data in St. John, USVI encouraged us to use it for the soil erosion assessment proposed in this analysis.

The RUSLE equation can be expressed as follows:

$$A_{RUSLE} = R \times K \times (LS) \times C \times P \quad (2)$$

where

A = computed annual soil loss,
R = rainfall-runoff erosivity factor,
K = a soil erodibility factor,
LS = a topographic factor combining slope length, *L*,
and land surface slope angle, *S*,
C = land cover and management, and
P = erosion-control practices

A_{RUSLE} in megagrams per square kilometer is the modeled soil erosion rate. The units for *A* are determined by the units for *R* and *K*, respectively. The remaining terms in Eq. 2 are dimensionless ratios to scale soil erosion estimates to experimental conditions (Wischmeier and Smith 1978; Renard et al. 1996; Smith et al. 2007).

The rainfall erosivity factor *R* (megajoule-millimeters per square kilometer per hour per year) was estimated following the method described in Renard et al. (1996). The soil erodibility factor *K* (megagram-hours per megajoule per millimeter) was taken from National Center for Environmental Information (2011). *L* and *S* are topographic factors, which are commonly combined into the LS factor. These factors are used to scale the length of upstream flow accumulation (22.13 m) and steepness (9%; 5.18°) at any location of the study watershed to the dimensions of the modeled experimental plot described in Renard et al. (1996) and modified by Kinnell (2005) who developed a formula for the GIS environment to extract the RUSLE-LS factor from a digital elevation model (DEM) as follow:

$$LS = (Flow\ accumulation \times cell\ size/22.13)^{0.4} \times (\sin[slope]/0.0896)^{1.3} \quad (3)$$

where "cell size" is the pixel size (in m) for a DEM and "flow accumulation" is the accumulated flow to each downslope cell. The topographic features used to calculate the compound RUSLE-LS factor were produced using elevation data at a 5 m horizontal resolution from a LIDAR-derived DEM.

The cover factor *C* values were taken from McCreery (2007), who calculated a composite *C* factor for each watershed based on land cover and road density using field observations and aerial photography. A total area of 23,000 m^2 and 5200m^2 of dirt roads were considered for CB and LB, respectively. We assigned a value of 1 for unpaved road and deforested areas, and zero for the impervious land cover fraction represented by paved roads. The last term in Eq. 2 which is *P* is an estimate of the efficiency of any field management employed to reduce soil erosion. A constant value of *P* = 1 was applied over the whole study area because no sediment management practices were reported during the simulation period.

MUSLE

There are no perennial streams in St. John, USVI and runoff associated with storms is generally short-lived (days) and discharges from ephemeral drainages called "ghuts." Due to the ephemeral nature of erosion events and runoff, averaged models like RUSLE which estimate long-term rates may not be adequate to address the event-wise soil erosion rates. Therefore, Williams and Berndt (1972) developed a sediment delivery equation that predicts soil erosion at watershed scale on an individual storm event basis. The modified USLE (MUSLE) uses most of the RUSLE parameters (Renard et al. 1996), but it replaces the average rainfall intensity parameter with a value for peak discharge and the runoff depth created by each storm event. The sediment yield to the receiving bays during Hurricane Otto was modeled using the MUSLE model on the two-study watershed. This equation can be expressed as follows:

$$A_{MUSLE} = 11.8 \times \left(Q \times q_p\right)^{0.56} \times K \times (LS) \times C \times P \quad (4)$$

where

A computed storm-based soil loss (mg km^{-2})
Q runoff depth (m^3)
q_p peak discharge (m^3 sec^{-1})

The curve number (SCS CN) method as described in *USDA TR55* (US Department of Agriculture 1986) was used to determine runoff depth "Q" and peak discharge (q_p). The peak discharge is the maximum runoff flow rate of the storm and was calculated following the method described by Snider (1972). The SCS curve number (CN) is one of the most important parameters for the peak discharge and total runoff calculations within the MUSLE model (Gudino-Elizondo et al. 2018a), and determine the amount of actual rainfall precipitation contributing to overland flow. SCS CNs values were assigned according to the land use class dominant in each watershed based on literature values (Dunne and Leopold 1987).

The ArcMap (Version 10.2; Environmental Systems Research Institute, Redlands, California, USA) software was used to map spatial variations in soil erosion and sediment yield rates from RUSLE-MUSLE calculation, respectively, by multiplying the factors of the models such as rainfall erosivity (R), soil erodibility (K), topography (LS) and vegetation cover (C), using different grid surfaces created with the spatial analyst tools within the ArcMap software at a 5 m horizontal resolution.

Marine sedimentation

Marine sediment accumulation (terrigenous) rates were obtained from submarine sediment traps deployed at the ephemeral stream outlets within the receiving bays at LB and CB (Fig. 1), and were used to compare to the modeled sediment yield during the Hurricane Otto. Details regarding the sediment traps and sedimentation rates calculations are described in Gray et al. (2012) and Kolupski (2011).

Results

Hydrochemistry

A summary of the stream water chemistry is presented in supplementary material 1 (S1). The concentrations of *Ca*, *Mg*, *Na*, and *K* are mainly controlled by a combination of bedrock geochemistry, land cover, and topography. The *Na* cation showed the highest concentrations of the entire dataset. In general, samples from CB (partially urbanized watershed) contained higher concentrations of all the analyzed elements than the samples collected in LB (undeveloped watershed).

Statistical analysis

The concentration values (S1) were statistically analyzed (t test) to determine spatial differences at 5% significance level ($p < 0.05$) between disturbed and undisturbed coastal watersheds as well as temporal differences between storms. The normally distribution of the data was assumed based on the Shapiro-Wilk test. Results of the t test analysis are shown in Tables 1 and 2; asterisks in bold numbers indicate significant differences.

Statistical differences were found in *Ca*, *Mg*, and *K* concentrations, while *Na* and *Si* did not differ significantly in the temporal comparison (July 20 and October 6–7 2010) of the stream water chemistry of samples collected in the LB watershed. By contrast, no significant differences were registered in the concentrations of all elements analyzed in the stream water samples collected at CB between storms (September and October of 2010). Table 1 shows the results of the temporal comparison at each watershed.

Statistical differences in *Ca* concentration were found in the spatial comparison between stream water samples collected in the two study watersheds during Hurricane Otto (October 5th–8th of 2010). Stream water samples from CB contained a higher concentration of

Table 1 t test results (p values) for Lameshur and Coral Bay

t test	Lameshur	Coral Bay
Ca	0.001*	0.097
Mg	0.001*	0.113
Na	0.113	0.141
K	0.038*	0.169
Si	0.185	0.201

Values with asterisk indicate significant differences

Ca, while all other element concentrations did not differ significantly. Table 2 shows the results of the spatial comparison of the two study locations.

Saturation index

The saturation index of the stream water composition was calculated using "The Geochemist's Workbench" software. Minerals with the highest probability of being dissolved by rock-water interactions on the bedrock were selected, and minerals with the highest potential of being precipitated as clay minerals in the soils. A higher *SI* value was found for illite in samples from CB, indicating that illite is more likely to precipitate as a clay mineral in CB than in the LB watershed. In supplementary material 2 (S2), Table S2-I shows the computed *SI* for the LB watershed, and Table S2-II shows the computed *SI* for the CB watershed.

X-ray diffraction analysis

Coral Bay

XRD results of the air-dried samples indicated that illite (I) is the most abundant clay mineral in the CB soil samples, followed by a peak in the plot representing the combination of chlorite (Cl) and kaolinite (K) concentrations (Fig. 3a).The smectite (Sm) peak appeared as a minor proportion. The heat treatment of the sample (Fig. 3b) preserved the highest concentration of *I*, making it the most abundant clay mineral component in the soil samples. Also, *Cl* and *K* peaks overlapping in the air-dried diffract-gram were solved with this physical treatment (heat); *K* collapsed and the remaining peak only represents *Cl*. Thus, the relative concentration pattern of clay minerals in CB was the following:

Table 2 *t* test results (*p* values) for the spatial comparison of Hurricane Otto

Independent samples	
t test	*p* values
Ca	0.030*
Mg	0.214
Na	0.366
K	0.062
Si	0.374

Value with asterisk indicate significant differences

I > Cl > K > Sm

Lameshur

The air-dried sample of LB indicated that *Cl* was the most abundant clay mineral, followed by a peak in the plot representing the combination of *Cl* and *K* concentrations (Fig. 4a). The *Sm* peak appeared in the diffractgram in minor proportion. The heated sample (Fig. 4b) kept the highest concentration of *Cl*, representing the largest component of clay minerals after this physical treatment. Therefore, the relative concentrations pattern of clay minerals in LB was the following:

Cl > K > I > Sm

A higher relative concentration of illite in CB samples, supported by higher SI values of this mineral on the stream water samples, may reflect the influence of rural urbanization (especially dirt roads density) in the dissolution of land-based materials by overland flow among the two study watersheds.

Scanning electronic microscope

A total of 20 soil samples were analyzed by scanning electron microscope (SEM). In general, a distinct spatial variation is apparent, regarding the presence of more rock fragments in the developed watershed (CB) and more clay cohesion in the reference watershed (LB), respectively. SEM analysis also suggests greater physical weathering in CB compared with the LB watershed samples, which is consistent with the previous comparison on the stream flow and sediment geochemistry analyses. Figure 5 shows *SEM* images representatives for the two study watersheds.

Soil erosion and sediment yield modeling

The *RUSLE-MUSLE* calculation was performed multiplying the factors of the model, such as rainfall erosivity (*R*), soil erodibility (*K*), topography (*LS*), and vegetation cover (*C*), using different grid surfaces created in ArcMap spatial analyst. The output data were mapped to visualize the spatial distribution of the computed erosion rates at the watershed scale in the two study locations (Figs. 6 and 7).

The rainfall erosivity factor (R_{RUSLE}) was calculated using annual mean precipitation of 1150 mm (70 mm of

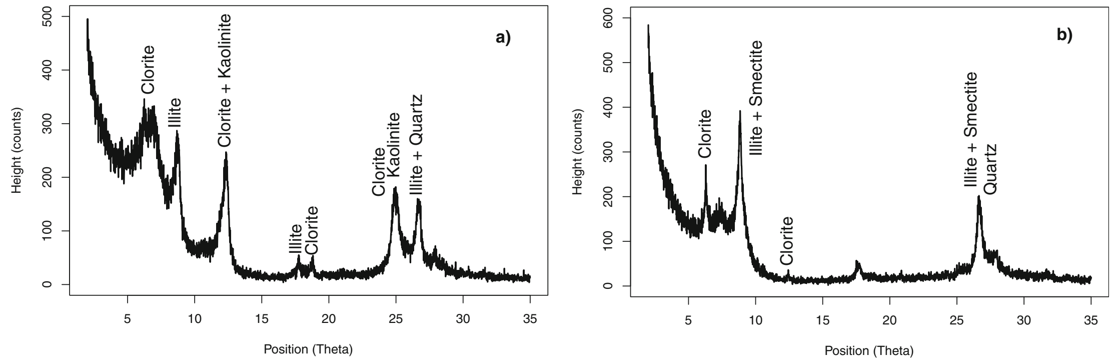

Fig. 3 **a** Air-dried and **b** heated diffractograms of the sample collected in Coral Bay watershed

standard deviation) estimated from three automatic stations in St. John, *USVI*. The R_{RUSLE} factor calculation results in 270 MJ mm $km^{-2}h^{-1}year^{-1}$ for the whole island, and the *K* factor (National Center for Environmental Information 2011) interval was 0–0.041 Mg h MJ^{-1} mm^{-1}. The storm event–wise erosivity factor (R_{MUSLE}) values resulted in 42.3 and 46.7 km^2 mm min^{-1} for LB and CB watersheds, respectively. These values form the basic input for *RUSLE-MUSLE* models to calculate the annual (2010) and the storm event–based (Hurricane Otto) soil loss.

The computed erosion rates were higher for the developed (CB) compared with the undeveloped (LB) watersheds with average values of 12 Mg km^{-2} $year^{-1}$ for CB and 3.15 Mg km^{-2} $year^{-1}$ for LB, by assuming an average density of 1500 kg/m^3. The average sediment yield modeled for Hurricane Otto was 8.4/2.1 Mg km^{-2} at LB and CB, respectively. Figures 6 and 7 show the spatial distribution of modeled erosion rates for the two study watersheds.

Soil erosion is strongly related to topographic slope, land use, and dirt roads density. Significant differences were observed between modeled soil erosion rates from CB (partially urbanized) compared with LB (undisturbed) watersheds. CB watershed erodes, on average, about 4 times more soil than LB per unit area per year. The *C* factor assigned to each study watershed also has a potential effect on the modeled soil erosion rates.

Spatial variations of sediment yield and coastal sedimentation for Hurricane Otto

We compared the modeled sediment yield from the MUSLE model with the observed sedimentation rates at the watershed outlets during the Hurricane Otto. The watershed sediment yield was generally lower compared to the sedimentation rates at the receiving bays. Sedimentation rates from October 5th to 8th (normalized over the 26 days of the sample period from Gray et al. 2012) were considered to estimate an approximate value of coastal sedimentation during the Hurricane Otto. Table 3 shows the sediment yield and coastal sedimentation for the two study watersheds.

Marine sedimentation was also higher in CB. The total drainage area (significant higher in CB) and the presence of coastal ponds (in LB watershed) has a

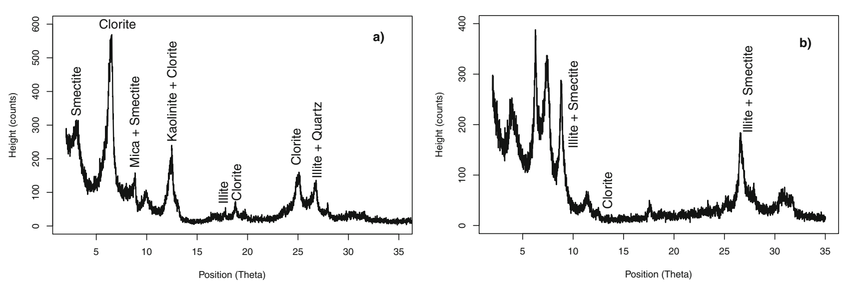

Fig. 4 **a** Air-dried and **b** heated diffractograms of the sample collected in Lameshur watershed

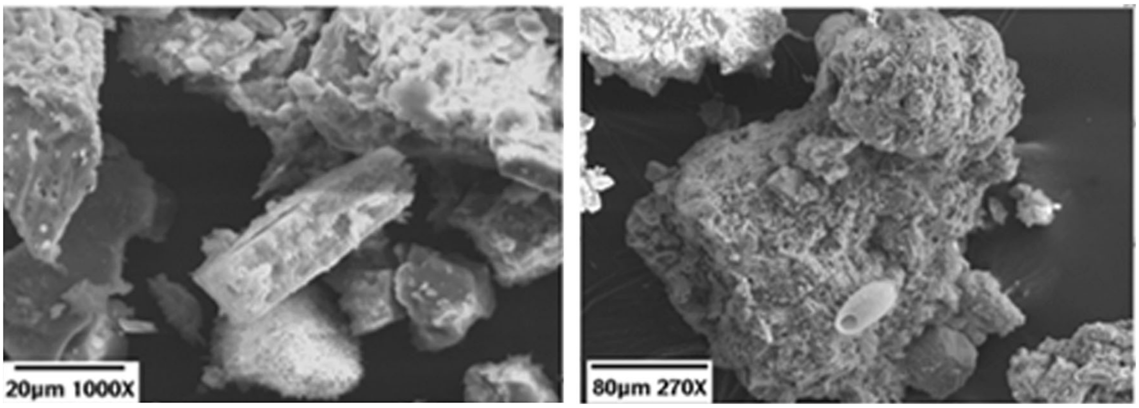

Fig. 5 SEM images of soil samples from **a** Coral Bay (× 1000) and **b** Lameshur (× 270) watersheds

potential effect on the sedimentation rates among the two study areas. Sedimentation rates are consistent with other studies (Rogers 1990; Kolupski 2011; Gray et al. 2012) reporting relatively similar values.

Discussion

Soil erosion rates (Ramos-Scharron and MacDonald 2007a, b) and the corresponding sediment loads (Gray

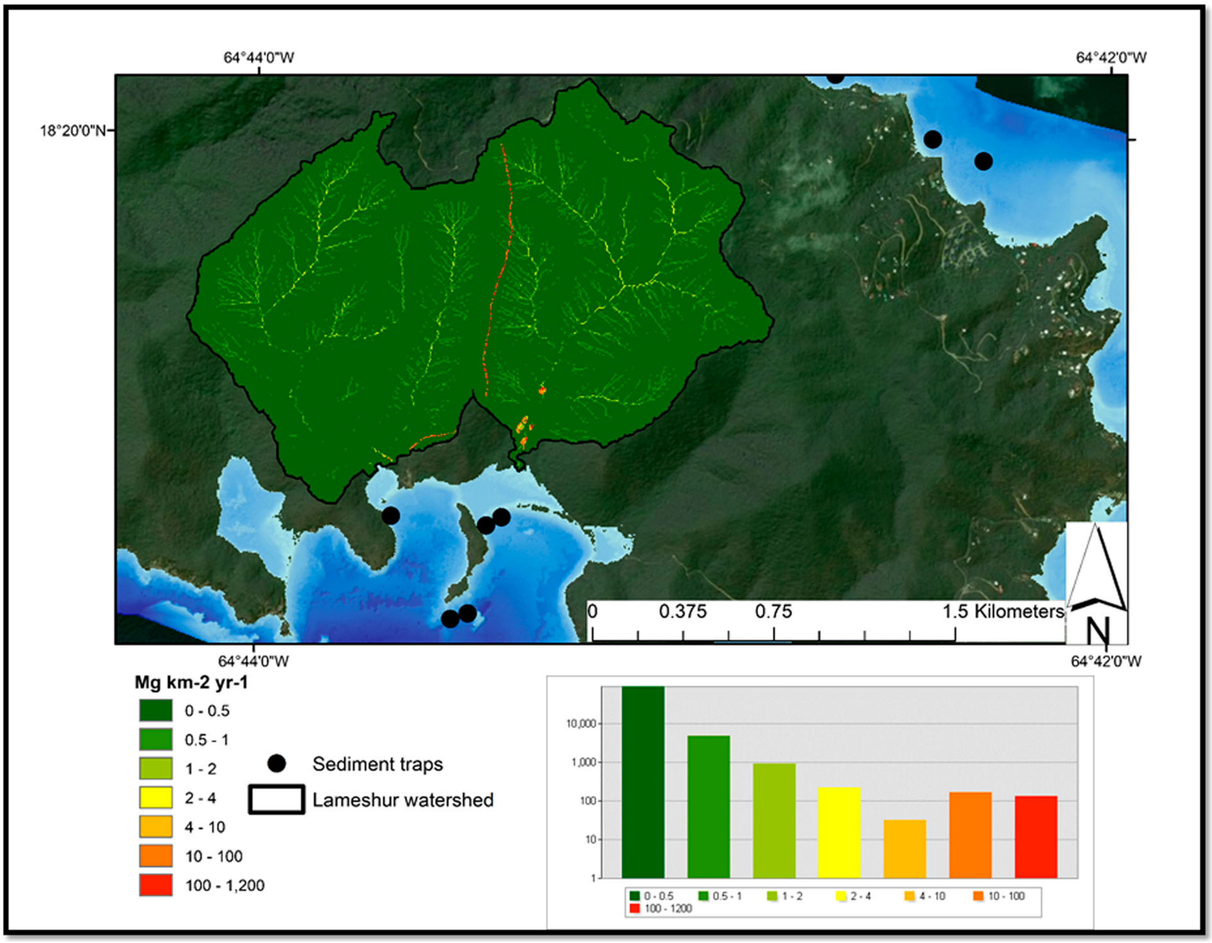

Fig. 6 Modeled mean annual (Mg km^{-2} year^{-1}) erosion rates for LB. Inset shows the frequency distribution of erosion values per pixel at the watershed scale

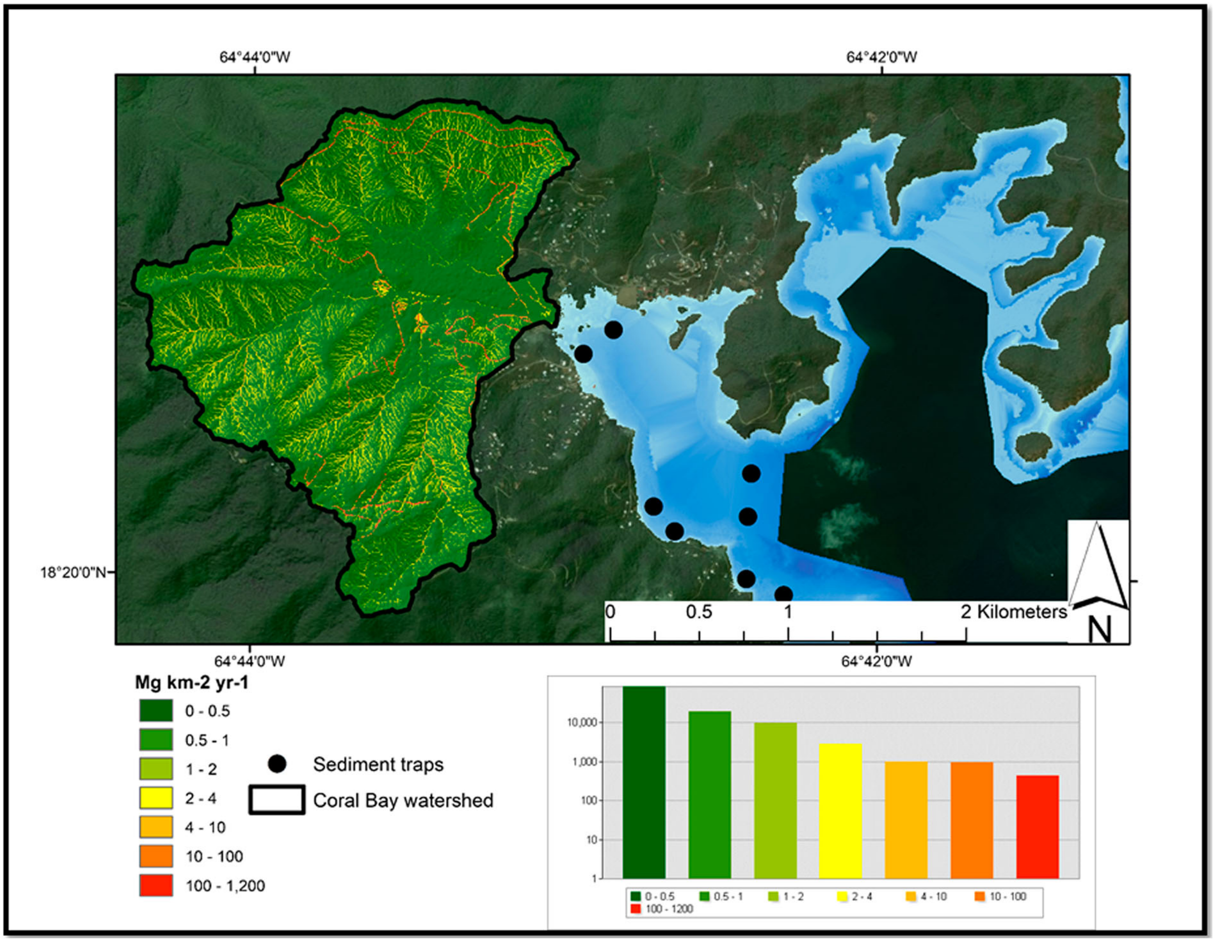

Fig. 7 Modeled mean annual (Mg km^{-2} year^{-1}) erosion rates for CB watersheds. Inset shows the frequency distribution of erosion values per pixel at the watershed scale

et al. 2012) into the bays adjacent to partially urbanized watersheds have been described as higher than in undeveloped (references) areas. However, few studies analyze the influence of rural urbanization and the corresponding impacts on the stream water composition and clay mineralogy. The contribution of the present research is the combined approach to address chemical and physical erosion at the watershed scale in St. John, *USVI* to evaluate the effect of rural/urban development for the Caribbean islands, and potentially for other urbanizing tropical watersheds.

The modeled soil erosion rates estimated in this study showed that are significantly different between the two studied locations: specifically, the CB (partially urbanized) watershed eroded more soil per unit area than the LB (undeveloped) watershed. Figures 6 and 7 show the spatial variation of soil erosion rates among the study watersheds. The insets on these figures highlight the contribution of dirt roads networks to enhance soil erosion as hotspots of sediment production, and potentially increase the delivery of land-based material to downstream ecosystems as reported previously in the literature (Anderson and MacDonald 1998; Ramos-Scharron and MacDonald 2007a, b; Gudino-Elizondo et al. 2018b; Wemple et al. 2017; Oliver et al. 2018; Pait et al. 2018; Gudino-Elizondo et al. 2019). Ramos-

Table 3 Watershed sediment yield and coastal sedimentation rates during Hurricane Otto

	Sediment yield (Mg)	Sedimentation (Mg)
Coral Bay	8.4	158
Lameshur	2.1	15

Scharron and Lafevor (2016) reported that precipitation excess on unpaved roads in St. John, USVI contributes for 25–62% of total watershed discharge, and potentially increase sediment production on unpaved roads, though the unpaved road network represents only 1% of the total watershed area. Our results show that dirt roads can increase soil erosion rates by two or more orders of magnitude, which are consistent with values previously reported by Ramos-Scharron and MacDonald (2007a) and Ramos-Scharron (2018). The methodology used in this paper to estimate the relative contribution of unpaved roads on sediment generation using RUSLE-MUSLE can be easily applied in other urbanizing watersheds, especially in remote areas with limited data.

The high resolution elevation map derived from *LIDAR* data, and the spatially distributed *K* factor values used in this analysis provide more accurate *RUSLE-MUSLE* estimations than earlier soil erosion modeling studies in St. John, *USVI*, such as those by Radke (1997) and McCreery (2007). The complex topography of St. John, *USVI* indicates that the *L* and *S USLE* factors play a critical role in the soil erosion modeling. Also, the *K* factor is extremely important because it represents the susceptibility of soils to erosion, based on its physical soil properties. The *K* factor value is often estimated with significant uncertainty using an average value per watershed. However, the St. John Erosion Model (STJ-EROS) developed by Ramos-Scharron and MacDonald (2007b) is, to the authors knowledge, the most robust sediment yield model in the study area. Future research linking the spatial variability of the dissolved load concentrations in stream water with a most sophisticated soil erosion modeling would help to further assess impacts on urbanization to the delivery of dissolved and suspended loads between the two study watersheds.

On the other hand, significant differences in stream water chemistry were found between the two study areas, reflecting the currently prevailing natural (reference) and anthropogenic conditions of each watershed. *Ca* cation concentration was statistically higher in the *CB* during Hurricane Otto, and it was mainly related to overland runoff from rural urbanized areas where construction materials, such as concrete, is the most likely source of dissolved *Ca* in stream water. In addition to *Ca*, variations in concentration of other elements such as *Mg*, *Na*, and *K* can be explained by the fact that these elements are more active in water-rock interactions than Si, which is less likely to undergo chemical weathering. *Ca* is not presented in the geochemistry of the bedrock as described by Rankin (2002), suggesting that this element, in particular, can be used as a tracer to evaluate impacts of urbanization on the dissolved load in the stream water samples. Non-significant differences in the spatial distribution for the remaining major elements concentration can be explained by the relatively homogeneity in bedrock minerology among the study watersheds.

Temporal differences in *Ca*, *Mg*, and *K* were found in stream water chemistry of samples collected from the LB watershed, while *Na* and *Si* were not significantly different between the two analyzed storm events in the reference (undeveloped) watershed. *Ca* and *K* concentrations were higher during the storm that occurred on September 19, while *Mg* was higher during Hurricane Otto. This difference can be attributed to the amount of rainfall during each storm (180 and 300 mm respectively) and to the diluting effects on the concentration of these elements within the watershed. Our results suggest that higher precipitation could produce greater dilution, reflecting on the concentration of dissolved elements in water runoff of each storm event. Nevertheless, no temporal differences were found in the dissolved composition of *CB*.

Na was found in stream water with the highest concentration of the entire dataset. The high content of *Na* can be related to the geochemistry of the bedrock, which is mainly comprised of keratophyre rocks (Rankin 2002). The *SI* results shows that the most likely mineral to be in dissolution in the stream water samples is calcite, which is an alteration product of feldspars which is present in the geochemistry of the bedrock in both study watersheds. By other hand, the most likely mineral to precipitate is illite, which is consistent with the results of the XRD diffraction analysis of the soil samples. Vegetation removal due to rural urbanization in tropical watersheds, especially the construction of unpaved roads, enhances mineral dissolution from the bare-soil by overland flow that can eventually form secondary minerals in low temperature environments as described by Liu et al. 2015. Also cut-slopes due to urban infrastructure can increase the time of residence for water-rock interaction and potentially increase the dissolved load concentration to downstream ecosystems.

X-ray diffraction analysis showed a greater relative concentration of illite in the samples from CB. Illite is a chemical alteration product of feldspars in hydric and hydrothermal environments. X-ray diffraction results may suggest that high concentration of illite can be attributed to human influences in the partially urbanized watershed, supported by the saturation index results showing that illite is the mineral more likely to be precipitated based on the chemical composition of the stream water samples. Although, it is important to note that some flow paths coming from the upper CB watershed may contain sediments from hydrothermally altered rocks (locality known as “Bordeaux Mountain”). So, the spatial variation of illite concentration could also be related to natural water-rock interactions in the CB watershed, which is consistent with the geochemistry the bedrock described by Rankin (2002). A more extensive sample collection would be helpful to better address human vs natural influences on stream flow and soil clay minerology.

Furthermore, the scanning electron microscopy (SEM) imagery showed, in general, more fragments of rocks in soil samples collected in *CB* compared with soil samples collected in LB watershed, which suggest that rural urbanization in *CB* may have important effects on physical erosion due to deforestation, enhancing the vulnerability to physical erosion as shown in Figure 5a and b. The textural analysis in soil samples reported by Gudiño Elizondo (2012) showed that approximately 80% of the samples are silt-sized particles, which are highly susceptible to soil erosion (Hjulstrom 1935). The energy required to suspend and transport sediment is greater for clays and sands compared with silts because larger grain sizes are more difficult to move. Although clay is a fraction finer than silt, clay particles have a cohesive force that binds them together to create a larger grain size, and that requires more energy for transport (Hjulstrom 1935) compared to actual coarser grains. Organic matter content ranged from low values of 3–6% and could be due to the complex topography that does not allow much lateral accumulation of organic matter in the two watersheds

The modeled sediment yield was compared with observed sedimentation rates at the watershed outlet during the Hurricane Otto. On average, 158 metric tons of terrigenous sediment were deposited at CB watershed outlet, and only 8.4 metric tons of soil loss were modeled from the watershed. In LB bay, a mean sedimentation of 15 metric tons was observed in the receiving bay based on sediment trap data, and only 2.1 metric tons of soil loss were modeled. The overall balance between watershed sediment yield and sedimentation rates at the bays resulted in higher observed sedimentation at the receiving bays than the modeled watershed sediment yield in both study locations (CB/developed and LB/undeveloped watersheds). The modeled sediment yield for the two study watersheds suggested that most of the eroded sediments from the upper watersheds, mainly generated from dirt roads, are being deposited preferentially within the catchment area and/or, in the case of LB, at the coastal sediment ponds rather than at the receiving bays. Is important to note that the sedimentation rates at the bays are influenced by many factors including advection of sediment and resuspension as described in Gray et al. 2012. Nevertheless, the modeled sediment yield could be improved using runoff measurements to better calibrate the MUSLE model, or even better, can be improved using more sophisticated modeling tools such as the STJ-EROS model (Ramos-Scharron, 2007b). Results from this investigation indicate that rural urbanization may increase the dissolved and suspended loads from watershed hydrology causing a potentially negative effect on sensitive downstream ecosystems, especially fringing coral reefs.

Conclusions

Our findings suggested that rural urbanization has important effects on stream flow composition and sediment yield in dry-tropical coastal watersheds in St. John, US Virgin Islands. This impact was mostly attributable to the loss of the native vegetation. Furthermore, the construction of dirt roads and the dissolution of urban material may also impact dissolved and suspended load rates at the watershed scale. Significant differences in soil erosion rates and sediment yield were observed between the two studied watersheds; these erosion rates were also reflected in the spatial variation of the dissolved composition of the stream water samples. Erosion rates and sediment yield (annual and specific for hurricane Otto) and dissolved composition of stream water samples were higher in CB (developed watershed). The methodology described in this paper can be used in other watersheds to evaluate anthropogenic impacts on stream water composition and sediment yield. Based on this analysis, the human influence can be related to deforestation activities and the dissolutions of urban-derived materials such as industrial paint, asphalts, among others construction materials. Future studies evaluating the effect of management practices such pavement or other stabilization of dirt roads and their impact on runoff quantity and quality, soil erosion, and sediment yield are crucial for proper sediment management in the study watersheds and potentially in other rural/urban tropical watersheds.

Acknowledgments Special thanks to Stephen V. Smith, Alejandro Hinojosa Corona, María Elena Solana Arellano, María Isabel Pérez Montfort, Matthew Brand, Jochen Schubert, Brett Sanders, Robert Harrington, Whitney Sears, Esther Araiza, Mario Vega, Eloisa Aparício, Ivonne Pedrin, Belinda Sandoval, and the Virgin Island Environmental Resource Station (VIERS).

Funding information This study was funded by Consejo Nacional de Ciencia y Tecnología (CONACyT), and the National Oceanic and Atmospheric Administration Ecological Effects of Sea Level Rise Program (award NA16NOS4780206). Field and some lab work were supported by funding from the American Recovery and Reinvestment Act (ARRA) and the National Oceanic and Atmospheric Administration Coral Reef Conservation Program to Dr. Sarah C. Gray from University of San Diego.

References

Anderson, D. M., & MacDonald, L. H. (1998). Modeling road surface sediment production using a vector geographic information system. *Earth Surface Processes and Landforms, 23*, 95–107. https://doi.org/10.1002/(SICI)1096-9837(199802)23:2<95::AID-ESP849>3.0.CO;2-1.

Bethke, C. M. (1996). *Geochemical Reaction Modeling*. USA: Oxford university press.

Brevik, E. C., Cerdá, A., Mataix-Solera, J., Pereg, L., Quinton, J. N., Six, J., & Van Oost, K. (2015). The interdisciplinary nature of SOIL. *SOIL, 1*, 117–129. https://doi.org/10.5194/soil-1-117-2015.

Brooks, G. R., Devine, B., Larson, R. A., & Rood, B. P. (2007). Sedimentary development of Coral Bay, St. John, USVI: a shift from natural to anthropogenic influences. *Caribbean Journal of Science, 43*, 226–243. https://doi.org/10.18475/cjos.v43i2.a8.

Chadwick, O. A., & Chorover, J. (2001). The chemistry of pedogenic tresholds. *Geoderma, 100*, 321–353. https://doi.org/10.1016/S0016-7061(01)00027-1.

Chen, T., Niu, R., Wang, Y., Li, P., Zhang, L., & Du, B. (2011). Assessment of spatial distribution of soil loss over the upper basin of Miyun reservoir in China based on RS and GIS techniques. *Environmental Monitoring and Assessment, 179*, 605–617. https://doi.org/10.1007/s10661-010-1766-z.

Dunne, T., & Leopold, L. B. (1987). *Water in Environmental Planning* (p. 818). New York: W.H. Freeman and Company ISBN 0-71670079-4.

Erns, W. G. (2000). *Earth Systems: Processes and Issues*. UK: Cambridge University Press.

Fabricius, K. (2005). Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. *Marine Pollution Bulletin, 50*, 125–146. https://doi.org/10.1016/j.marpolbul.2004.11.028.

Garcia-Ruiz, J. M. (2010). The effects of land uses on soil erosion in Spain: a review. *Catena, 81*, 1–11. https://doi.org/10.1016/j.catena.2010.01.001.

Gardner, T. A., Cote, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean Corals. *Science, 301*(5635), 958–960. https://doi.org/10.1126/science.1086050.

Gray, S.C., Gobbi, K.L., & Narwold, P.V. (2008). *Comparison of Sedimentation in Bays and Reefs below Developed versus Undeveloped Watersheds on St. John, US Virgin Islands*. 11th International Coral Reef Symposium, Ft. Lauderdale, Florida, 7-11 July 2008 Session number 20.

Gray SC, Sears WT, Kolupski DeGrood AM, & Fox MD (2012) Factors affecting land-based sedimentation in coastal bays, US Virgin Islands. Proceedings of the 12th International Coral Reef Symposium, Cairns, Australia July 9th-13th, 2012.

Gudiño Elizondo, N. (2012). Erosión e intemperismo en las cuencas Coral Bay y Lameshur, Isla Saint John de Islas Vírgenes, EUA. Master Degree Thesis, Centro de Investigación Científica y de Educación Superior de Ensenada, México.

Gudino-Elizondo, N., Biggs, T., Bingner, R. L., Yuan, Y., Langendoen, E., Taniguchi, K., Kretzschmar, T., Taguas, E. V., & Liden, D. (2018a). Modelling ephemeral gully erosion from unpaved urban roads: Equifinality and implications for scenario analysis. *Geosciences, 8*, 137. https://doi.org/10.3390/geosciences8040137.

Gudino-Elizondo, N., Biggs, T., Castillo, C., Bingner, R. L., Langendoen, E., Taniguchi, K., Kretzschmar, T., Yuan, Y., & Liden, D. (2018b). Measuring ephemeral gully erosion rates and topographical thresholds in an urban watershed using unmanned aerial systems and structure from motion photogrammetric techniques. *Land Degradation & Development, 29*, 1896–1905. https://doi.org/10.1002/ldr.2976.

Gudino-Elizondo, N., Biggs, T., Bingner, R. L., Langendoen, E., Kretzschmar, T., Taguas, E. V., Taniguchi, K., Liden, D., & Yuan, Y. (2019). Modelling runoff and sediment loads in a developing coastal watershed of the US-Mexico Border. *Water, 2019*(11), 1024. https://doi.org/10.3390/w11051024.

Harden, C. P. (2001). Soil erosion and sustainable mountain development. *Mountain Research and Development, 21*(1), 77–83. https://doi.org/10.1659/0276-4741(2001)021[0077:SEASMD]2.0.CO;2.

Henry, A., Mabit, L., Jaramillo, R. E., Cartagena, Y., & Lynch, J. P. (2013). Land use effects on erosion and carbon storage of the Rio Chimbo watershed, Ecuador. *Plant Soil, 367*, 477–491.

Holland, H. D. (1978). *The Chemistry of the Atmosphere and Oceans*. New York: Wiley.

Hontoria, C., Rodriguez-Murillo, J. C., & Saa, A. (1999). Relationships between soil organic carbon and site characteristics in Peninsular Spain. *Soil Science Society of America Journal., 63*, 614–621. https://doi.org/10.2136/sssaj1999.03615995006300030026x.

Hubbard, D.K., Stump, J.D., & Carter, B. (1987). Sedimentation and reef development is Hawksnest, Fish and Reef Bays, St. John, U.S. Virgin Islands. Virgin Islands Resource

Management Cooperative. Biosphere Reserve Research Report No 21.

Jenny, H. (1961). Derivation of state factor equations of soils and ecosystems. *Soil Science Society of America Journal, 25*, 385–388. https://doi.org/10.2136/sssaj1961.03615995002500050023x.

Keesstra, S. D., Geissen, V., Van Schaik, L., Mosse, K., & Piiranen, S. (2012). Soil as a filter for groundwater quality. *Current Opinions in Environmental Sustainability, 4*, 507–516. https://doi.org/10.1016/j.cosust.2012.10.007.

Kinnell, P. (2005). Why the universal soil loss equation and the revised version of it do not predict event erosion well. *Hydrological Process, 19*, 851–854. https://doi.org/10.1002/hyp.5816.

Kolupski, M. (2011). Sedimentation in coastal bays with coral reefs: impacts of watershed development, Saint John, USVI. Master Degree Thesis, University of San Diego, USA.

LAStools - Efficient tools for LiDAR processing. Version 111216, Martin Isenburg 2011. http://rapidlasso.com/LAStools [accessed on June 2016].

MacDonald, L. H., Anderson, D. M., & Dietrich, W. E. (1997). Paradise threatened: land use and erosion on St. John, US Virgin Islands. *Environmental Management, 21*(6), 851–863.

MacDonald, L. H., Sampson, R. W., & Anderson, D. M. (2001). Runoff and road erosion at the plot and road segment scales, St. John, US Virgin Islands. *Earth Surface Processes and Landforms., 26*, 251–272. https://doi.org/10.1002/1096-9837(200103)26:3<251::AID-ESP173>3.0.CO;2-X.

Martinez-Mena, M., Lopez, J., Almagro, M., Boix-Fayos, C., & Albaladejo, J. (2008). Effect of water erosion and cultivation on the soil carbon stock in a semiarid area of South-East Spain. *Soil and Tillage Research, 99*, 119–129. https://doi.org/10.1016/j.still.2008.01.009.

McCreery, H.F. (2007). "The effect of Anthropogenic Development on Sediment Loading to Bays on St. John, U.S. Virgin Islands, Master Degree Thesis. Massachusetts Institute of Technology, USA.

McHugh, M. (2007). Short-term changes in upland soil erosion in England and Wales: 1999 to 2002. *Geomorphology, 86*(1-2), 204–213. https://doi.org/10.1016/j.geomorph.2006.06.010.

Millward, A. A., & Mersey, J. E. (1999). Adapting the RUSLE to model soil erosion potential in a mountainous tropical watershed. *Catena, 38*(2), 109–129. https://doi.org/10.1016/S0341-8162(99)00067-3.

Moore, D. M., & Reynolds, R. C., Jr. (1996). *X-ray diffraction and the identification of clay minerals*. New York: Oxford University of Press.

National Center for Environmental Information. NOAA, 2011. http://data.nodc.noaa.gov/coris/data/NOAA/nos/Summit2Sea/USVirginIslands/. Accessed 15 July 2011.

Oliver, L. M., Fisher, W. S., Fore, L., Smith, A., & Bradley, P. (2018). Assessing land use, sedimentation, and water quality stressors as predictors of coral reef condition in St. Thomas, U.S. Virgin Islands. *Environmental Monitoring and Assessment, 190*, 213. https://doi.org/10.1007/s10661-018-6562-1.

Pait, A. S., Galdo, F. R., Jr., Hartwell, S. I., Apeti, D. A., & Mason, A. L. (2018). An assessment of nutrients and sedimentation in the St. Thomas East End Reserves, US Virgin Islands. *Environmental Monitoring and Assessment, 190*, 270. https://doi.org/10.1007/s10661-018-6628-0.

Radke, J. D. (1997). Detecting potential erosion threats to the coastal zone: St. John, USVI. *International Journal of Marine Geodesy, 20*, 235–254. https://doi.org/10.1080/01490419709388107.

Ramos-Scharron, C. E. (2012). Effectiveness of drainage improvements in reducing sediment production rates from an unpaved road. *Journal of Soil and Water Conservation, 67*(2), 87–100. https://doi.org/10.2489/jswc.67.2.87.

Ramos-Scharron, C. E. (2018). Land disturbance effects of roads in runoff and sediment production on dry-tropical, settings. *Geoderma, 310*, 107–119. https://doi.org/10.1016/j.geoderma.2017.08.035.

Ramos-Scharron, C. E., & LaFevor, M. C. (2016). The role of unpaved roads as active source areas of precipitation excess in small watersheds drained by ephemeral streams in the Northeastern Caribbean. *Journal of Hydrology, 533*, 168–179. https://doi.org/10.1016/j.jhydrol.2015.11.051.

Ramos-Scharron, C. E., & MacDonald, L. H. (2005). Measurement and prediction of sediment production from unpaved roads, St John, US Virgin Islands. *Earth Surface Process and Landforms, 30*, 1283–1304. https://doi.org/10.1002/esp.1201.

Ramos-Scharron, C. E., & MacDonald, L. H. (2007a). Runoff and suspended sediment yields from an unpaved road segment. *Hydrological Processes, 21*, 35–50. https://doi.org/10.1002/hyp.6175.

Ramos-Scharron, C. E., & MacDonald, L. H. (2007b). Development and application of a GIS-based sediment budget model. *Journal of Environmental Management, 84*, 157–172. https://doi.org/10.1016/j.jenvman.2006.05.019.

Rankin, D.W. (2002). Geology of St. John, U.S. Virgin Islands. U.S. Geological Survey Professional Paper 1631. U.S. Department of Interior, U.S. Geological Survey.

Rawlins, B. G., Ferguson, A. J., Chilton, P. J., Arthurton, R. S., & Rees, J. G. (1998). Review of agricultural pollution in the caribbean with particular emphasis on small island developing states. *Marine Pollution Bulletin., 36*, 658–668. https://doi.org/10.1016/S0025-326X(98)00054-X.

Renard, K. G., Foster, G. R., Weesies, G. A., McCool, D. K., & Yoder, D. C. (1996). *Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). USDA Handbook Number 703.* Washington, D.C., USA: USDA.

Rogers, C. (1990). Responses of coral reefs and reef organisms to sedimentation. *MARINE ECOLOGY PROGRESS SERIES. Vol., 62*, 185–202.

Rogers, C.S., & Teytaud, R. (1988). Marine and terrestrial ecosystems of the Virgin Island National Park and biosphere reserve. U.S. Department of Interior. Biosphere reserve report No. 29.

Snider, D. (1972). National engineering handbook; section 4: Hydrology; chapter 16: Hydrographs Natural Resources Conservation Service.

Trimble, S. W., & Crosson, P. (2000). U.S. soil erosion rates: myth and reality. *Science, 289*(5477), 248–250. https://doi.org/10.1126/science.289.5477.248.

U.S. Department of Agriculture (1986). Natural Resources Conservation Service. Urban Hydrology for Small Watersheds, TR55. https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044171.pdf. Accessed 15 March 2017.

Wemple, B. C., Browning, T., Ziegler, A. D., Celi, J., Chun, K. P., Jaramillo, F., Leite, N., Ramchunder, S. J., Negishi, J. N., Palomeque, X., & Sawyer, D. (2017). Ecohydrological disturbances associated with roads: Current knowledge, research needs, and management concerns with reference to the Tropics. *Ecohydrology, 11*, 1–23. https://doi.org/10.1002/eco.1881.

Williams, J. R., & Berndt, H. D. (1972). Sediment yield computed with universal equation. *Journal of Hydraulic Division, ASCE, 98*(12), 2087–2098.

Wischmeier, W. H. (1976). Use and misuse of the universal soil loss equation. *Soil and Water Conservation, 31*(1), 5.

Wischmeier, W.H., Smith, D.D. (1978). Predicting rainfall erosion losses – a guide to conservation planning. U.S. Department of Agriculture handbook No. 537.

Woodbury, R. O., & Weaver, P. L. (1987). *The vegetation of St. John and Hassel Island, USVI National Park Service, Southeast Region, Research/Resources Management Report SER-83*. Atlanta: Georgia.

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