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Linking Soil Loss to Sediment Delivery in Two Estuaries in Puerto Rico

Nekesha Bernadette Williams

University of South Florida

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Linking Soil Loss to Sediment Delivery in Two Estuaries in Puerto Rico

by

Nekesha Bernadette Williams

A dissertation submitted in partial fulfillment of the requirements for the degree of Doctor of Philosophy
College of Marine Science
University of South Florida

Co-Major Professor: Ashanti Johnson, Ph.D.
Co-Major Professor: Barnali Dixon, Ph.D.
Albert Hine, Ph.D.
Mark Luther, Ph.D.
Stacy Nelson, Ph.D.

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Keywords: Erosion, GIS, Radionuclide, RUSLE, Sedimentation

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DEDICATION

I dedicate this work to my mothers. I come from a line of strong women! My paternal grandmother (Marisa James) became a market vendor to support her large family upon the death of her husband. My maternal grandmother (Mathina Williams), cannot read or write, but is one of the wisest person that I know. My birth mother (Wilma Francis), a teenage, single mother sacrificed a lot so that her only child can have a good education and build a good life for herself. My aunts were also the epitome of strength. It seem as though there was nothing they could not do and always faced challenges head on. We carry the same DNA so anything less is unacceptable! I thank God for their strength and for their being wonderful examples. To my cousins Abigail, Julia and Jo-Ann who are more like sisters than cousins, always by my side and without even knowing it, great stress relievers. To the men in my family, keeping on being men!

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ABSTRACT

Enhanced soil loss from the watershed is a major environmental issue. Increased soil loss from a watershed can potentially increase sediment delivery and loading to aquatic ecosystems such as rivers and estuaries. An increase in sediment delivery and loading to freshwater and transitional marine ecosystems can impact water quality and supply specifically by: (1) reducing water clarity, (2) transporting nutrient and pollutant laden sediments and (3) reducing the storage capacity of reservoirs. To address these broader environmental impacts of increased sediment delivery and loading to aquatic ecosystems it is imperative that potential source areas of sediments available for transport are identified in the watershed. It is also important that sediment source areas are linked to sediment transport and delivery to aquatic ecosystems.

This study attempted to establish a link between soil loss from watersheds and sediment delivery in two estuaries on the island of Puerto Rico. The two estuarine systems used in this investigation were the Rio Espiritu Santo (RES) riverine-estuarine system and the Jobos Bay (JB) Estuary. Soil loss from both study watersheds was estimated using RUSLE. Sediment cores and surface grab samples were collected from both estuaries. In addition, soil samples were collected from the two watersheds. Gamma analyses were performed in order to measure activity concentrations of $^{137}\text{Cs}$ and excess $^{210}\text{Pb}$ in sediment cores, surface and soil samples. $^{137}\text{Cs}$ inventories were computed for each core collected from both watersheds. Also, grain-size and LOI were...
performed on the sediments to describe the sedimentological characteristics of collected sediment cores, surface samples and soil samples.

A conceptual framework was developed and implemented for linking sediment production, availability (supply), transport and delivery to study estuaries. Results from the RUSLE model indicated that soil loss within both watersheds were low with patchy instances of erosional hotspots. These results did not provide any information on sediment supply or insights into the hydrologic connectivity of the study watersheds. $^{137}$Cs inventories computed from the RES watershed indicated that sediment cores located further upstream had the highest inventories. With reference to the JB Estuary, statistical analysis showed that location had an effect on distribution of $^{137}$Cs in surface samples within the bay. Sedimentological characteristics varied between cores.

The implementation of the conceptual model in both study watersheds allowed for the identification of potential source areas of sediments that were available for transport and delivery to adjacent aquatic systems. This investigation revealed that to link soil loss to sediment delivery it is essential that key processes and variables (rainfall, soils, LULC and geomorphology) must be included in the analysis. This conceptual model may be a valuable tool for monitoring and managing soil loss within the watershed and consequently, addressing problems of increased sediment delivery to aquatic and transitional marine ecosystems such as estuaries.
CHAPTER 1
BACKGROUND

1.0. Introduction
Soil erosion is a natural phenomenon that has been enhanced by human activities such as agriculture, urbanization, deforestation and mining (van Andel et al., 1990; Uri, 2001). Accelerated soil erosion, often a product of land use change, is a global issue, which can impact economies and natural ecosystems worldwide (Pimentel et al., 1995; Coulter and Ortega-Larrocea, 2006). Soil detached from the earth’s surface may be transported via surface overland flow to adjacent stream networks where it may be deposited in reservoirs and/or downstream coastal environments such as estuaries (Nagle et al., 1999; Edgar et al., 2000; Tamene et al., 2007). Increased sediment discharge can reduce the capacity of reservoirs to store water, increase turbidity in aquatic systems, and introduce nutrients and toxic chemicals to sensitive coastal ecosystems (Kennish, 2002; Verstraeten et al., 2002). Recognizing that activities within a watershed may have deleterious impacts on coastal ecosystem functions, an understanding of landscape processes and their influence on soil erosion, sediment yield and sedimentation rates is critical when developing and implementing management practices within watersheds and its associated coastal environments including the protection of aquatic ecosystems.

2.0. Literature Review

2.1. Soil Erosion
Soil erosion, the process whereby soil is detached, transported and deposited, is one of the most ubiquitous environmental issues plaguing the world (Wei et al., 2009).
Erosion is part of the natural denudation process. Various human activities have accelerated this natural process, resulting in both on-site and off-site impacts (Strahler and Strahler, 1998). Impacts of soil erosion include: reduction in soil fertility and productivity, desertification as well as the reduction of water quality and supply (Lal, 2003 and Bossio et al., 2010). Given the heightened consciousness of the potential influence of humans on the global carbon cycle, attention is now given to soil erosion and its’ influence on the carbon budget (Lal, 2003; Hancock et al., 2010). Factors influencing soil erosion in watersheds include: climate, parent material, topography, land use/land cover and conservations (Morgan, 1979). The combined effect of such variables may determine erosional rates and sediment yields for a given area. Determining erosion rates and sediment yields are critical as enhanced sediment loss from the landscape can increase sediment deposition (sedimentation) in coastal environments.

2.2. Sedimentation
Sediment deposition in coastal environments is also a natural phenomenon. Loose material (specifically sediment) originating from upland areas may move through a watershed by wind or water sometimes accumulating in streams, reservoirs as well as various coastal ecosystems such as estuaries (Jain et al., 2001). Rivers alone constitute about 85 percent of the total solid material that enter marine systems (Open University Course Team, 1999; Anderson et al., 2004). A variety of materials that may be characterized as radioactive and/or hazardous have been released either directly or accidentally into the environment; many of which will eventually make their way into aquatic systems. Adsorption of these materials to sediments is a major pathway by which such material can be introduced into nearshore coastal environments (Barros et al., 2004).
The potential to harbor harmful pollutants indicate the importance of monitoring and measuring sediment accumulation rates in these highly sensitive areas. An increase in sediment flux to coastal waters can also impact important habitat and nursing areas for many fish and invertebrate species (Ellison, 1998; Edgar et al., 2000). In order, to protect these ecologically significant areas, it is necessary to understand the connection between landscape processes and coastal dynamics, with special emphasis on the land-ocean interface.

2.3. Land-River-Coast-Ocean Interface

Linking soil erosion to sedimentation in-stream and in coastal ecosystems such as estuaries is critical in land and coastal management efforts (Clark et al., 1985; Lane et al., 1997; Rawlins et al., 1998). Research focused on land-ocean interactions is essential when addressing such complex environmental issues. Tackling soil erosion and consequently sedimentation in coastal environments requires an understanding of sediment transport, delivery and deposition within and outside of the landscape (Jain and Kothyari, 2000; Prosser et al., 2001). Pringle (2003) defines hydrologic connectivity as the “water mediated transfer of matter, energy and/or organisms within or between elements of the hydrologic cycle”. Under natural conditions, it can be assumed that the transfer of material between hydrological compartments may not result in ecosystem degradation (Montgomery, 2007). On the other hand, due to anthropogenic alterations of the landscape, connectivity may be disrupted and as a result, erosion and sedimentation regimes may be affected, potentially leading to degradation of soil and water quality (Lexartza-Artza and Wainwright, 2009). Allan (2004) discusses the effects of landscape changes on stream and river quality. The author highlights anthropogenic activities in the
watershed adversely impacting the ecological integrity of streams and rivers as a result of
the high connectivity between lotic systems and the adjacent landscape (Allan, 2004).
Verbist and others (2010) demonstrate this intimate linkage between human activity and
sediment loading to river systems in the Sumberjaya, sub-district, West Lampung,
Sumatra, Indonesia. In addition to impacts on stream and river systems, landscape
changes result in increased soil loss from watersheds and enhanced sedimentation to
coastal environments such as estuaries are major concerns to coastal resource managers
(Prosser et al., 2001). Figure 1.1 is a simplified representation of this complex land-
-ocean interface.

Figure 1.1 Simplified representation of the sediment production, sediment transport and
sediment deposition within landscape and coastal environments.
2.4. Estuaries
Estuaries are unique ecosystems that are located at the interface between land and ocean (Cardoso da Silva and Carmona Rodrigues, 2004). Due to their location in the landscape, estuaries are recipients of material from both marine and land areas (Yeager et al., 2006). Enhanced sediment inputs from upland areas are a major concern for coastal managers as sediments within estuaries can alter benthic habitats, increase turbidity and introduce potentially toxic pollutants to these environments (Larcombe and Woolfe, 1999; Mai et al., 2002; Thrush et al., 2003; Acevedo-Figueroa, 2006). Various studies have been conducted to examine the relationship between landscape properties and estuarine sediment contamination (Paul et al., 2002; Hale et al., 2004 and Rodriquez et al., 2007). Paul and others (2002) evaluated the influence of landscape metrics on estuarine sediment conditions in the Mid-Atlantic and Southern New England Regions. The landscape metrics used in this study was divided into two categories (1) land cover pattern and (2) point source pollution inputs. Authors discovered that sediment contamination of small estuaries was related to land cover and pollution inputs to these ecosystems (Paul et al., 2002). This study demonstrates that estuaries with watersheds less than 260 km² are responsive landscape metrics.

In a similar study conducted by Hale et al. (2004), authors identified specific land metrics contributing to estuarine benthic conditions. Results from this study indicated that landscape metrics are related to estuarine conditions and biodiversity of estuarine bottom communities (Hale et al., 2004). Specifically, authors observed that watersheds with a higher percent composition of wetlands, low urban and agricultural areas tend to be associated with higher benthic indices and higher biodiversity (Hale et al., 2004). Rodríguez and others (2007) attempted to model land-ocean interactions in three regions.
within the Virginian Biogeographic Province. Authors suggested that using land use/cover type to predict estuarine conditions can be challenging. In addition, it was observed that data amount, type and quality may produce better models relating human activities to coastal ecosystem health (Rodriguez et al., 2007). This study also highlights the importance of considering factors in the watershed as well as surveying coastal environments.

2.5. Factors Influencing Soil Erosion

2.5.1. Climate

Climate is one of the most important factors in the erosion process as it provides the energy necessary for soil to be detached and moved from its place of origin. Agents of erosion include: wind, water, ice and other geologic agents (Crosson, 1997). Of these, water and wind are considered to be the major types. Wind erosion is often associated with dry-arid climates, ice with cold areas and water in humid regions (Song et al., 2005). Water erosion, which is considered to be the most destructive form of erosion (Wei et al. 2009), is the primary focus of this dissertation. Precipitation, specifically its intensity and duration, is critical in determining the erosivity of a soil. The general premise is that the greater the intensity and the longer the duration of the rainfall event, the more erosion that will occur (Clark et al., 1985). There are many forms of water erosion. These sub-processes of water erosion include and are not limited to: sheet erosion, rill erosion, interrill erosion, gully erosion, pipe erosion and tunnel erosion (Bryan, 2000). While all may not work in isolation of each other, they may all produce varying effects on a single landscape (Bryan, 2000).
2.5.2. Soil erodibility

While precipitation provides the energy where by soil is eroded and transported, the properties of soil determine the degree of soil loss in a given area (Rhoton et al., 2007). Soil erodibility depends upon the intrinsic properties of the soil. Some of the properties of soil, which contributes to its susceptibility to erosion include: chemical composition, percent organic matter permeability, moisture content and many more (Song et al., 2005; Perez-Rodriguez et al., 2007). The chemical composition and organic matter content are important properties influencing soil aggregation or soil structure (Morgan, 1979). Soils with stable aggregates tend to be more resistant to the erosive power of runoff (Franzluebbers, 2002). While any or many soil properties will influence erosion it is aggregate stability in combination with shear strength and consistency, which may have a major impact on soil erosion (Bryan, 2000; Rhoton et al., 2003). With this understanding, it can be assumed that soils with poor aggregate stability have a greater susceptibility to erosion than ones with more aggregates (Linsley et al., 1982).

A variety of soil aggregate/erodibility indices have been developed and/or used to evaluate variation of soil loss under differing soil phases (Bryan, 1968). Rhoton and others (2003) developed an aggregate/erodibility index for soil in southeastern Arizona. Singh and Khera (2008) investigated soil erodibility under different land use in relation to soil loss and runoff from a plot in Punjab, India. For Ando soils in Kyusyu, Japan, Egashira and others (1983) identifies soil aggregate as an appropriate index for estimating soil erodibility.

Prior to looking at soil properties, it is important to determine the susceptibility of parent material to soil erosional processes. Ozdemir and Askin (2003) evaluated the vulnerability of geological parent material to erosion. The parent material used in this
investigation include: alluvial, andesite, basalt and gypsum (Ozdemir and Askin, 2003). Authors determined that of the four types of parent material, gypsum had the greatest soil erodibility potential whereas basalt had the lowest potential. In addition, they observed that andesite had the greatest amount of aggregates, while gypsum had the least of the four types of parent material (Ozdemir and Askin, 2003). Tamene and others (2006) analyzed variables influencing sediment yields in northern Ethiopia. One of the factors evaluated was lithology. Authors observed that shale and marl were highly erodible thus producing high sediment yields whereas sandstones and metavolcanic material were considered less erodible (Tamene et al., 2006).

2.5.3. Topography
Slope length and steepness affects erosion in that the longer and steeper the slope the more erosion that will occur (Clark et al., 1985). Tamene and others (2006) examined the influence of several variables on soil erosion/sediment yield in a mountainous dryland area in Northern Ethiopia. Some of the variables used in this investigation included: land cover, slope, catchment area, mean annual precipitation, parent material, gully erosion and many more (Tamene et al., 2006). Through the use of statistical analyses such as Pearson’s Correlation, Step-wise regression and multiple regression analysis, the key variables that exhibited the greatest influence to sediment yields were identified. This study demonstrated that terrain gradients, land cover, lithology and gully erosion can increase sedimentation in reservoirs (Tamene et al., 2006).

Vetter (2007) examined the influence of slope angle, geology and population and livestock densities on soil erosion with four areas within the Herschel district of South Africa. This author observed that severe cases of erosion appeared on sedimentary rocks
and alluvium while areas covered by dolerite and basalt seem to be more resistant to erosion. In the case of slope angle, flat or gently sloping areas (<10 degrees) were most affected by severe erosion; Vetter attributed this to sheet erosion. On steeper slopes, only small areas appeared to have experienced severe erosion (Vetter, 2007).

2.5.4. Land use/land cover and conservation practices

With increasing population growth and consequently a rising need for food, shelter and goods and other services, major land conversion is occurring on a local and global scale. Through such activities, natural ecosystems and resources have experienced varying levels of degradation (Foley et al., 2005). A critical impact of land use/land cover changes (LULC), which is a concern for many, is soil erosion not only at present, but also in the distant past (van Andel et al., 1990, Trimble and Crosson, 2000 and Montgomery, 2007).

Vegetation has a large impact on soil erosion as it expresses the degree to which the earth’s surface is exposed to an erosive agent such as water. Land under differing usage can experience different levels of soil loss. Kosmas and others (1997) investigated soil loss from a variety of landscapes in the northern Mediterranean region. Authors compared soil loss from agricultural lands with the following crops: vines, olives, rain-fed cereals, eucalyptus plantation or natural shrubland. They observed from this study that in hilly areas where vines were planted experienced the greatest soil loss (Kosmas et al., 1997). They also observed that some croplands, experienced seasonal soil loss variations (i.e. rain-fed cereal) (Kosmas et al., 1997). In contrast, certain vegetative covers can serve to reduce erosion potential in catchments; for example, forests (Fu et al., 1999; Jun et al., 2010; Ouyang et al., 2010).
The adaption of conservation practices within catchments can limit the amount of soil loss from erosion prone areas thus being available for transport to adjacent waterways and downstream areas. Types of conservation used depend upon the land use practice. In areas where clear cutting of forest is occurring, land managers may utilize forest best management practices (BMPs) such as establishing or maintaining stream riparian areas, pre-harvest planning, stream crossings and construction of forest roads (Aust and Blinn, 2004). In some instances, forest roads may produce soil erosion; in this case, an erosion mat can be used to reduce sediment yield from these sediment source areas (Grace, 2000). With reference to agriculture, applying some cover material such as mulch or grass can reduce soil loss (Ngatunga et al., 1984). In the event that conservation practices are not adopted or removed in a particular area, such areas may experience soil loss (Martinez-Casasnovas and Sanchez-Bosch, 2000). To predict and monitor soil loss, a variety of techniques have been developed and employed. The section below discusses these techniques in more detail.

2.6. Soil Erosion: Prediction and Monitoring

A variety of methods have been developed and used to monitor and measure soil erosion. Soil loss can be directly measured through the use of erosional plots (Van wallenghem et al., 2010); however other indirect methods for monitoring soil erosion have been employed. For instance, environmental radionuclides such as $^{137}\text{Cs}$ and $^{210}\text{Pb}$ have been used to assess soil erosion and sediment yield in river basins (Walling, 1999). Another method, involves the application of erosion models to spatially and temporally address soil erosion in the landscape (Merritt et al., 2003). The use of models for
measuring soil erosion have gained great acceptance as it is argued that using methods such as erosion plots do not consider major soil types (Okoba and Sterk, 2006). Erosional models such as the Soil Water and Assessment Tool (SWAT), Water Erosion Prediction Project (WEPP), Agricultural Nonpoint Source (AGNPS) model and the Revised Universal Soil Loss Equation (RUSLE) have been developed to predict spatially distributed soil erosion and sediment yields (e.g. Leon et al., 2004; Shi et al., 2004). Though these models were created for a similar purpose, they vary in complexity with reference to there spatial and temporal limitation. By testing models that vary spatially and temporally, the investigators can make suggestions as to what models may be applicable to data poor countries. Since, data limitation may serve as a hindrance to model application (Jha and Chowdary, 2007). However, by integrating geospatial technologies such as Geographic Information System (GIS) and Remote Sensing (RS), such models can be applied on larger scales (Hartkamp et al., 1999; Baigorria and Romero, 2007). Furthermore, these models may be relatively successful in predicting sediment loss within watersheds however, it is necessary to validate these models. Uncertainty analyses must be conducted in order to determine the model accuracy. Due to this uncertainty, understanding the major process involved in sediment production and transport may assist in improving model accuracy.

2.7. Sediment Production and Transport

2.7.1. Sediment transport capacity

An important component of monitoring soil loss in watersheds is estimating and/or modeling sediment flux. The concept of sediment transport capacity can be utilized to estimate flux (Prosser and Rustomji, 2000). The rate at which soil is eroded
depends on soil detachment and transport capacity of runoff (Julien and Simons, 1985). Mapping and modeling sediment transport capacity within watersheds can help identify potential sediment source areas (Rustomji and Prosser, 2001). Julien and Simon (1985) developed an equation to predict sediment transport based on a power relationship between slope and discharge (Equation 1.1). Values for exponents were developed from empirical experimentation/formulation/relationships.

\[ q_s = \phi q^m (\sin \beta)^n \tau^{\epsilon} (1 - \tau_0/\tau)^\epsilon \]  

**Equation 1.1**

where: \( q_s \) represents sediment flux \([\text{kg m}^{-1} \text{s}^{-1}]\); \( q \) is water flux \([\text{m}^3 \text{m}^{-1} \text{s}^{-1}]\); \( \beta \) is slope angle, \( i \) is rainfall intensity \([\text{m}^1 \text{s}^{-1}]\); \( \tau_0, \tau \) are critical shear stress and shear stress \([\text{Pa}]\), respectively and \( m, n, \phi, \epsilon \) are empirical or physical based coefficients.

Mitasova et al., (1996) have developed an alternative sediment transport equation that is used in the Unit Stream Power – Erosion and Deposition model (USPED) model. The variables of this model were obtained from the Universal Soil Loss Equation (USLE)/Revised Soil Loss Equation (RUSLE) (Equation 1.2). The underlying assumption of this model is that sediment flow can be estimated from sediment transport capacity.

\[ T = R^* K^* C^* P^* A^m (\sin b)^n \]  

**Equation 1.2**

where: \( T \) is sediment transport capacity, \( R \) is the rainfall-runoff erosivity factor \([\text{MJ mm/ ha hr yr}]\); \( K \) is soil erodibility factor \([\text{tonne ha hr/ ha MJ mm}]\); \( C \) is the cover-management factor \([\text{Unitless}]\); \( P = \) is the conservation practice factor \([\text{Unitless}]\); \( A \) represents the upslope contributing area per unit contour width\([\text{m}]\); \( b \) is degree slope; \( m, n \) represents constants that vary according to soil properties and flow type (Mitasova et al. 1996; Mitasova et al., 2001); and \( A^m \) is used as a measure of water flux (Mitasova et al., 1996).
2.7.2. *Sediment yield and sediment delivery*

Sediment yield is often defined as sediment discharge from a watershed (Lane et al., 1997). Units of measure associated with this variable is mass per unit time, or in other terms, it is a measure of how much sediment is moving out of a watershed area over a period of time (Lane et al., 1997). Estimation of sediment yields is important in addressing issues associated with reservoir sedimentation, channel morphology, water quality, conservation and planning (Kothyari and Jain, 1997). Sediment yield is not always measured therefore, sediment delivery ratio (SDR) is sometimes used to estimate this variable (Ouyang and Bartholic, 1997). There are many ways in which to compute SDR (Ouyang and Bartholic, 1997). The general equation for the computation of SDR is as follows:

\[ SDR = \frac{SY}{E} \quad \text{Equation 1.3} \]

where: \( SY \) is Sediment Yield and \( E \) is the gross erosion per unit area above a measuring point.

This equation estimates SDR based on knowledge of sediment yield and erosion over an area, however other factor relationships have been established with SDR.

USDA SCS (1972), Refro (1975) and Vanoni (1975) all established relationships between SDR and drainage area. In the case where, watershed topography is recognized as influencing SDR values, Maner (1958), Williams and Berndt (1972) and Williams (1977) developed SDR equations, which considered variables such as slope, gradient and relief length. In addition, SDR values have been estimated using particle size (Wallings, 1983). Since these earlier works, additional attempts have been made in developing a “more accurate” method for SDR prediction. One such effort was attempted by Diodata and Grauso (2009).
Diodata and Grauso (2009) acknowledge challenges associated with the estimation of SDR using other traditional and commonly used models. Authors proposed a new model, known as the SDR-Spatially Invariant Model (SDR\textsubscript{sim}). The rational driving the development of the SDR\textsubscript{sim} was to “skip over the limitations in other models…” (Diodato and Grauso, 2009). This model took into account the hydromorphology of an area in order to capture the influence of rainfall and watershed morphology on sediment yield/SDR. A comparison between SDR\textsubscript{sim} model and other general models such as the SDR –Area model and SDR Slope model indicated that these two simplified models performed poorly when evaluated against validated data. Authors suggested that SDR\textsubscript{sim} may be effective in predicting SDR values however, it was also mentioned that there may be limited applicability in utilizing this model at different sites around the world. Some of the key limitations observed by authors are: (1) parameters used in the model may not be sufficient in addressing the spatial variability of SDR data and (2) local parameters optimization may be appropriate when applying the/a model to specific areas due to geographic variability (Diodata and Grauso, 2009).

2.8. Sediment Geochronology

Estuarine areas can serve as a source of sediments to coastal oceans or it can be a sink for material as a result of river inputs (Gao and Collins, 1992). These ecosystems are considered to be prime areas for sediment deposition due to their locations in the landscape (Sanders et al., 2006). Sediments deposited in estuaries may serve as records for environmental history (Arcega-Cabrera et al., 2009). While natural processes may play a significant role in an estuarine system, anthropogenic influences are also of importance. Determining the dominant processes acting in these areas is critical (Hubeny
et al., 2009). Sediment geochronology is a technique used to investigate environmental history of watersheds.

A common method of determining environmental history is through the use of environmental radionuclides. Radiogeochronology is a sediment dating method, which involves the use of various radionuclides (Stout et al., 2002). The type of radionuclide used in the analyses depends on the level of accuracy desired and the time period of interest (Table 1). For instance, \(^{14}\text{C}\) a cosmogenic radionuclide can be used to determine historical averages (thousands of years) in sediments (Alvisi and Frignani, 1996). However, this involves many meters of sediments whereas isotopes such as \(^{137}\text{Cs}\) and \(^{210}\text{Pb}\) can be used to provide information from a few years to decades (Stout et al., 2002).

**Table 1.1. Examples of radionuclides commonly used as tools for dating.**

<table>
<thead>
<tr>
<th>Radionuclides</th>
<th>Half-Lives (yrs)</th>
</tr>
</thead>
<tbody>
<tr>
<td>(^{14}\text{C})</td>
<td>5,730</td>
</tr>
<tr>
<td>(^{137}\text{Cs})</td>
<td>30.2</td>
</tr>
<tr>
<td>(^{210}\text{Pb})</td>
<td>22.3</td>
</tr>
<tr>
<td>(^{239}\text{Pu})</td>
<td>2.4*10^4</td>
</tr>
<tr>
<td>(^{240}\text{Pu})</td>
<td>6.6*10^6</td>
</tr>
<tr>
<td>(^{226}\text{Ra})</td>
<td>1,620</td>
</tr>
<tr>
<td>(^{21}\text{Si})</td>
<td>276</td>
</tr>
<tr>
<td>(^{228}\text{Th})</td>
<td>1.91</td>
</tr>
</tbody>
</table>

*Data in table was adapted from information contained in Valette-Silver (1993).

Radioisotopic dating has been effectively applied to sediments sampled from lagoons, estuaries and other coastal environments (Valette-Silver, 1993; Ramesh et al., 2002; Stout et al., 2002; Jha et al., 2003). Coastal ecosystems are examples of changing environments that are influenced by changes in sea level, urban development and deforestation; all of which can alter the amount of sediment, organic and inorganic materials entering these environments (Open University Course Team, 1999; Panayotou,
2002). Although the sediment records for such environments may not be reliable, it was suggested that both $^{137}\text{Cs}$ and $^{210}\text{Pb}$ be used in such analyses (Kircher and Ehlers, 1998).

2.9. Common Radioisotopic Tracers

Two of the most common radionuclides used to monitor historical pollution trends and calculate sedimentation rates include $^{137}\text{Cs}$ and $^{210}\text{Pb}$. Even though other radionuclides have been used in various studies, these two radionuclides are considered the most appropriate when investigating modern sedimentation rates (Ramesh et al., 2002). The use of $^{210}\text{Pb}$ in geochronology studies was first suggested in 1963 by Goldberg; however this technique was not applied until 1971 in the dating of lake sediments (Appleby and Oldfield, 1978).

Total $^{210}\text{Pb}$, a naturally occurring radionuclide, is separated into two components: “supported” $^{210}\text{Pb}$ and “unsupported” or “excess” $^{210}\text{Pb}$. Supported $^{210}\text{Pb}$ is found in equilibrium with $^{226}\text{Ra}$, constituting only a small portion of the total $^{210}\text{Pb}$, whereas unsupported or excess $^{210}\text{Pb}$ is formed from the decay of $^{222}\text{Rn}$ in the atmosphere and water column and contributes a major portion of total $^{210}\text{Pb}$ found in the environment. The half-life of $^{210}\text{Pb}$ is 22.3 years (Rubio et al., 2003). $^{210}\text{Pb}$ is commonly used in geochronology. Similar to $^{137}\text{Cs}$, it is suitable for analyzing “recent” sediments (Panayotou, 2002).

$^{137}\text{Cs}$ is an anthropogenic nuclear fission product, which has been distributed and deposited in both aquatic and terrestrial systems worldwide, primarily through atmospheric fallout. This isotope adsorbs strongly to fine particles such as micaceous clays, silts and humic material and it is deposited within the sediment profile. One of the benefits of using this radioisotope to date sediments is that it can be historically
referenced through the occurrence of maximum global fallout, which occurred around 1963. The half-life of $^{137}\text{Cs}$ is approximately 30.2 yrs, which makes it suitable for measuring recent changes in the environment (Kirchner and Ehlers, 2002; Rubio et al., 2003).

3.0. Purpose and Objectives
The purpose of this study is to examine the relationship between soil erosion, sediment yields and sediment delivery in tropical estuaries, located in Puerto Rico. The overall goals for this study are to:

1. Describe watershed morphology and characteristics of study watersheds to link sediment yield, transport and delivery;
2. Identify sediment source areas for sediment delivery to an aquatic system in the context of supply-limited and transport-limited processes;
3. Conduct radionuclide inventories and analyses of sediment cores to link soil erosion processes to sediment deposition.
4. Analysis of statistical relationships between radionuclide distributions ($^{137}\text{Cs}$) and sediment characteristics and determine effects of location on the properties of sediment cores.

3.1. Significance of Study
There is increased concern over the impacts of upstream activities on downstream processes. To address complex environmental issues such as accelerated soil erosion and enhanced sedimentation rates in coastal ecosystems such as estuaries, innovative and integrative methodologies must be adopted and applied. Taking an integrative approach may lead to a better understanding of the interrelationship between coastal watersheds and their associated estuarine systems. In addition, this approach will contribute significantly to the science behind decision-making, which may lead to more effective land and coastal zone management. Hancock and others (2001) examined the dynamics
of sediment transport in Western Port, Australia by determining sediment accumulation rates within the bay.

4.0. Study Area
Puerto Rico is the fourth largest island in the Caribbean with an estimated area of 8,895 km² (Daly et al., 2003). The island is bordered by the Atlantic Ocean to the north and the Caribbean Sea to the south. With an estimated population density of 438 individuals/km², it is considered to be one of the most densely populated areas in the United States and its territories (Martinuzzi et al., 2007). Geographically, the island is mountainous with extensive coastlines in the north and south (Field, 2003; Boose et al., 2004). Cordillera Central, the mountain range of Puerto Rico, separates the island’s northern and southern coastal plains, the highest peak being Cerro de Punta, measured at an elevation of 1350 m (Malmgren and Winter, 1999). The climate of Puerto Rico is tropical (Malmgren and Winter, 1999). The central mountain range on the island intercepts the northeast trade winds hence the northern portions of the island receives a higher amount of rainfall as compared to the southern parts (Carter et al., 2000, Malmgren and Winters, 1999).

Puerto Rico is a volcanic arc-terrane. Volcanic rocks are categorized as volcaniclastic and epiclastic. Sedimentary rocks are from the late Jurassic to Pliocene and Eocene age and intrusive mafic and felsic plutonic rocks of the Cretaceous and early Tertiary age. Overlaying these rocks are young Olicene aged sedimentary rocks and sediments (Bawiec, 1999). Coastal resources on the island are in abundance (Hunter and Arbona, 1995). These resources include but are not limited to: beaches, mangrove forests and lagoon, and bioluminescent bays. Given the island’s high population density and its
unplanned urban/suburban expansions, fresh water and coastal resources are adversely impacted (Hunter and Arbona, 1995 and Martinuzzi et al., 2007). Some of greatest impacts to these systems include sedimentation, pollution and erosion (Field, 2003).

For this study, two coastal watersheds were chosen in Puerto Rico (Figure 1.2). The Jobos Bay (JB) watershed is situated on the south-central coast and the Rio Espiritu Santo (RES) is located in the north-eastern coast of Puerto Rico. The JB watershed was selected for this study because it drains directly into the Jobos Bay Estuary, which is part of the National Oceanographic and Atmospheric Administration-National Estuarine Research Reserve program (NOAA-NERRS). The headwaters of the RES drain a portion of the Caribbean National Forest (CNF).

![Figure 1.2. Location of study watersheds on the island of Puerto Rico](image)
4.1. *Rio Espiritu Santo (RES)*

The RES river and estuary system are located on the north-eastern coast of Puerto Rico, with a watershed area of approximately 75 km². Average annual rainfall received in that area is 3,600 mm (Covich et al., 2009). RES drains the Luquillo Experimental Forest (LEF), which is part of the CNF, the only tropical forest system within the United States Forest System (Pyron and Covich, 2003). RES is one of nine rivers that drains the CNF and flows out into the Atlantic Ocean. The headwaters of this river are geomorphically steep with large boulders and the occasional pool feature, which varies in depth (Figure 1.3; Covich et al., 2009). Downstream, the river enters a low elevation coastal-plains area, where the mouth or river drains into an estuarine area that is bordered by mangrove forests. Mangrove forests are a part of the vegetation in area designated as the Rio Espiritu Santo Natural Reserve (Figure 1.4). A low-head dam (also known as a weir is used to raise the level of a river or stream) was built in the upper reaches of the watershed approximately 5 km from the coast. This dam was constructed in 1984 by the Puerto Rico Aqueduct and Sewage Authority (Benstead et al., 1999).
Figure 1.3. Headwaters of Rio Espiritu Santo, Rio Grande, Puerto Rico.

Figure 1.4. Estuarine area of Rio Espiritu Santo, Rio Grande, Puerto Rico.
Geologically, RES is very diverse. Within its watersheds there are approximately nine different geological units while twenty eight different soil types were identified in the area (Figures 1.5 and 1.6). Land use follows a gradient with elevation, in the headwaters, 90 m upwards from the LEF boundary is covered by forest. Below that urban areas and pastoral lands become more frequent (March, 2001). It is projected that the Luquillo area will have the highest population growth on the island and as such, increased stress on the natural resources for the island is expected (Ortiz-Zayas and Scantena, 2004).

Figure 1.5. Geologic map units within RES watershed.
4.2. Jobos Bay (JB)

The JB estuary is the second largest estuarine bay on the island of Puerto Rico. The JB watershed is approximately 167 km$^2$ in size (Zitello et al., 2008). Average annual rainfall in the area is approximately 1130 mm (Laboy-Nieves et al., 2009). Watershed area is dominated by a variety of different land cover types, which includes: urban and agriculture in the north and mangrove wetlands; and small communities and industries in the south (Laboy-Nieves et al., 2009). Mangrove forest populates the shorelines of this watershed (Figure 1.7).
Within the JB watershed, 12 known geologic units and 29 soil types were identified (Figures 1.8 and 1.9, respectively). The predominant soil types within the watershed are beach deposits (calcareous sand), volcanic cobbles and coral reef debris in the upland areas. Lagoon and swamp deposits are within the mangrove wetland areas. Rio Seco, an intermittent creek, is the only source of freshwater that discharges (surface flow) directly into the bay (Laboy-Nieves et al., 2009). The bay area is approximately 31 km² and average depth is around four meters. Estimated residence time for a water mass in JB is 5.5 days (Laboy-Nieves et al., 2009). JB experiences a mixed, diurnal tidal signature and can be considered an intertidal estuarine system (Field et al., 2003).
Figure 1.8. Geologic map units within JB watershed.

Figure 1.9. Soil mapping units within JB watershed.
CHAPTER 2
WATERSHED GEOMORPHOLOGY, SOIL CHARACTERISTIC AND SOIL EROSION IN TWO WATERSHEDS IN PUERTO RICO

1.0. Introduction

Linking soil erosion to sedimentation in-stream and in coastal ecosystems such as estuaries is critical in land and coastal management efforts (Clark et al., 1985; Lane et al., 1997; Rawlins et al., 1998). Research focused on land-ocean interactions is essential when addressing such complex environmental issues. Tackling soil erosion and consequently sedimentation in coastal environments requires an understanding of sediment transport, delivery and deposition within and outside of the landscape (Jain and Kothyari, 2000; Prosser et al., 2001). Tropical coastal ecosystems are especially vulnerable to the impacts of soil erosion because of their natural environmental settings as well as anthropogenic activities such as urbanization, agriculture and deforestation and in some cases tourism (Baldwin, 2000; Dadson et al., 2003; Neto, 2003; Brooks et al., 2007; Prouty et al., 2008).

Mountainous tropical watersheds are considered erosion prone due to a highly erosive climate and rugged terrain (Dadson et al., 2003). Climate, the active force, provides the energy for soil detachment and removal from landscape (Morgan, 1979; Brooks et al. 2003). Precipitation amount, intensity and frequency influences soil detachment and transport (Clark et al., 1985; Nearing et al., 2005). In humid environments, “splash detachment” is a major process by which soil particles can be dislodged and made available for transport via surface overland flow (Martz, 1992). Topography, a passive force in the erosion process, influences flow depth and flow velocity (Brooks et al., 2003)
and Liu and Singh, 2004). The Mayor et al. (2009) study demonstrated the importance of
topographic parameters on runoff connectivity in the landscape.

Topographic parameters that are often measured are slope, aspect, profile and plan
curvatures (Zevenbergen and Thorne, 1987). Slope is the rate of inclination or gradient,
which can be expressed as a percentage or in degree measure (Pennock, 2003). Aspect is
the compass direction of slope (Pennock, 2003). Profile and plan curvatures pertain to
the shape or terrain. Profile curvatures describe concavity and convexity in the
landscape. It also makes reference to acceleration and deceleration of flow. Plan
curvatures refer to convergence and divergence of flow within the landscape (Pennock,
2003). Landform shape is relevant to soil erosion as it may illustrate the level of
connectivity within the landscape (Western et al., 1999; Leuven et al., 1985). Flow
pathways for water on the soil surface or through the soil are due to the landscape shape
and curvature (Gregorich et al., 1998). Pachepsky and others (2001) demonstrated that
topography had an influence on water retention in soils. Authors observed that less
concavity across of slope correlated to lower water retention potential (Pachepsky et al.,
2001). This observation alludes to the potential for increase erosion where there is a
dominance of convex slopes. Rieke-Zapp and Nearing (2005) observed that slope shape
influenced sediment yields. Uniform, nose and convex-linear slopes had greater
sediment yields when compared to concave linear and head slopes. Toeslopes were
identified as areas of sediment deposition (Rieke-Zapp and Nearing, 2005).

Topography and soil characteristics are also intimately linked to soil erosion
process (Flugel, 1997). Larsen and others (1999) observed that slope wash was higher on
sandy loam soils than on silty clay loam soils. Godsey and others (2004) have observed
that soil and microtopography are important determinants of flow paths (runoff) within watersheds. These flow paths in some respects can be viewed as potential sediment routing areas in the landscape. Soil characteristics such as infiltrability and hydraulic conductivity have also been evaluated for their potential influence on runoff generation (Zimmerman et al., 2006). Rhoton and others (2003) observed that runoff had an inverse relationship with infiltration on Ferrihydrite soils. Bonell and others (1981) observed that saturated overland flow in a tropical forest was influenced by soil storage capacity, temporal variations in rainfall intensity and spatial variability of soil hydraulic conductivity. Liu and Singh (2004) emphasized the importance of evaluating runoff and its influence on soil erosion critical to evaluate runoff and its generating processes as it influence erosion.

While climate, topography and soil play a major role in the soil erosion process, there are other factors such as vegetation and human activities, which contributes to this phenomenon (van Andel et al., 1990; Uri, 2001). All of these variables interact in complex and unique ways producing major challenge in soil loss estimation (Lu et al., 2004). Hancock (2009) assessed the influence of rainfall amount and intensity erosion and sediment transport using a numerical model for erosion and deposition. Different rainfall scenarios were modeled over a 1000 year period. Results from this study indicated that catchments did not differ from each other hydrologically and geomorphologically over the 1000 years simulation period (Hancock, 2009). Additionally, the model predicted that an increase in rainfall amount and intensity may also increase sediment transport rates. Gumbs and Lindsay (1982) reported that runoff and soil loss from plots undergoing agricultural activities in the northern range of
Trinidad were high due to seasonal variations in precipitation, crop type and slope. The Coulter and Ortega-Larrocea (2006) study highlighted the need for applying a soil geomorphological approach when evaluating soil erosion in a tropical dry forest ecosystem. As demonstrated by these studies, soil erosion varies considerably. Such variability adds to challenges when investigating this phenomenon in the landscape.

Soil erosion varies in time and in space. Geographic Information Systems (GIS) has the potential to address both spatial and temporal variability in soil erosion (De Roo, 1998). This technology has the capability to maintain large databases, which can describe spatial heterogeneity in landscape (Jain et al., 2010). In the past 30 years, with an increase in computer power, there has been a rise in the development of models to assess catchment erosion and sediment transport (Merritt et al., 2003). Examples of such models include: Universal Soil Loss Equation (USLE), Agricultural Non-Point Source Model (AGNPS), The Griffith University Erosion System Template (GUEST) and Hydrologic Simulation Program, Fortran (HSPF) to name a few. As stated by Merritt et al. (2003) “there is no best model” and model selection must be based on the intended application. While both GIS and erosion models have their limitations; both technologies when combined have the potential to identify the physical processes influencing soil erosion and sediment yields in the landscape (Fistikoglu and Harmancioglu, 2002; Renschler and Harbor, 2002; Boardman, 2006; Jain et al., 2010).

It is critical to measure or monitor erosional processes in tropical areas particularly in steep areas as mis-management could lead to impacts on-site and off-site (El-Swaify, 1997). A visual representation of geomorphology and soil characteristics
may provide insight into the processes contributing to soil loss within these tropical watersheds (Vitek et al., 1996).

The primary goal of this paper is analysis and visualization of landscape parameters such as geomorphology and soil characteristics as they play a critical role in soil erosion, sediment transport and hydrological processes within the watershed. The secondary goal of this paper is to compare potential effects of these characteristics on soil loss from two tropical watersheds on the island of Puerto Rico. The specific objectives are: (1) to investigate the spatial pattern of soil erosion using RUSLE and (2) to generate a spatial representation of geomorphic and soil characteristics in study watersheds.

2.0. Study Areas
Refer to chapter 1 study area section for detailed description of study sites including map.

3.0. Data Source and Methods
3.1. Data Source
Model input data layers for the Revised Universal Soil Loss Equation (RUSLE) were obtained from various sources and at various resolutions. The digital elevation model (DEM) for the island of Puerto Rico was obtained from the United States Geological Survey (USGS) at a resolution of 30 m. The dem was used to derive slope length factor (LS). K-factor raster layer was obtained from National Oceanographic and Atmospheric Administration (NOAA)-Center for Coastal Monitoring and Assessment (CCMA) Summit-to-Sea project at a resolution of 30 m (NOAA-CCMA, 2005). Land cover data was obtained from the National Land Cover Database (NLCD) for the year 2001 at a resolution of 30 m (USGS, 2003). Parameter-Elevation on Independent Slopes
Model (PRISM) data for the island of Puerto Rico at a resolution of 225 m was acquired from the Climate Source Inc. and used to derive erosivity factor (R). This dataset was resampled to a resolution of 30 m. All datasets were re-projected to similar coordinate system in order to maintain accuracy in analysis.

3.2. Topographic and Soil Analysis

DEM's are a common data source in hydrologic and geomorphic studies (Wise, 2000). A base dem for the island of Puerto Rico was used to generate geomorphic attributes for each of the study watersheds. The following attributed were extracted: slope, aspect, plan and profile curvature. Note that plan and profile curvatures, which are measures of convexity and concavity (Table 2.1). Soil characteristics were obtained from soil surveys and mapped using ArcGIS 9.2 (Boccheciaimp, 1977; Huffacker, 2002).

<table>
<thead>
<tr>
<th>Landform Curvature</th>
<th>Convex</th>
<th>Concave</th>
<th>Flat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Profile</td>
<td>Negative</td>
<td>Positive</td>
<td>Zero</td>
</tr>
<tr>
<td>Planform</td>
<td>Positive</td>
<td>Negative</td>
<td>Zero</td>
</tr>
</tbody>
</table>

3.3. Determination of RUSLE factors

RUSLE, a derivative of the original Universal Soil Loss Equation (USLE) has been applied in tropical areas (Cox and Madramootoo, 1998; Hoyos, 2005). Average soil loss by water erosion can be estimated using RUSLE via Equation 2.1.

\[ A = R \cdot K \cdot L \cdot S \cdot C \cdot P \]  

**Equation 2.1**

where:
A = is the average soil loss (tonne/ha year)
R = is the rainfall-runoff erosivity factor (MJ mm) (ha hr year)^{-1}
K = is soil erodibility factor (tonne ha hr) (MJ mm hr)^{-1}
LS = represents the slope length and slope steepness factors, respectively (unitless).
C = is the cover-management factor (unitless)
P = is the conservation practice factor (unitless)
3.3.1. Rainfall-runoff erosivity (R)
The erosivity factor (R) for both study watersheds were derived using PRISM dataset. PRISM is a statistical-geographical approach to mapping climate using point data, elevation data (DEM) and other spatial datasets to estimate climatic patterns on annual, monthly and event-based climatic elements. The resulting datasets are gridded and GIS compatible allowing for integration into RUSLE model (PRISM Guide, 1998). Annual PRISM estimates for the island of Puerto Rico were used to derive R. Rainfall distribution in each of the study watershed was extracted from the original PRISM raster mapping average annual rainfall for the entire island of Puerto Rico. To generate the original PRISM raster, 30-year average total precipitation from the time period 1963 - 1995 was used. The data used in this analysis was gathered from 109 stations (108 National Weather Service Stations and 1 University of Puerto Rico Station). The units of measurement for precipitation are mm (Daly, 2002).

Computation of R-factors for the individual watersheds was completed in a series of steps. In the first, individual PRISM precipitation maps for each of the study areas were extracted from the original raster layer. Second step involved the estimation of mean annual erosivity (MAE) (Equation 2.2). The input raster (P-mm/year) were the individual precipitation maps obtained in the first step. Thirdly, the resulting layers were then multiplied by the estimated \( I_{30} \) for each individual watershed (Equation 2.3). \( I_{30} \) is the maximum rainfall intensity over a 30-minute period of time. This value simply indicates how hard it rains (depth/time) (Renard and Friemund, 1994). The \( I_{30} \) values for the JB and RES watershed are 50.8 mm/year and 59.9 mm/year, respectively. \( I_{30} \) was obtained from the National Climate Data Center (NCDC, 2009). The original units of measure for this data were inches per year. These values were converted to mm per year by
multiplying values by 25.4 mm. Finally, the final raster was divided by 1000 in order to express R as (MJ mm) (ha hr year)^{-1}.

\[
\text{Mean Annual Erosivity} = (9.28 \times P - 8.838)
\]
\text{Equation 2.2}

where:
P = \text{mm/year}

\[
R = ([\text{MAE}] \times I_{30})
\]
\text{Equation 2.3}

where:
I_{30} = \text{mm/year}

3.3.2. Soil erodibility (K)
The K-factor map produced by National Oceanographic and Atmospheric Administration- Center for Coastal Monitoring and Assessment (NOAA-CCMA) Summit-to-Sea project was derived from the SSURGO soil database of the USDA-NRCS. K-factor maps for each of study watersheds were extracted from a K-factor map created for NOAA-CCMA Summit-to-Sea project. The resolution of the original raster was 30 m. These K-factor maps were metricized from US units into (tonne ha hr) (MJ mm hr)^{-1} using a conversion factor of 0.1317. This was accomplished using ArcGIS raster calculator. Typical range of K values in SI units is 0 to 0.1. Low K values (less than and equal to 0.025) indicates a low soil erodibility potential whereas higher K values (greater than and equal to 0.04) suggests that such soils have a greater erosion potential (Brady and Weil, 2008).

3.3.3. Slope length and steepness factor (LS)
LS factor maps were obtained from a 30 m dem that was mosaic into a seamless dem from data processed by the USGS. A multi-step process was required to compute the LS factor for study watersheds. Step 1. Involved the computation of slope from the dem using ArcGIS Spatial Analyst 9.2. Step 2. A flow accumulation map derived from
the previously created slope map using the flow accumulation tool in Spatial Analyst, Hydrology toolbox, 9.2. Equation 2.4 was employed in ArcGIS Spatial Analyst, Raster Calculator into complete calculation of flow accumulation:

$$\text{(Flow Accumulation (Flow Direction ([DEM])))} \quad \text{Equation 2.4}$$

The resulting raster was reclassed using the reclassification rules below:
0 = 0.5; >0 to <5 = values remains unchanged and 5 = 5.

The Army Corp of Engineers LS equation (2.5) was applied in ArcGIS Spatial Analyst, Raster Calculator:

$$\text{(Pow(FlowAccumulation)\times Resolution/22.1,0.4)\times}$$
$$\text{Pow(Sin([Slope]\times 0.01745)/0.09, 1.4)\times1.4) \quad \text{Equation 2.5}$$

3.3.4. Cover management (C) and conservation support-practices (P)

Land cover data for the year 2001 for the Commonwealth of Puerto Rico was acquired from NLCD established by the Multi-Resolution Land Characteristics Consortium (USGS, 2003). The land cover distinguishes between 16 classes. An additional nine classes are available in coastal areas (USGS, 2003). Land cover within each watershed was extracted using the watershed vector shapefile as a mask in ArcGIS 9.2. The resulting layer was then reclassed based on established C-factor values (Cox and Madramootoo, 1998 and Jabbar et al., 2005). Table 2.2 provides reclass values for selected land cover and source information. P-factor values typically range from 0 to 1. Values closest to zero represent the application of various conservation practices, which help reduce soil erosion. A value of one indicates that no conservation management practice is being applied in an area. In this study, P-factor value was assumed to be 1 for all land cover types.
Table 2.2. Reclassification values for selected land cover within study watersheds.

<table>
<thead>
<tr>
<th>Land Cover Type</th>
<th>C-factor Values</th>
<th>Information Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Open Water</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, Open Space</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, Low Intensity</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, Medium Intensity</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, High Intensity</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Barren Land</td>
<td>1.0000</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Evergreen Forest</td>
<td>0.001</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Shrub/scrub</td>
<td>0.01</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Herbaceous</td>
<td>0.01</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Cultivated Crops</td>
<td>0.12</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Woody Wetlands</td>
<td>0.001</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Emergent Herbaceous Wetlands</td>
<td>0.001</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Pasture</td>
<td>0.01</td>
<td>Jabbar et al., 2005</td>
</tr>
</tbody>
</table>

4.0. Results and Discussion

RUSLE was completed in an ArcGIS 9.2 interface. Figures 2.1-2.5 contain input layers created and used in soil computation. Inputs layers are described below.

4.1. R-Factor

Rainfall-runoff erosivity at both sites follows a general pattern with elevation. R-factor increases with increasing elevation at both watersheds. The highest R-factor values were estimated for the mountain ridges and lowest values are in the coastal plains (Figure 2.1). Rainfall erosivity factors ranged from 891 to 2560 (MJ mm) (ha hr year)$^{-1}$ within the RES watershed (Table 2.3). In contrast, R-factor for the JB watershed ranged from 423 to 871 (MJ mm) (ha hr year)$^{-1}$ as one moves from coastal plains to the mountain range (Table 2.4). About 29% of the RES watershed area shows R-factor value between 891 and 1207, which occurs mostly in the lower reaches (coastal plains) of the watershed (Table 2.5). Comparably, approximately 27% of the land area, primarily in the headwater/tropical rainforest region of the study watershed R-factor values is estimated at 2240-2560 (MJ mm) (ha hr year)$^{-1}$ (Table 2.5). Approximately 50% of the JB watershed
area, primarily in the coastal plains region shows R-factor values of 423-489 (MJ mm) (ha hr year)$^{-1}$ (Table 2.6). Higher R-factor values are found in the northern-eastern most parts of the study watershed. R-factor values ranged from 722-871 (MJ mm) (ha hr year)$^{-1}$ occurred only in 8% of the watershed (Table 2.6). RES is located on the north-eastern side of Puerto Rico, which is subjected to effects of the north eastern trade winds. This portion of the island receives a large amount of rainfall due to the orographic effect created by the La Cordillera Central mountain range as compared to the southern coastal areas where JB is situated. Differences in rainfall between the northern and southern parts of the island influences drainage density and flow persistence in stream network. The northern parts of the island have a greater amount of streams as compared to the southern part (Field, 2003).

Figure 2.1. Spatial distribution of R factor for RES and JB.
Table 2.3. Values for R, K, LS and C-factors in the RES watershed.

<table>
<thead>
<tr>
<th>R-factor (MJ mm)* (ha hr year)^{-1}</th>
<th>K-factor (tonne ha hr)^{-1} (MJ mm hr)^{-1}</th>
<th>LS-factor (unitless)</th>
<th>C-factor (unitless)</th>
<th>Soil Loss tonne (ha yr)^{-1}</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maximum 891.0</td>
<td>0.002</td>
<td>0.00</td>
<td>0.0001</td>
<td>0.00</td>
</tr>
<tr>
<td>Average 1725.5</td>
<td>0.019</td>
<td>43.5</td>
<td>0.5000</td>
<td>1069</td>
</tr>
<tr>
<td>Minimum 2560.0</td>
<td>0.036</td>
<td>87.0</td>
<td>1.0000</td>
<td>2158</td>
</tr>
</tbody>
</table>

Table 2.4. Values for R, LS, K and C-factors in the JB watershed.

<table>
<thead>
<tr>
<th>R-factor (MJ mm)* (ha hr year)^{-1}</th>
<th>K-factor (tonne ha hr)^{-1} (MJ mm hr)^{-1}</th>
<th>LS-factor (unitless)</th>
<th>C-factor (unitless)</th>
<th>Soil Loss tonne* (ha yr)^{-1}</th>
</tr>
</thead>
<tbody>
<tr>
<td>Minimum 423</td>
<td>0.002</td>
<td>0.00</td>
<td>0.0001</td>
<td>0.000</td>
</tr>
<tr>
<td>Maximum 871</td>
<td>0.036</td>
<td>80.0</td>
<td>1.0000</td>
<td>781.0</td>
</tr>
<tr>
<td>Average 647</td>
<td>0.019</td>
<td>40.0</td>
<td>0.5000</td>
<td>390.5</td>
</tr>
</tbody>
</table>

Table 2.5. Spatial distribution of R-factor values for RES

<table>
<thead>
<tr>
<th>R Factor</th>
<th>% Coverage</th>
<th>km^2</th>
</tr>
</thead>
<tbody>
<tr>
<td>891-1207</td>
<td>29.0</td>
<td>21.7</td>
</tr>
<tr>
<td>1207-1559</td>
<td>14.7</td>
<td>11.0</td>
</tr>
<tr>
<td>1559-1904</td>
<td>13.5</td>
<td>10.1</td>
</tr>
<tr>
<td>1904-2240</td>
<td>15.4</td>
<td>11.6</td>
</tr>
<tr>
<td>2240-2560</td>
<td>27.4</td>
<td>20.6</td>
</tr>
</tbody>
</table>
Table 2.6. Spatial distribution of R-factor values for JB.

<table>
<thead>
<tr>
<th>R Factor</th>
<th>% Coverage</th>
<th>km²</th>
</tr>
</thead>
<tbody>
<tr>
<td>423-489</td>
<td>50.3</td>
<td>69.0</td>
</tr>
<tr>
<td>489-555</td>
<td>19.5</td>
<td>26.8</td>
</tr>
<tr>
<td>555-634</td>
<td>12.0</td>
<td>16.4</td>
</tr>
<tr>
<td>634-722</td>
<td>10.2</td>
<td>13.9</td>
</tr>
<tr>
<td>722-871</td>
<td>8.0</td>
<td>10.9</td>
</tr>
</tbody>
</table>

4.2. K-Factor

K-factor is a measure of soil vulnerability to erosion due to its inherent properties. K-factor values for soils between watersheds were relatively similar (Figure 2.2; Tables 2.3-2.4). A large portion of the RES watershed showed K-factor values between 0.013-0.031 (tonne ha hr) (MJ mm hr)^{-1} covering approximately 73 % of the watershed area. About 2 % of the study area showed high K values of 0.031 to 0.036 (tonne ha hr) (MJ mm hr)^{-1}. These values were present primarily at the outlet of this riverine watershed, within the estuarine portion of the system. Similar to the RES watershed, K-factor values between 0.013 to 0.031 (tonne ha hr) (MJ mm hr)^{-1} occupied a large part of the JB study area, possessing an area coverage of about 66 %. Highest soil erodibility values were located along the coastline of the associated estuary in narrow strips. These locations coincide with mangrove forested areas. Overall, soils in RES and JB may be low to moderate. By evaluating key soil properties such as soil texture, soil hydrologic group and hydraulic conductivity provides insight into the runoff potential from various zones within the watershed.
4.3. **LS-Factor**

Increase in slope length and steepness corresponds with higher overland flow velocities and higher erosion potential (Onori et al., 2006). The effect of topography on soil erosion is represented by the LS factor in the RUSLE model. RES is characterized by steep slopes in the headwaters with minimum slopes in the coastal plains. The range of values of the LS factor was slightly greater for the RES watershed (0 to 87) when compared to those for the JB watershed (0 to 80). For the sake of comparison, the scales for LS factor values were standardized. Results indicated that approximately 40% of the RES study area shows an LS-factor value of 1-2, which covers an area of about 30 km². Less than 2% of the area had high LS-factor values of 4-5 (Figure 2.3). The spatial distribution of LS factor is presented in figure 2.3 and summarized in tables 4 and 5. About 70% of the JB watershed has LS-factor values of less than 1. Only about 2% of
the area has high LS-factor values between 4-5. A large portion of that watershed consists of coastal plains, which is primarily used for agriculture.

![Map of Rio Espiritu Santo and Johns Bay with LS-Factor legend](image)

**Figure 2.3.** Spatial distribution of LS-factor for RES and JB. Scale on maps were standardized for comparison purposes.

### 4.4. C and P-factor

Table 2.7 contains the percent distribution of selected land cover types in each of the watersheds. The dominant land use type in the RES watershed is evergreen forest (tropical rainforest), which covers approximately 62% of the watershed area. The headwaters of this river drain a portion of the Caribbean National Forest (CNF). Herbaceous-grasslands were also observed to occupy a large area in the RES. It is important to note that while this area may not experience active tillage, the land may be used for grazing purposes by farmers (USGS, 2003). While not included in table 2.7, the land cover type Hay/Pasture was classified in the RES watershed it occupied.
approximately 4% of the area. This land cover type was not identified in the JB watershed. In contrast, herbaceous-grasslands were the dominant land cover type in JB watershed, occupying approximately 50% of the land area (Table 2.7). Evergreen forests also dominated the landscape, covering an estimated 20% of the land area (Table 2.7). Nine percent of the JB watershed is used to cultivate crops while only 0.20% of the RES area was used for the same purpose (Table 2.7).

Figures 2.4 and 2.5 are the C-factor and P-factor input layers used in the computation of RUSLE for both watersheds. C-factor values ranged from 0.0001 to 1 overall. Approximately 69% of the RES study area had C-factor values of 0.0002 to 0.007 (Table 2.8). Less than 1% of the study area had C-factor values of 0.117 to 1 (Figure 2.4 and Table 2.7). This may be due to the fact that there is lower amount of exposed area that may be susceptible to erosion. Approximately 54% of the JB study area had C-factor values of 0.007 to 0.117 with approximately 10% of the study area had high C-factor values from 0.117-1 (Table 2.8). The JB watershed had a larger percentage of agricultural and grassland area than RES therefore, a possible reason for the 10% of this watershed possessing high C-factor values. In modeling soil loss from the watersheds it was assumed that P-factor, which describes the erosion control practices employed within the study area to be 1 (Figure 2.5). There is the potential for overestimation of RUSLE values in both watersheds as a P-factor of 1 suggests that no conservation practices are being applied in the watershed therefore, soil loss may maximize for these areas.
Table 2.7. Percent distribution of selected land cover types in study watersheds.

<table>
<thead>
<tr>
<th>Landcover Type</th>
<th>RES (%)</th>
<th>JB (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Development, Low Intensity</td>
<td>5.0</td>
<td>4.0</td>
</tr>
<tr>
<td>Development, Medium Intensity</td>
<td>3.0</td>
<td>5.0</td>
</tr>
<tr>
<td>Evergreen Forest</td>
<td>62.0</td>
<td>20.0</td>
</tr>
<tr>
<td>Cultivated Crops</td>
<td>0.20</td>
<td>9.0</td>
</tr>
<tr>
<td>Herbaceous (Grassland)</td>
<td>16.0</td>
<td>50.0</td>
</tr>
<tr>
<td>Woody Wetland</td>
<td>3.0</td>
<td>5.0</td>
</tr>
<tr>
<td>Barren Land</td>
<td>0.09</td>
<td>0.6</td>
</tr>
</tbody>
</table>

Table 2.8. Percent distribution of C-factor values in each watershed.

<table>
<thead>
<tr>
<th>C factor</th>
<th>RES (%)</th>
<th>JB (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.00001</td>
<td>10.9</td>
<td>11.3</td>
</tr>
<tr>
<td>0.0001-0.007</td>
<td>68.7</td>
<td>25.6</td>
</tr>
<tr>
<td>0.007-0.117</td>
<td>20.1</td>
<td>53.6</td>
</tr>
<tr>
<td>0.117-1</td>
<td>0.30</td>
<td>9.50</td>
</tr>
</tbody>
</table>

Figure 2.4. Spatial distribution of C-factor for RES and JB.
4.5. Soil Loss For RES Using RUSLE

Estimated soil loss potential for the RES watershed indicated that average soil loss for this watershed is approximately 1069 tonne (ha yr)$^{-1}$ (Table 2.3). Combined soil loss from the Guadiana watershed in Puerto Rico, which is almost half the size of RES is approximately 607 tonne (ha yr)$^{-1}$ (del Mar Lopez et al., 1998). In general, soil loss within the RES watershed was relatively low (actual soil loss estimate: 0-8 tonne (ha yr)$^{-1}$). There were a few cells, primarily in the headwaters that may be considered erosional hotspots (Figure 2.6). The total area of these hotspots is less than 0.1% (Table 2.9). Most of landscape experienced a potential soil loss of 1-2 tonne (ha yr)$^{-1}$ (Table 2.9). About 80% of the RES watershed is covered by natural vegetation such as tropical forests and herbaceous grasslands. Vegetation tends to protective the soil surface from the impacts of rainfall. In the case of the RES watershed, it can be assumed that the lower
than expected soil loss potential may be attributed to fact that a large portion of the watershed is covered by vegetation.

**Table 2.9.** Spatial distribution of RUSLE (A) in study watersheds.

<table>
<thead>
<tr>
<th>RUSLE (tonne (ha yr)$^{-1}$)</th>
<th>Spatial Coverage (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>RES</td>
</tr>
<tr>
<td>Less than 1</td>
<td>6.10</td>
</tr>
<tr>
<td>1-2</td>
<td>93.4</td>
</tr>
<tr>
<td>2-3</td>
<td>0.50</td>
</tr>
<tr>
<td>3-4</td>
<td>0.04</td>
</tr>
<tr>
<td>4-5</td>
<td>0.006</td>
</tr>
</tbody>
</table>

**Figure 2.6.** Potential soil loss with RES watershed. Erosional hotspots highlighted by blue circle.
### 4.6. Soil Loss For JB Using RUSLE

Average soil loss due to water erosion in the JB watershed is approximately 390.5 tonne (ha yr)$^{-1}$. A large portion (~99%) of the study area experiences a low erosion potential (Table 2.9). Zitello and others (2008) characterized subwatersheds in JB to determine potential effect of sediment input to benthic communities in bay. In this investigation, subwatersheds with steep slopes in the headwaters had a greater sediment contribution to the Bay when compared to those that are coastal and low elevation. A true comparison between soil loss from this study and Zitello et al. (2008) could not be made with great confidence as erosion potential was conducted at different scales. Authors reported that approximately 11% of JB watershed is used from agricultural, which is comparable to the 9% estimated from the land cover data used develop the C-factor variable in the present study (Table 2.7). Potential soil loss from coastal plains areas ranged from 0 to 33 tonne (ha yr)$^{-1}$, which is still indicative of a “low” annual erosion rate. Similar to RES, there are few areas within this watershed that may be characterized as erosion hotspots (Figure 2.7). JB has extensive coastal plains in its watershed and expected the dominant landform shape is flat. Being that the landscape is mostly flat, these areas may act as potential depositional surface for soil eroded in the headlands at higher elevation.
4.7. Watershed Morphology

4.7.1. Slope analysis

Slope analysis is an important part of assessing the vulnerability of the landscape to water erosion. Slope steepest affects runoff and sediment transport. It is expected that increasing slope gradient, increases surface runoff therefore, sediment transport capacity may be enhanced. Percent slope for RES ranged from 0-147 %. The majority of the study area (43 km$^2$) possessed gentle (9 %) to strong slopes (33 %) (Table 2.10, Figure 2.8). Steep slopes occupied approximately 4% of the watershed area. These areas are located primarily in the headwaters of the watershed where it is densely vegetated, which can stabilize soils reducing the potential for soil erosion. In the RES watershed, the low soil loss potential in the watershed and the complex terrain may reduce sediment yield.
(supply-limited system) and also limit transport of eroded material from one zone to the next (transport-limited system). In contrast flat areas (0-7 % slopes) dominate the JB watershed (Table 2.11, Figure 2.9). Less than 4 % of this watershed possesses extreme slopes, which also originates from the mountainous interior of the island (Table 2.11). A flat coastal plain such as the one in JB, with a very small portion steep slopes could influence runoff rates, sediment transport and sediment delivery. This flat terrain may serve as a major storage for sediments eroded from upland areas. The extensive coastal plains region, in JB watershed can limit the transport of sediments through the watershed potentially reducing sediment delivery and deposition to adjacent stream networks and estuary. In the event that is evidence of enhanced sedimentation within the Bay, it may be assumed that adjacent land area is acting as a possible source of sediments to streams and the estuary. In this regard, sheet erosion may be the dominant mechanism by which soil is eroded and transported from one zone to the next.

**Table 2.10.** Slope distribution in the RES watershed.

<table>
<thead>
<tr>
<th>Percent Slope</th>
<th>Area (km²)</th>
<th>Percent Coverage</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-9</td>
<td>18.19</td>
<td>24.2</td>
</tr>
<tr>
<td>9-21</td>
<td>21.43</td>
<td>28.5</td>
</tr>
<tr>
<td>21-33</td>
<td>21.32</td>
<td>28.4</td>
</tr>
<tr>
<td>33-49</td>
<td>11.17</td>
<td>14.9</td>
</tr>
<tr>
<td>49-147</td>
<td>3.07</td>
<td>4.1</td>
</tr>
</tbody>
</table>
Figure 2.8. Slope map for RES watershed using 3D visualization

Table 2.11. Slope distribution in the JB watershed.

<table>
<thead>
<tr>
<th>Percent Slope</th>
<th>Area (km$^2$)</th>
<th>Percent Coverage (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-7</td>
<td>100.54</td>
<td>60.30</td>
</tr>
<tr>
<td>7-20</td>
<td>24.67</td>
<td>14.80</td>
</tr>
<tr>
<td>20-34</td>
<td>20.34</td>
<td>12.20</td>
</tr>
<tr>
<td>34-50</td>
<td>15.56</td>
<td>9.30</td>
</tr>
<tr>
<td>50-118</td>
<td>5.68</td>
<td>3.40</td>
</tr>
</tbody>
</table>
4.7.2. Aspect

Aspect indicates the direction of slope. General aspect direction ranges from 0-360 degrees. Aspect maps generated with ArcGIS overlaid with stream network with areas shows the direction water flow is convergence in the landscape (Figures 2.10 and 2.11). Note that stream networks in RES watershed drains into the Atlantic Ocean, which is towards the northern coastline however, for the purpose of display, the watershed has been rotated. The orientation of the JB watershed has not been changed as streams in this area drain to the south towards the bay. Aspect has a significant influence on the microclimate of an area. As mentioned previously, RES has a greater abundance of stream networks when compared to the fact that this watershed is located on the windward side of the island, while JB is located on the leeward side. With reference to RES, most slopes are in a northern direction. Slope direction coupled with the prevailing
north-eastern trade winds has a marked effect on precipitation patterns, with increased rainfall at higher elevation. The increase rainfall in the headwaters may be a major driving force for erosion in the watershed however, due to dense forest cover at higher elevations; the soil loss potential is reduced. The dense tropical canopy may provide a protective covering, reducing the impact of rain drops on the underlying soils. In addition, the reduced impact of rainfall due to canopy cover particularly in the headwaters may limit runoff in this watershed. Runoff depth and velocity has a major influence on soil detachment and sediment transport. JB has a drier climate when compared to RES, which is the rain shadow effect created by the central mountain range on the island. The lower amounts of rainfall received in this soil, my limit the amount of runoff generated in this watershed. The lack of runoff in this watershed reduces the ability for sediment to be detached and transported from point of origin thus serving as a potential explanation for the low soil loss potential predicted using RUSLE.
Figure 2.10. Aspect map for RES watershed illustrating slope direction using 3D visualization.

Figure 2.11. Aspect map for JB watershed illustrating slope direction using 3D visualization.
4.7.3. Planform curvature

Planform curvature as defined earlier, highlights areas of flow convergence and divergence. This terrain attribute assisted with delineating drainage network in watersheds; figures 2.12 and 2.13 shows the planform curvature for both study watersheds. Based on simple spatial comparison of statistics generated for convergence and divergence of water flow, approximately 46% of RES watershed exhibit divergent water flow compared to the 28% estimated for JB (Table 2.12). With regards to flow convergence (areas of concavity-planform curvature), RES had a higher spatial distribution of this landscape feature than JB. The higher occurrence of convergent features in the landscape corresponds to the fact that the northern side of the island has more rivers than the southern coastline due to the greater amounts of rainfall received in that area (Field, 2003). As demonstrated by slope analysis, JB has an extensive coastal plains area. This dominant “shape” for this watershed may increase the depositional or storage capacity for the watershed possibly reducing the ability of sediment transfer through the system once deposited in this floodplain. Consequence, increased aggradations’. The greater proportion of divergence flow (areas of convex-planform curvature) in the RES watershed in indicates the amount of ridges in the landscape. This distribution of this feature in the landscape provides insight into the complexity of terrain.

Table 2.12. Spatial distribution of divergent and convergent of water flow in study watersheds.

<table>
<thead>
<tr>
<th>Planform Curvature</th>
<th>Spatial Coverage (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Convex (Divergent Flow)</td>
<td></td>
</tr>
<tr>
<td>RES</td>
<td>45.8</td>
</tr>
<tr>
<td>JB</td>
<td>28.2</td>
</tr>
<tr>
<td>Flat</td>
<td>39.8</td>
</tr>
<tr>
<td>Concave (Convergent Flow)</td>
<td>14.4</td>
</tr>
<tr>
<td></td>
<td>6.2</td>
</tr>
</tbody>
</table>
Figure 2.12. Planform curvatures overlain with stream networks for RES. Mapped using 3D visualization.

Figure 2.13. Planform curvatures overlain with stream networks for JB. Mapped using 3D visualization.
4.7.4. Profile curvature

Climate, topography, soil type, vegetation and land conservation practices are key inputs into the RUSLE model. These factors interact in complex ways in order to generate a response variable, which in this model is A- tonne (ha yr)$^{-1}$. Topographically, RES watershed can be considered complex rather than simple. A map of profile complexity demonstrated the level of complexity in this watershed (Figure 2.14). The dominant shape of this watershed is concave slopes (Table 2.13). With reference to profile curvature, concave slopes denote deceleration of water flow. Inferences that can be made here include: (1) reduced flow velocity, (2) decreased runoff potential and consequently decreased soil erosion/ sediment transport potential in this watershed. In addition, deceleration of water flow can be suggestive of flow convergence. Since profile curvature can be used as a proxy for identifying potential erosional and depositional areas in the landscape. One can assumed based on profile curvature, that there is dominance of depositional areas in the RES watershed as compared to erosional area. This may be contrary to what is expected for this type of landscape given the steepness of watershed however, the headwaters of RES is completely covered by tropical forests. As demonstrated by other studies, dense forest cover tend to reduce soil erosion potential by increasing infiltration rates, reducing runoff and protecting soils from erosive impacts of raindrops (Pimentel and Kounang, 1998). JB besides having a predominantly flat terrain, this watershed had a relatively high slope concavity, which alludes to depositional areas when evaluating profile curvature. Slope convexity in this landscape is relatively low suggesting a low topographic potential for soil erosion (Table 2.13, Figure 2.15).
Table 2.13. Spatial distribution of acceleration and deceleration of water flow in study watersheds.

<table>
<thead>
<tr>
<th>Profile Curvature</th>
<th>Spatial Coverage (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>RES</td>
</tr>
<tr>
<td>Convex (Divergent Flow)</td>
<td>12.1</td>
</tr>
<tr>
<td>Flat</td>
<td>42.3</td>
</tr>
<tr>
<td>Concave (Convergent Flow)</td>
<td>45.6</td>
</tr>
</tbody>
</table>

Figure 2.14. Profile curvature map for RES watershed. Whiter areas indicate flow deceleration and dark areas represents flow acceleration. Mapped using 3D visualization.
Coincidence analysis (cell by cell) between RUSLE and predicted erosion and deposition from profile curvature indices was conducted. It was determined that approximately 46% of area with an estimated soil loss of 1-2 tonne (ha yr)$^{-1}$ corresponding with predicted depositional areas (Table 2.14). Approximately 60% of the JB area that experience soil loss of less than 1 tonne (ha yr)$^{-1}$ coincided with flat or floodplain areas (Table 2.15). Less than 5% of the estimated soil loss potential probably originated from erosional areas. Based on the coincidence tables (Tables 2.14 and 2.15) presented below, it can be assumed that depositional areas and flat areas may experience a greater soil loss potential than predicted erosional areas. The implication of such results is that sheet erosion via surface overland flow may be a major process by which soil is erosion from the surface as detached soils are removed in a thin uniform layer across the surface.
Table 2.14. Occurrence of RUSLE values with erosional and depositional areas in percentage for RES watershed.

<table>
<thead>
<tr>
<th>RUSLE (tonne/ha/yr)</th>
<th>Less than 1</th>
<th>1-2</th>
<th>2-3</th>
<th>3-4</th>
<th>4-5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Erosion</td>
<td>0.0</td>
<td>11.8</td>
<td>0.1</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Flat</td>
<td>5.8</td>
<td>36.0</td>
<td>0.2</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Deposition</td>
<td>0.3</td>
<td>45.6</td>
<td>0.3</td>
<td>0.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>

Table 2.15. Occurrence of RUSLE values with erosional and depositional areas in percentage for JB watershed.

<table>
<thead>
<tr>
<th>RUSLE (tonne/ha/yr)</th>
<th>Less than 1</th>
<th>1-2</th>
<th>2-3</th>
<th>3-4</th>
<th>4-5</th>
</tr>
</thead>
<tbody>
<tr>
<td>Erosion</td>
<td>4.6</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Flat</td>
<td>59.8</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Deposition</td>
<td>35.4</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
</tbody>
</table>

4.8. Soil Characteristics

Soil texture is an important factor influencing erosion and soil hydrology. Grain-size determines the susceptibility of particles to removal from overland flow. It also determines infiltration rates and hydraulic conductivity of soils. In both watersheds, soils are mainly of silty clay loam texture (Boccheciamp, 1977). Maps showing the hydrologic properties of soil in the study watersheds are mapped below.

4.8.1. Geology and soil types

There are eight known geologic units and 28 different soil types in RES (Figures 2.16 and 2.17). The dominant geologic unit is the Hato Puerco Formation, which is located primarily in the headwaters of the study area. This formation is marine deposited calcareous mudstone. Age of these rocks is approximate 97 to 92 million years (Huffacker, 2002). With regards to soil type, approximately 40% of this study area is comprised of the Los Guineous–Yunque-Stony rock land association. Fifty percent of this association is composed of Los Guineous soils, 31% is Yunque soils, and about 10% is stony rock land (Boccheciamp, 1977). This association is found on steep slopes, but
are covered by hardwood rainforests which may stabilize soils, reducing their susceptibility to slippage and possibly erosion (Bocchecciamp, 1977). In addition, both Los Guineous and Yunque soils belong to soil order Ultisols, specifically from the humults suborder. The humult suborder is characterized as being well drained soils with high organic matter. A high organic content enhance aggregate formation and stability, which may lead to reduced soil erosion.

Figure 2.16. Geological mapping units for RES watershed.
JB has 12 known geologic mapping units and 29 different soil types (Figures 2.18-2.19). Within this watershed, alluvium is the most prevalent geologic unit and is primarily located in the coastal flood plains area. Descalabrado-Rock land complex makes up 13% of this area being the most dominant soil type in the JB watershed. This complex can be found on steep slopes and on ridge top. It experiences medium to rapid runoff and is vulnerable to soil erosion. Soils in the Descalabrado series tend to be shallow to rock, approximately 1.5 ft to bedrock (Boccheciamp, 1977). Rainfall intercepting shallow soils can cause saturated zones stimulating saturated overland flow. Saturated overland flow may be a major mechanism by which soil may be eroded and transported to other area in the watershed or areas of converging flow.
Figure 2.18. Geologic mapping units for JB watershed.

Figure 2.19. Soil mapping units for JB watershed.
4.8.2. Soil hydrologic properties

Majority of the RES watershed have a N/A classification for runoff based on the soil survey however, 28% of the area experienced rapid runoff (Figure 2.20). Over 50% of the soils within study area had belonged to hydrologic group A (Figure 2.21). This group is typically characterized as having a high infiltration rate soil is deep and well drained to excessively drained sands and or gravel. Conversely 36% of the watershed has soils that have very slow infiltration (Figure 2.21). These areas of slow infiltration correspond to areas where runoff is classified as rapid, which is located along the western boundary of the watershed. Figure 2.22 show spatial distribution of average saturated hydraulic conductivity (ksat). A large portion of this watershed RES exhibits a low ksat (0.9 – 2.7 micrometers/second). Given that only a small percent of the study area is characterized by rapid runoff, the soils in this areas is protected by vegetation therefore, the potential soil loss is minimal.
Figure 2.20. Spatial representation of runoff for RES watershed based on soil survey information.

Figure 2.21. Spatial representation of soil hydrologic group for RES based on soil survey information.
A visual inspection of JB indicates that a large portion of this watershed exhibited an average ksat between 2.7-21.2 micrometers per second (Figure 2.23). With regards to hydrologic group, approximately 65% of the study area has a classification of D, suggesting that soils have a very slow infiltration rates (Figure 2.24). Similar to RES, JB had a large portion of its area (~38%) non classifiable for runoff however, most other soils fell within the category of experiencing medium runoff (Figure 2.25). Based on the distribution of the physical properties above, it may be assumed that surface overland flow may be visible during a storm event. The soils in this watershed are relatively shallow, ksat is relatively high and infiltration rates are slow. All of these factors combined may create a situation of rapid runoff, which can potentially detached and remove soil from the land surface.
Figure 2.23. Spatial representation of average hydraulic conductivity for JB based on soil survey information.

Figure 2.24. Spatial representation of soil hydrologic group for JB based on soil survey information.
Figure 2.25. Spatial representation of runoff for JB watershed based on soil survey information.

5.0. Summary

The RES watershed is primarily a riverine environment with a riverine/estuarine ecosystem at the mouth of the river. Sediment, a product of soil erosion may have the potential to cause temporal changes in channel morphology (depending on residence time of river). Increase sediment input to the river can influence aquatic habitat within the river and smaller tributaries in the upper portions of the watershed. Further downstream, at the riverine/estuarine interface, increase sediment input from the watershed has the potential to increase turbidity thus influencing primary productivity. Figure 2.26 shows the sediment plume originating from the mouth of the RES watersheds.

Fryirs and others (2007) describe the different types of landscape (dis)connectivity. Authors identified buffers, barriers and blankets as features that may influence the transfer of energy and material between landscape compartments or
between systems (e.g. uplands and estuaries) (Fryirs et al., 2007). Buffers are defined as landscape features that prevent sediment from entering the stream channel. Barriers affects sediment transport in stream network and blankets tend to impede surface and subsurface interactions (Fryirs et al., 2007). Based on these definitions and the complexity of the watershed, it may be assumed that the dominant landforms influencing connectivity in the landscape may be presences of buffers and blankets. The local terrain has potentially created a transport-limited and supply limited system. In a transport-limited system, sediment is available for detachment however, the mechanism by which this material may be moved to another compartment in the environment is hindered. In RES, the dominance of depositional area in the landscape, may serve as barriers to sediment transport. On the other hand, this area can also be considered supply-limited in many parts of the watershed soil is not available for detachment or transport as it is protected by vegetation. In event that such areas are converted to another type of land use such as agriculture or pasture, this watershed may no longer be considered supply-limited as soils may be vulnerable to detachment and eventually transport by surface overland flow. These factors working together may influence soil loss potential in RES however, additional data such aerial photos is necessary in order to identify these dis-connective features in the landscape. The occurrence of the features in the landscape can have major implication for sediment delivery processes.
In contrast, the JB watershed lacks major rivers due to the lower amount of rainfall received by the southern coast of the island due to the rain shadow effect from the central mountain range (Field, 2003). Rio Seco, in addition to the smaller channels and man-made canals may be major conduits by which sediment is transported from the upper watersheds to the Jobos Bay estuary. Sediment entering this ecosystem may be trapped by mangrove forests established along a portion of the southern edge of the watershed however, excess sediments have the potential to interrupt gas exchange in the roots, which may cause death to mangroves (Figure 2.27). In the case of JB, the dominant landforms affecting connectivity in this watershed may barriers and blankets. Potential barriers may be natural and anthropogenic. Natural barriers may include relief where as human alteration to stream network (construction of canals) (Zitello et al.,
The coastal plains of JB may produce a blanket effect as it may reduce the potential for soil to be eroded and for deposition in bay. RUSLE estimates soil loss potential via rill and sheet erosion. In the coastal plain region, one can assume that the dominant process may be sheet erosion. Sheet erosion removes a uniform thin layer of soil over the surface whereas rill erosion is channelized soil removal.

![Figure 2.27. Aerial photography of sediment distribution within the Jobos Bay Estuary.](image)

Watershed morphology and soil characteristics play a critical role in the soil erosion process as they potentially influence sediment transport capacity. Results from RUSLE model is affected by uncertainties in the data and in the computation of each of the input. Therefore, results of this model are dependent on the accuracy of inputs. Despite these uncertainties, this study is preliminary assessment of soil loss potential in two contrasting watersheds on the island of Puerto Rico. In light of the lack of field
validation this preliminary assessment can be used to as a baseline for more detailed study on watershed geomorphology and soil loss potential in these watersheds using a geospatial technology. This study is an initial effort to develop and integrative research framework linking soil erosion to sedimentation rates in estuaries.
CHAPTER 3
SEDIMENT DYNAMICS IN A TROPICAL RIVERINE-ESTUARINE SYSTEM

1.0. Introduction
Estuaries are unique and dynamic environments that are located at the interface of watersheds and the ocean. These transitional ecosystems are inhabited by various organisms such as bivalves, worms and crustaceans, which serve to influence the biogeochemical and ecological processes of estuaries (Thrush et al., 2004). In addition, estuaries are highly productive fisheries, prime locations for sheltered ports and major areas of coastal development (Dyer, 1979). As a result, estuaries are one of the most exploited and impacted ecosystems in the world, as they are connected with the human activities within the landscape and at the coast (Blaber, 2002). Pollution from land-based sources is a major concern for coastal managers. Specifically, increased nutrient inputs, industrial effluent and sediment can degrade estuarine ecosystem functions (Thrush et al., 2003). Mai and others (2002) measured chlorinated and poly aromatic hydrocarbons (PAH) in riverine and estuarine sediments in the Pearl River Delta, China. Authors reported that the concentrations for chlorinated hydrocarbons and PAHs may cause biological impairments in organisms inhabiting these ecosystems. While in some cases, the direct source of pollution to a system can be identified, determining a single source of pollution is often difficult (Basnyat et al., 1999; Cutter, 1989; Cebron et al., 2003). The term non-point source pollution (NPS) is utilized for instances in which a discrete pollutant’s source cannot be identified. A ubiquitous NPS is sediment.

Sediment input to coastal environments is a natural geologic process however, changes in the landscape has increased sediment supply to coastal ecosystems such as
Syvitski and others (2005) assessed the impact of anthropogenic activities on sediment flux to oceans. They mentioned that rivers in Africa and Asia tend to exhibit reduced sediment delivery to oceans due to increased retention in reservoirs whereas Indonesian rivers tend to have a greater sediment load to the coastal ocean (Syvitski et al., 2005). Milliman and Syvitski (1992) evaluated the processes controlling sediment delivery from small mountainous watersheds. They suggested that small mountainous streams may have a greater potential to discharge their sediment load to oceans when compared to larger rivers (Milliman and Syvitski, 1992).

Phillips and Slattery (2007) acknowledge the complexities of sediment delivery to the ocean via fluvial processes. These authors describe the seaward trend in slope, discharge and stream power in a lower coastal plain river. In this study, Phillip and Slattery (2007) observed that discharge may decrease due to in-stream and transitional depression in the coastal reaches. Slope and stream power decreased in lower reaches of the river in the fluvial channel and at the fluvial-estuarine transition possibly reducing sediment delivery to coastal zone (Phillips and Slattery, 2007). These findings suggest increased deposition of sediment material (particularly fines) in estuaries due to decrease in flow, slope and energy. Phillip (1991) developed a sediment budget for the Pee-Dee River basin and Waccamaw River and Winyah Bay estuary. It was estimated that only about 4% of the soil eroded from the landscape reaches the estuary (Phillip, 1991). This study highlights the complex nature of linking soil erosion from upland areas, sediment transport through landscape and downstream to sediment deposition in estuaries.

Sediment deposited in estuarine areas may serve various functions such as the development of mangrove forests along sub-tropical and tropical coastlines (Alongi et al.,
2005). In addition, sediment deposits in estuaries can provide a historical record of local environmental changes (Valette-Silver, 1993; Phillip and Slattery, 2006; Hubeny et al., 2009). A common method of measuring depositional rates within estuaries and other ecosystems involves the use of radionuclides. A variety of radionuclides have been used to measure sediment accumulation. Examples of these include: $^{234}$Th and $^7$Be, which have half lives on the order of 24 and 53 days, respectively (Thomas and Ridd, 2004). Besides those radioisotopes listed above, $^{137}$Cs (half-life = 30 years) and $^{210}$Pb (half-life = 22 years) are radionuclides that been extensively used to evaluate sediment accumulation rates and depositional patterns in estuaries (Pfitzner et al., 2004; Corbett et al., 2007).

The determination of sediment accumulation and deposition rates is critical to assessing estuarine ecosystem health. Enhanced sedimentation rates in transitional marine environments continue to be a major concern for coastal managers. Erosion/sedimentation continue to be a major environmental problem in the Caribbean (Kjerfve et al., 2002). The main purpose of this study is to describe modern sediment dynamics and sedimentation patterns in a tropical riverine-estuarine system in Caribbean. The specific objectives are:

1. to determine the sedimentological characteristics of cores collected from study area
2. to describe the vertical and spatial distribution of $^{137}$Cs and $^{210}$Pb in sediment cores
3. to conduct radionuclide inventories for $^{137}$Cs and Excess $^{210}$Pb
4. to determine statistical relationships between $^{137}$Cs and selected sediment characteristics and the effects of location on the properties of sediment cores
2.0. Study Area

The Rio Espíritu Santo (RES) watershed and a smaller associated watershed along the coast cover an area of approximately 75 km$^2$ (Figure 3.1) and is located on the north-eastern part of Puerto Rico. Headwaters of RES are located in the Caribbean National Forest (CNF). CNF is the only tropical rainforest within the United States Forest Service system. The total forest area measures 112.68 km$^2$ (Huffacker, 2002). Elevation within CNF includes areas that are classified as steep to very steep (Huffacker, 2002). Climate in this area is described as tropical marine with prevailing north-eastern trade winds (Boccheciamp, 1977). Over 60% of the watershed is covered by tropical evergreen forests with a small portion used for agriculture (0.20%) (see chapter 2 for more details). There are 9 geologic mapping units and approximately 28 different soil types in the watershed.

![Figure 3.1. Location of RES watershed on the island of Puerto Rico.](image)

The RES estuary is classified as a river-estuarine system (Smith et al., 2008). The lower reaches of the estuary are mangrove forests, while pastoral lands are in the upper
boundaries of estuary. This estuary is relatively small, with widths ranging from 12 to 55 m. Landward limit of estuary extends no more than 7 km upstream from estuarine mouth (Smith et al., 2008). Average depths in estuary are approximately 1 to 6 m (Smith et al., 2008). RES is strongly stratified hence it may be classified as a salt wedge. Tidal range in RES estuary is less than one meter as a result this system may exhibit general characteristics of a micro-tidal estuarine system (Smith et al., 2008). A low-head dam (also known as a weir is used to raise the level of a river or stream) was built in the upper reaches of the watershed approximately 5 km from the coast. This dam was constructed in 1984 by the Puerto Rico Aqueduct and Sewage Authority (Benstead et al., 1999). Annual freshwater flow to the estuary is approximately 1.68 m³s⁻¹ (Smith et al., 2008).

3.0. Methods

3.1. Sampling and Sample Preparation

Four push cores of varying lengths were collected from RES riverine-estuarine system (Figure 3.2). Core locations and general characteristics are presented in table 3.1. Sediment cores were collected in exposed banks of the major channel and on the edge of adjacent mangrove forests. Cores were extruded and sectioned in 0.5 cm intervals for the first 10 cm and 1 cm intervals from 10 cm to base of cores. Extrusion of cores were completed at the study site, afterwards the samples were frozen and shipped to the University Of South Florida College Of Marine Science, Aquatic Radiogeochemistry Laboratory for further processing. Upon reaching the laboratory, samples were weighed and one-half of each sample was archived. The remaining portion of each sample was freeze-dried prior to undergoing geochemical and sedimentological analyses. Soil samples were also collected on land from the headwaters of RES (Figure 3.2).
Figure 3.2. Sampling location of sediment cores and soil samples in RES watershed. Note that some of the cores are very close together hence the overlay in soil samples.

Table 3.1. Sediment core locations, length and date of collection.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Sample Type</th>
<th>Core Length (cm)</th>
<th>Date Collected</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESC1</td>
<td>18.37364 N</td>
<td>65.81887 W</td>
<td>Core</td>
<td>52</td>
<td>July, 2007</td>
</tr>
<tr>
<td>ESC2</td>
<td>18.40305 N</td>
<td>65.81410 W</td>
<td>Core</td>
<td>66</td>
<td>August, 2008</td>
</tr>
<tr>
<td>ESC3</td>
<td>18.38954 N</td>
<td>65.81659 W</td>
<td>Core</td>
<td>42</td>
<td>August, 2008</td>
</tr>
<tr>
<td>CCB4</td>
<td>18.40921 N</td>
<td>65.80476 W</td>
<td>Core</td>
<td>70</td>
<td>July, 2007</td>
</tr>
<tr>
<td>RES1</td>
<td>18.31972 N</td>
<td>65.82495 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>RES2</td>
<td>18.31981 N</td>
<td>65.82486 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>EV1</td>
<td>17.93658 N</td>
<td>66.22481 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>EV2</td>
<td>17.93658 N</td>
<td>66.22481 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>EV3</td>
<td>17.93658 N</td>
<td>66.22481 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>EV4</td>
<td>18.34636 N</td>
<td>65.82404 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>EV5</td>
<td>18.34644 N</td>
<td>65.82404 W</td>
<td>Soil</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
</tbody>
</table>

3.2. Sedimentological Analysis

3.2.1. Sediment distribution

Grain-size distribution was determined for both sediment and soil samples. The traditional wet-sieving method and Micromeritics Saturn DigiSizer ® 5200 (a high
resolution laser particle size analyzer) was used for this analysis. To perform this analysis, approximately three grams of freeze-dried sample was first dispersed using approximately 10 ml of a 5% solution of Sodium Tripolyphosphate (calgon) solution. The dispersed sediment and soil samples were then wet-sieved using a 63 µm mesh sieve. This procedure separated larger fraction and sand-sized (LFS) constituents from finer material (particles \( \leq 63 \) µm). Particles larger than 63 µm were collected from sieved, oven-dried and weighed. Finer material were analyzed further using the Saturn DigiSizer to determine percent silt-sized particles (4 µm < silt \( \leq 63 \) µm) and clay-sized particles (\( \leq 4 \) µm).

3.2.2. Loss on ignition (LOI)

Organic matter and carbonate content was determined from a sequential ignition methodology (Dean et al., 1974 and Heiri et al., 2001). Sediment and soil samples were finely grounded and homogenized by stirring. Powered samples were then oven dried for 12-24 hours at 105 °C in crucibles in order to standardize the sample weights and to extract any moisture in samples. Samples were cooled in dessicators and sample weights were obtained. These samples were then placed in a furnace for 4 hours at 550 °C. Weight loss after 550 °C was used to compute percent organic matter (O.M.). Carbonate was determined from weight loss after ignition at 950 °C for 1.5 hrs using samples that had previously undergone the 550 °C treatment (see Appendix A for detailed LOI procedures).

3.3. Gamma Analysis

3.3.1. \(^{137}\)Cs determination

Sediment and soil samples were prepared for gamma analysis using procedures described in Johnson-Pyrtle and Scott (2001). Approximately, one to two grams of
samples were sealed in plastic test tubes and were assayed for gamma emitters using two Canberra® high purity germanium well detectors connected to a Canberra Genie multi-channel analyzer, which records the gamma spectra in 4096 channels. These detectors were calibrated using U.S. National Institute of Standards and Technology 4357 Ocean Sediment multiline and Canberra Industries MGS-5 sediment standards. \(^{137}\)Cs was identified a specific energy peak at 661 keV in every other sample (every 1 cm in top 10 cm and every 2 cm below until depth of 40 cm). A Peruvian soil standard, Columbia River sediment river sediment standard, an Ocean sediment standard prepared by NIST was used to verify detector performance. Since it is believed that \(^{137}\)Cs has a strong affinity to clays, resulting values for samples were normalized against clay. Total \(^{137}\)Cs inventories were calculated for each core using the following equation:

\[
I = k \sum \rho_i t_i C_i
\]

**Equation 3.1**

Where \(I\) = the \(^{137}\)Cs inventory (Bq/m\(^2\)) for each sediment core, 
\(C_i = \) \(^{137}\)Cs (Bq/g) measured in each increment 
\(\rho_i = \) the density for each sediment increment (g/cm\(^2\)) 
\(t_i = \) the thickness of each increment (cm) 
\(k = 10,000\), a constant for converting Bq/cm\(^2\) to Bq/m\(^2\).

3.3.2. Excess \(^{210}\)Pb determination

Both well detectors and a planar detector were used to measure \(^{210}\)Pb activity concentrations in sediment and soil samples. Sealed plastic vials containing 1-2 grams of sample (well detectors) and aluminum tin cans (planar detector) containing approximately five grams of sediments were allowed reach secular equilibrium during a period of no less than 30 days prior to analysis. This time period is assumed to be the equivalent of 8 half lives of \(^{222}\)Rn, the immediate daughter of \(^{226}\)Ra. \(^{214}\)Pb is a daughter product of \(^{222}\)Rn and a precursor to \(^{210}\)Pb. The activity of \(^{226}\)Ra can be determined by
determining the activity of $^{214}\text{Pb}$. Samples were analyzed for gamma emitters mentioned above for at least 24 hrs. $^{210}\text{Pb}$ and $^{214}\text{Pb}$ were identified by specific energy peaks of 46 keV and 351 keV, respectively. Energy and efficiency performance were calculated and factored prior to determining the actual activities (see Appendix D and E). Excess $^{210}\text{Pb}$ (also known as unsupported $^{210}\text{Pb}$) can be estimated by subtracting $^{214}\text{Pb}$ ($^{226}\text{Ra}$) activity from the total $^{210}\text{Pb}$ activity. Gamma activities were also normalized with clay (Aalto et al., 2003).

3.4. Statistical Analysis
Statistical analysis was performed using SAS Enterprise Guide 4.2. Regression analyses were performed to assess relationship between $^{137}\text{Cs}$ radionuclide activity concentrations and sediment characteristics (clay, silt, mud fraction (clay + silt) and organic matter). A One-way ANOVA was performed to assess the influence of location on the distribution of $^{137}\text{Cs}$ in RES riverine-estuarine system.

4.0. Results and Discussion
Grain-size, organic matter and calcium carbonate content varied among cores. Such variability may be attributed to core sampling location and dominant environmental processes at each site. The average distance between cores is estimated at 2,253 m. The longest distance occurred between ESC1 and CCB4 in estuary (4,184 m). ESC1 being the most upstream core and CCB4 located within the estuary. The shortest distance was between sediment cores ESC2 and ESC3 at an estimated distance of 805 m (0.5 miles).
4.1. Sediment Composition

4.1.1. Sediment cores

Table 3.2 shows the mean percent clay, silt and LFS in all sediment cores collected from RES estuary. ESC1 and ESC3 are sediment cores collected inland and closer to head of estuary whereas ESC2 and CCB4 were collected near the mouth of the estuary (Figure 3.2). In general, cores taken landwards (ESC1 and ESC3) on average had a greater distribution of clay and silt-sized particles when compared to ESC2 and CCB4 (Table 3.2). Since these sediment cores were taken higher upstream, it may be assumed that the major input of sediments to these areas is land-based. Conversely, LFS and silt-size particles were the dominant in these sediment types (grain-size) in ESC2 and CCB4 cores. The higher amounts of LFS in these cores may be indicative of marine inputs as these cores are located closer to the mouth of the estuary. Specifically, there is a neighboring beach area that may supply sand-sized particles to these sites. The textural patterns observed for ESC2 and CCB4 may suggest that these sites are high energy depositional environments.

<table>
<thead>
<tr>
<th></th>
<th>ESC1</th>
<th>ESC3</th>
<th>ESC2</th>
<th>CCB4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Clay (%)</td>
<td>21.7</td>
<td>18.9</td>
<td>8.7</td>
<td>15.8</td>
</tr>
<tr>
<td>Silt (%)</td>
<td>51.1</td>
<td>61.0</td>
<td>49.7</td>
<td>48.7</td>
</tr>
<tr>
<td>Larger Fraction and Sand (%)</td>
<td>27.1</td>
<td>20.1</td>
<td>41.6</td>
<td>35.8</td>
</tr>
</tbody>
</table>

Evaluation of grain-size distribution down core profiles indicated some spatial patterns between core collection sites. Silt-sized particles are the dominant sediment type in ESC1 core. On average over 51% of the sediments core is comprised of silt-sized particles (Table 3.2). With the exception of the first 1.5 cm, and a poorly sorted layer
that is approximately 1-2 cm thick below surface layer, silt distribution remains relatively constant with depth in ESC1 core (Figure 3.3). The sediment characteristics of ESC3 are similar to ESC1 in that silt-sized particles also dominate this sediment core. From 1.5 cm depth in core to about 4 cm depth, there is a poorly sorted segment in core that is mainly comprised in silt-sized particles and LFS. This layer is approximately 3 cm thick (Figure 3.4). The sampling locations for ESC1 and ESC3 are spatially close to each other hence, it may be expected that they would exhibit similar sediment depositional patterns. In these two cores, inconsistency in energy may be observed. At the surface of each cores, there is some indication of a high energy event however, the dominance of silt-sized particles in both cores may suggest lower energies may have prevailed.

![Grain-size distribution for ESC1 core.](image)

**Figure 3.3.** Grain-size distribution for ESC1 core.
CCB4 is the most seaward core collected within the study river estuarine system. Very distinct segments can be identified in this core (Figure 3.5). Silt-size particles dominate the surface layers of this core. This section concludes at a depth of 5.5 cm. Beneath this section, there is a 9 cm thick sandy layer, which may be indicative of marine input from the local beach and marine source area. This layer may also be a result of higher energy flow from the river during a flooding event, which may have carried finer materials out to sea. Below this segment silt becomes the predominant sediment type, possible low energy environment, which may allow for the deposition of these size particles. ESC2 was also collected close to the mouth of the estuary. This core exhibits an inverse pattern to the sediment distribution of CCB4. There is a silt segment in this core that is not thick or appears at the same depth of the sandy segment, which is visible
in the CCB4, but it does have an alternate pattern of LFS-silt-LFS. This pulse of silt-sized particles may also be representative of a potential pulse of terrestrial derived sediments moved downstream during a flooding even in the river.

Figure 3.5. Grain-size distribution for CCB4 core.
4.1.2. Soil samples

The soil samples below are not organized in any particular order. RES1 and RES2 were collected along the main tributary of river (Espiritu Santo) whereas EV1-EV5 was collected along a smaller, but significant tributary draining into Espiritu Santo.

Based on the graph provided below there are no obvious trends in grain-size distribution between the soil samples (Figure 3.7). What this graph shows is that LFS is the dominant sediment type in some of the soil samples.
4.2. Organic Matter and Carbonate Content

Organic matter (O.M.) in the sediment and soil samples is primarily comprised of partially decayed leaves and woody debris. In certain cores, ESC2 for example, very fibrous material, which has been identified as root material from mangroves along the shoreline of the riverine-estuarine channel, are present. The identifiable fragments of carbonates in sediments were shells. These pieces were too small to allow for identification of organisms associated with the carbonate fragments.

4.2.1 Sediment cores

In the CCB4 core both O.M. and carbonate content is highly variable with depth (Figures 3.8-3.9). Within the ESC2 core, both O.M. and carbonates increased in depth (Figures 3.10-3.11). ESC2 on average had the highest O.M. content when compared to other cores (Table 3.3). The ESC2 core was collected adjacent to mangroves forest that

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**Figure 3.7.** Sediment distribution in soil samples collected on land in the headwaters of RES riverine-estuarine system.
lined the shores of the estuarine system, which may provide an explanation for high amounts of O.M. estimated for this core. The carbonate content was also higher in this core as compared to the other cores. This may be due to a higher occurrence of shelled organisms living in the protected mangrove as opposed to other core locations, which in its surrounding areas were urban or other forms of vegetated land cover such as pasture.

Table 3.3. Average O.M. and carbonate content in sediment core.

<table>
<thead>
<tr>
<th>Sample Core</th>
<th>O.M. (%)</th>
<th>Carbonate (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESC1</td>
<td>14.0</td>
<td>2.6</td>
</tr>
<tr>
<td>ESC3</td>
<td>15.8</td>
<td>4.3</td>
</tr>
<tr>
<td>ESC2</td>
<td>48.7</td>
<td>7.8</td>
</tr>
<tr>
<td>CCB4</td>
<td>20.5</td>
<td>5.2</td>
</tr>
</tbody>
</table>

Figure 3.8. Percent O.M. profile for CCB4 sediment core.
Figure 3.9. Percent carbonate profile for CCB4 sediment core.

Figure 3.10. Percent O.M. profile for ESC2 sediment core.
Carbonate content and O.M. content in the ESC3 core exhibits some variability (Figures 3.12-3.13). O.M and carbonate in the ESC1 does not exceed 20% (Figures 3.14-3.15). ESC1 possessed the lowest carbonate when compared to the other cores. This may be attributed to location (core located furthest upstream), where some shelled organisms typically found in estuarine environment may not be found at this location. In addition, this location may not have a significant marine influence even though it is located less than 4,828 m (3 miles) from the mouth of the estuary.
Figure 3.12. Percent O.M. profile for ESC3 sediment core.

Figure 3.13. Percent carbonate profile for ESC3 sediment core.
Figure 3.14. Percent O.M. profile for ESC1 sediment core.

Figure 3.15. Percent carbonate profile for ESC1 sediment core.
4.2.2. Soil samples

Soil samples collected from the RES watershed possessed less than 25 % O.M. With samples such as RES1, EV4 and EV5 containing less than 10 % organic matter (Figure 3.16). In addition, all samples contained less than 3 % carbonates (Figure 3.17). It may be assumed that potential sources of organic matter for these samples are leaf litter and other forms of detritus at these sites. Carbonates sources at these locations may be shells from terrestrial invertebrates.

Figure 3.16. Percent O.M. in soil samples collected from RES watershed.
Figure 3.17. Percent carbonates in soil samples collected from RES watershed.

4.3. Gamma Analysis

$^{137}$Cs is an anthropogenic radionuclide that has been introduced to the environment via controlled discharges from nuclear plants (Avery, 1996). However, the primary source of this radionuclide originates from the testing of nuclear weapons in from 1945 to 1980 (Beck and Bennett, 2002). In addition, the Chernobyl accident in 1986 also released a recognizable amount to radioactive material into the atmosphere. Radioactive debris, which was injected into the atmosphere, is eventually deposited on the earth surface. The primary source of $^{137}$Cs to the Caribbean is global atmospheric fallout (Alonso-Hernandez et al., 2006). Figure 3.18 shows historical records of $^{137}$Cs fallout over the Miami area.
4.3.1. $^{137}$Cs distribution in sediment cores

$^{137}$Cs profiles for this estuary are representative of an unstable environment (Figures 3.19-3.22). As a result, sedimentation rates could not be computed for this study area. On the other hand, these profiles provide valuable insight into radionuclide retention and distribution in the RES river-estuarine system. At various depth intervals within CCB4, ESC2 and ESC3, $^{137}$Cs goes to zero. This may be an indication of “old material” being supplied to this area whereas the presence of $^{137}$Cs indicates “newer” or “younger material” being introduced to the system via storm or flooding events. Another possible explanation for the erratic $^{137}$Cs profiles in sediment cores may be reflective of biological activities (bioturbation). Each core has a relatively high percent O.M. content even at great depths, therefore it is possible that organisms burrow down into the sediments thus altering the $^{137}$Cs profiles for these sediment cores.
Figure 3.19. $^{137}$Cs activity normalized with clay for sediment core ESC1.

Figure 3.20. $^{137}$Cs activity normalized with clay for sediment core ESC3.
Figure 3.21. $^{137}$Cs activity normalized with clay for sediment core ESC2.

Figure 3.22. $^{137}$Cs activity normalized with clay for sediment core CCB4.
$^{137}$Cs distributions for all cores were plotted on a single graph for the sake of comparison (Figure 3.23). Three major events (storm or flood) may have occurred, which probably supplied $^{137}$Cs or newer material to the system (Figure 3.24). The pulses that are visible on the graph suggests to some degree a possible time lag in the delivery as in certain instances the pulse or peaks occurs at a shallower depth for those cores collected at the estuarine mouth (CCB4 and ESC2) and at deeper depths in the more inland cores (ESC1 and ESC3).

Figure 3.23. Down core distribution of $^{137}$Cs activity for all sediment cores.
Figure 3.24. Down core distribution of $^{137}$Cs activity for all sediment cores with possible flooding events highlighted on graph.

4.3.2. $^{137}$Cs inventories for sediment cores

$^{137}$Cs inventories were computed for each core. Due to the fact that $^{137}$Cs was detected at the base of core, the calculate values is not a “true” representation of the total inventory of $^{137}$Cs in these cores. The inventory values reported herein actually represent minimum inventories. Spatially, inventories in general, decrease towards the mouth of the estuary with the exception of ESC2. ESC1, the core collected furthest upstream, had the highest total inventory (7.98 dpm/cm$^2$) of all cores (Table 3.4). This decreasing trend may be explained by observations which suggests that $^{137}$Cs behaves like a conservative element (soluble) in marine environments. However, in freshwater systems this radionuclide may be associated with fine-grained sediments (Murdock et al., 1995; Volpe 2002).
ESC2 had the lowest estimated inventory (1.22 dpm/cm$^2$) of all the cores (Table 3.3). In the ESC2 sediment core, LFS is the dominant grain-size at various intervals (Table 3.2). It is observed that $^{137}$Cs has a strong affinity for fine-grained sediment (particularly clays) in freshwater systems (Francis and Brinkley, 1976). With reference to this core, it may be assumed that this dominance of LFS has reduced the absorption potential of this radionuclide hence its lower inventory when compared to the other sediment cores (Avery, 1996). In addition, ESC2 has on average the highest percentage of O.M. Various studies have reported that increased organic matter in soils can reduce the ability of $^{137}$Cs to be adsorbed onto the mineral fraction of soils (Avery, 1996; Staunton et al., 2002).

ESC2 is the shortest core in length collected from this estuary. To allow for a better comparison without length of core being a confounding factor, inventories were computed using activity concentration up to a depth of 20.5 cm. It was observed that even though inventories were calculated based on a “shallower” depth, ESC2 still possessed the lowest inventories, providing some support to the assumption that O.M. and sediment composition playing a major role in $^{137}$Cs adsorption (Table 3.5). Figure 3.25 show a map with $^{137}$Cs inventories computed for sediment cores in the RES riverine estuarine system.

<table>
<thead>
<tr>
<th>Core ID</th>
<th>Inventory (dpm/cm$^2$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>CCB4</td>
<td>6.86</td>
</tr>
<tr>
<td>ESC2</td>
<td>1.22</td>
</tr>
<tr>
<td>ESC3</td>
<td>7.73</td>
</tr>
<tr>
<td>ESC1</td>
<td>7.98</td>
</tr>
</tbody>
</table>
Table 3.5. $^{137}$Cs inventories computed from a maximum depth of 20.5cm

<table>
<thead>
<tr>
<th>Core ID</th>
<th>Inventory (dpm/cm$^2$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESC1</td>
<td>4.67</td>
</tr>
<tr>
<td>ESC3</td>
<td>3.00</td>
</tr>
<tr>
<td>ESC2</td>
<td>1.22</td>
</tr>
<tr>
<td>CCB4</td>
<td>3.13</td>
</tr>
</tbody>
</table>

Figure 3.25. $^{137}$Cs inventories for all sediment cores in RES riverine-estuarine system.

4.3.3. $^{137}$Cs distribution in soil samples

Gamma analysis was performed on soil samples. The highest level of $^{137}$Cs was detected in soil sample RES1 (4.04 dpm g$^{-1}$ clay) whereas EV1 had the lowest (0.24 dpm g$^{-1}$ clay) (Figure 3.26). The RES soil sample has an abundance of LFS with small amounts of clay and silts. This sample also has the lowest percent O.M (3.2 %), when compared to the other samples. High levels of $^{137}$Cs were also detected in EV2. Both RES1 and EV1 have an abundance of LFS sized particles. Additional testing is necessary
to discover whether these particles may have an influence on $^{137}\text{Cs}$ distribution in these soils. A common trend witnessed for soil samples RES2, EV1 and EV3 is that these samples are comprised of at least 18% O.M. Staunton and others (2002) investigated the potential role of O.M. in radiocaesium adsorption in soils. Authors reported that the type and amount of organic matter may decrease adsorption of this nuclide. In these tropical soils it is unclear whether distribution of $^{137}\text{Cs}$ is affected by the high amounts of O.M. present in these samples.

![Figure 3.26. $^{137}\text{Cs}$ activity concentration in soil samples. Activities normalized with clay.](image)

4.3.4. $^{210}\text{Pb}$ distribution

The major inputs of $^{210}\text{Pb}$ to the water column include: 1) atmospheric fallout from the decay of $^{222}\text{Rn}$ gas, which is primarily released from continents, 2) in-situ decay of $^{226}\text{Ra}$ (via $^{222}\text{Rn}$) in the water column and 3) rivers via weathering of rocks and soils, leaching from soils (Swarzenski et al., 2003). Excess $^{210}\text{Pb}$ has been used extensively as
an environmental tracer. This radionuclide is often used to describe sediment depositional history in various coastal environments (He and Wallings, 1996). Figures 3.27 to 3.30 show the excess $^{210}\text{Pb}$ activities down core for each sediment core. Average excess $^{210}\text{Pb}$ for ESC1, ESC3, ESC2 and CCB4 are 10.4, 5.13, 1.71 and 15.7 dpm g$^{-1}$ clay, respectively. $^{210}\text{Pb}$ activity tends to increase as one move in a seaward direction. Rajashekara et al. (2008) observed a similar trend in sediment samples collected from the Kali and Nethravathi Rivers in Karnataka, India. CCB4, the core located closest to the beach/ocean area has the highest $^{210}\text{Pb}$ activity. A possible explanation for this is that the coastal ocean may be acting as a source of $^{210}\text{Pb}$ at this site. Of the four cores, CCB4 is the only sediment core that goes to zero at depth. It is important to note that this is the longest core collected from this system (Figure 3.30). This observation may suggest that in order to acquire a more complete depositional history for this environment, longer sediment cores should be collected.

The excess $^{210}\text{Pb}$ profile for ESC2 shows an interesting pattern that is unique when compared to other cores (Figure 3.29). At the surface of the core, up to a depth of about 5 cm, $^{210}\text{Pb}$ values goes to zero. This pattern suggests that there were no recent inputs of $^{210}\text{Pb}$ to this site. There are small peaks in activity at various depths however, this may suggest a flux to the site via a flooding event. Another possible explanation for this pattern may be due to removal of sediments from this zone from the estuary via physical processes such as floods, tides and/or currents. When the sedimentological characteristics of this sediment core are taken into consideration, ESC2 has the lowest amount of clay as compared to the other cores. It also has the lowest mud fraction (< 63 µm) of all four cores. Excess $^{210}\text{Pb}$ similar to $^{137}\text{Cs}$ has an affinity to finer particles
therefore, the lower activity of $^{210}\text{Pb}$ may be attributed to the small amounts of finer particles specifically clays in this sediment core (He and Wallings, 1996). Figure 3.31 shows a map of excess $^{210}\text{Pb}$ inventories for sediment cores collected in RES riverine-estuarine system.

Figure 3.27. $^{210}\text{Pb}_{\text{ex}}$ activity normalized with clay for ESC1 sediment core.
Figure 3.28. $^{210}\text{Pb}_{\text{ex}}$ activity normalized with clay for ESC3 sediment core.

Figure 3.29. $^{210}\text{Pb}_{\text{ex}}$ activity normalized with clay for ESC2 sediment core.
Figure 3.30. \( ^{210}\text{Pb}_{\text{ex}} \) activity normalized with clay for CCB4 sediment core.

Figure 3.31. Excess \( ^{210}\text{Pb} \) inventories computed for sediment cores in RES riverine-estuarine system.
4.1.5. $^{210}$Pb distribution in soil samples

As mentioned before, RES1 and RES2 were collected along the main river tributary of river (Espiritu Santo) in the headwaters of the watershed. Comparing these sites, we see that the resulting excess $^{210}$Pb activity in each of these samples is very different with RES having the highest activity (29 dpm g$^{-1}$ clay) (Figure 3.32). No trend in $^{210}$Pb activity was visible when comparing all of the soil samples (Figure 3.32).

![Figure 3.32. $^{210}$Pb$_{ex}$ activity normalized with clay for soil samples collected from RES watershed.](image)
4.4. Statistical Analysis

4.4.1. Potential effect of sediment characteristics on $^{137}$Cs
Regression analyses were performed to evaluate the effect of selected sediment characteristics on $^{137}$Cs distribution for each core. No statistically significant relationships were determined for $^{137}$Cs and any of the sediment characteristics in ESC1, ESC2 and CCB4 used in the analyses (Table 3.6). For sediment core ESC3 statistically significant relationships were detected for all sediment characteristics with the exception of percent O.M. Results from these analyses indicate that sediment type (clay and silt) may have an effect on $^{137}$Cs distribution in the ESC3. Silt-sized particles are the dominant grain-size in this core (61%) and the abundance of this sediment type in ESC3 may have an influence on $^{137}$Cs distribution at this sampling site.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>N</th>
<th>Clay (%)</th>
<th>Silt (%)</th>
<th>Mud (%)</th>
<th>O.M. (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>ESC1</td>
<td>22</td>
<td>0.05</td>
<td>0.004</td>
<td>0.02</td>
<td>0.005</td>
</tr>
<tr>
<td>ESC2</td>
<td>16</td>
<td>0.09</td>
<td>0.38</td>
<td>0.16</td>
<td>0.0</td>
</tr>
<tr>
<td>ESC3</td>
<td>23</td>
<td>0.41*</td>
<td>0.68**</td>
<td>0.65***</td>
<td>0.02</td>
</tr>
<tr>
<td>CCB4</td>
<td>25</td>
<td>0.07</td>
<td>0.01</td>
<td>0.0</td>
<td>0.02</td>
</tr>
</tbody>
</table>

* indicates statistical relationship was significant at $\alpha=0.5$
** indicates statistical relationship was significant at $\alpha=0.001$
*** indicates statistical relationship was significant at $p<0.0001$

4.4.2. Potential effects of location on $^{137}$Cs
A One-way ANOVA was performed to determine the effect of location on $^{137}$Cs distribution. The analysis was significant, F-value (9.67), $p<0.0001$ ($r=0.26$). Results suggest that the location of sediment cores influences the spatial distribution of $^{137}$Cs. This trend is visible when considering the $^{137}$Cs inventories computed for sediment cores where as one moves seawards, the inventories generally decrease. These results may be supported by observations where $^{137}$Cs behaves conservatively in the marine environment.
due to the greater amount of cations competing for space on particulates. In contrast, in freshwater systems there is less cation competition, therefore $^{137}$Cs may be more readily adsorbed and transported via sediments.

4.5. Relating Sediment Characteristics to Soil Properties

A 30 m and 60 m buffer was created to determine soil characteristics and landuse closest to the sampling sites. With reference to sediment core identified as ESC1, the primary soils surrounding this site are Pandura- very stony land complex and Coloso, silty clay loam. These soils are stark contrast from each other. The Pandura mapping unit is found on steep slopes, where soils are shallow and susceptible to erosion. Coloso silty clay loam experience occasional flooding most likely due to slow permeability and has a seasonally high water table. Soil characteristics listed above suggests the potential of this site to be a recipient of sediment material eroded from neighboring areas. In addition, this area is part of a floodplain that may experience seasonal hydric conditions. This is best supported by the fact that upon sampling, a thick buffer of bamboo lined the channel of this river.

ESC3, which is the site located closest to the ESC1, in some respect represent a transitional environment. Typical land use surrounding this core include: pasture, emergent/herbaceous and woody wetlands. Soil mapping units surrounding this core are: tidal swamp material, wet alluvial land and Corcega sandy loam. The Corcega sandy loam soil type is typical on river floodplains and experience periodic flooding, hence the presence of wetland forest within this area. The ESC2 sediment core was collected adjacent to mangrove forest in tidal swamps. CCB4, the seaward core was also taken adjacent to mangrove forests lining the channels of the estuaries. In all sediment cores
there is a dominance of silt-sized particles (Table 3.2). The textural classification for sediment cores is silt loam. This may be attributed to the dominance of this sediment type in soils within this watershed with some implication that the watersheds may be supplying this riverine estuarine environment with sediments. This leads to the question of: *What are potential source areas for these sediments?* While some soil samples were collected in the watershed, the quantity collected is not sufficient enough for speculating potential source areas of sediment in the watershed.

### 5.0. Conclusion

There is great variability in both the $^{137}$Cs and excess $^{210}$Pb activity in all sediment cores. Due to this variability it was not possible to determine sedimentation rates at any of the sampling sites. The sedimentological characteristics, source inputs, hydrological and climatic forcings may help explain the depositional patterns of radionuclides deposited in the sediments of this tropical estuarine system. However, further geochemical and hydrodynamic studies must be conducted. The resolution of the soils data was very low therefore, meaningful comparisons with sediments samples could not be made with great confidence. While sedimentation rates were not determined for this study site, valuable information was gained into the spatial distribution of radionuclides along a tropical riverine-estuarine system in the Caribbean. To date, the author is unaware of a similar study carried out in this region.
1.0. Introduction

Jobos Bay is the second largest estuary in Puerto Rico. It is a semi-enclosed bay situated on the south-central coast of the island (Figure 4.1) with a surface area of approximately 11 km$^2$ and a maximum depth of 10 m (Field, 2003 and Dieppa, 2008). The Jobos Bay Estuary is considered to be a coastal plain estuary with mangrove forests along the shoreline of the bay. Mangroves on the northern and eastern portion of the bay are part of the Aguirre forest (Field, 2003). The extensive mangrove forests along the western boundary of the bay is known as Mar Negro. Mar Negro together with Cayos Caribe (reef fringed, mangrove islands) is the designated Jobos Bay National Estuarine Research Reserve System (NERRS), which was established by the NOAA in 1981 (NERRS, 2010). Besides mangrove forests, local ecosystems of significance include upland dry forests, coral reefs, lagoons and seagrass beds. Ecological important species such as the brown pelican, West Indian manatee, Pelegrine Falcon and hawksbill turtle inhabit this area (NERRS, 2010). In addition to its ecological significance, Jobos Bay is used for recreational purposes including: fishing, ecotourism and other commercial activities (NERRS, 2010).
The Jobos Bay Estuary is very unique in that the major source of freshwater to the system is via ground water. Rio Seco is the only surface water channel that drains into the bay (Zitello et al., 2008). Tides in the bay are classified as mixed, but mainly diurnal in nature and based on the United States Fisheries Wetland Classification System, Jobos Bay is considered an intertidal estuarine system (Cowardin et al., 1979). Low tides are visible during the beginning of the year and high tides occur around October during periods of heavy rains (Field, 2003). Residence time in bay is approximately 5.5 days (Morelock and Williams, 2010) and currents flow in a westward and southern direction (Figure 4.2). Nearshore regions, adjacent to reefs, are wind and wave driven. With reference to surface and deep currents, movements are influenced by wind direction and speed as well as tides (Morelock and Williams, 2010).
Figure 4.2. Direction of currents in Jobos Bay. This figure is adopted from Morelock and Williams, 2010)

A large part of the Jobos Bay watershed is within the Salinas and Guayama municipalities. These municipalities possess various land use/cover types, including: upland forests, pasture, agriculture, residential and industry. The proximity of land use types such as agriculture, residential and industry may have an impact on estuarine health. In a recent study by Aldarondo-Torres and others (2010) reported that sediment quality in Jobos Bay is low to moderately polluted with trace metal, polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs). Olsen and Valiela (2010) conducted an investigation on the effects of sediment nutrient enrichment on sea grass grazing with Jobos Bay Estuary. Results from this study suggested that increased nutrients in sediment can potentially reduce the ability of sea grass to defend itself against
grazing. The increased nutrients may lower the production of chemical defense compounds in sea grass (Olsen and Valiela, 2010). This study highlights the importance of monitoring the flux, fate and transport of nutrients and pollutants (sorbed onto sediments) within the bay for effective management of this tropical ecosystem.

In an effort to contribute to the effective management of this ecologically significant estuary, the spatial and vertical distribution of $^{137}$Cs and $^{210}$Pb was determined to gain information regarding sedimentation regimes in the bay. Both $^{137}$Cs and $^{210}$Pb have been used extensively for this purpose (Ramesh et al., 2002; Aalto et al., 2003; Sanders et al. 2006; Brooks et al., 2007; Alonso-Hernandez et al. 2006). An investigation conducted by Alonso-Hernandez et al. (2006) for Cienfuegos Bay, Cuba, is most similar to the current study.

$^{137}$Cs is an anthropogenic radionuclide that has been introduced into the natural environment during nuclear weapons testing, accidental releases (Chernobyl accident in 1986) and discharges from nuclear processing plants (Avery, 1996; Bennett and Beck, 2002). Maximum inputs occurred during the 1963 followed a decrease coinciding with a ban on atmospheric testing (Alonso-Hernandez et al., 2006). The half-life of this radionuclide is approximately 30 years. In contrast, $^{210}$Pb is a naturally occurring radionuclide from the $^{238}$U series. Specifically, $^{210}$Pb is a radioactive daughter product of $^{222}$Rn emitted from rocks and soils. Half-life for $^{210}$Pb is approximately 22 years (Noller et al., 2000). Both $^{137}$Cs and $^{210}$Pb are particle reactive, their adsorption to particles and subsequent settling and accumulation in sediments can provide a historical record of environmental change for an area (Corbett et al., 2007). With reference to $^{210}$Pb, the vertical distribution of this radionuclide and its in situ decay can be used to determine
sediment accumulation rates ideally over the past 100-120 years (Swarzenski et al., 2003).

This chapter describes sedimentological and radionuclide data obtained for the purpose of conducting a preliminary assessment of sedimentation regimes within Jobos Bay. To the author’s knowledge, this is the first representation and mapping of $^{137}$Cs and $^{210}$Pb distribution within this estuary.

2.0. Methods

2.1. Sample Collection, Location and Preparation

Ten surface sediment samples were collected from Jobos Bay estuary in 2008 using a petit ponar grab sampler, collecting approximately the top 5 cm. A total of seven cores were collected within the bay using a push corer. In addition, two sediment cores were collected from a transitional area between agricultural land and Mar Negro, a protected mangrove forest that is part of the Jobos Bay National Estuarine Research Reserve (JBNERRS). The locations of grab samples and sediment cores are shown in figure 4.1. Table 4.1 contains detailed location and general characteristics for grab samples and sediment cores. Sediment cores were extruded and sliced immediately after sampling into 0.5 cm sections for the first 10 cm. Below this point, sediment cores were sliced into 1.0 cm sections to bottom. Each individual section was weighed, archived and stored in a freezer until analysis. Sediment grab samples were also weighed, archived and frozen until analysis. Prior to analysis, sediment samples were freeze-dried.
Table 4.1. Sediment core locations, length and date of collection

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>Latitude</th>
<th>Longitude</th>
<th>Sample Type</th>
<th>Core Length (cm)</th>
<th>Date Collected</th>
</tr>
</thead>
<tbody>
<tr>
<td>AF1</td>
<td>17.95099</td>
<td>-66.24952</td>
<td>Core</td>
<td>23</td>
<td>August, 2008</td>
</tr>
<tr>
<td>AF2</td>
<td>17.95146</td>
<td>-66.24662</td>
<td>Core</td>
<td>26</td>
<td>August, 2008</td>
</tr>
<tr>
<td>ACN1</td>
<td>17.95644</td>
<td>-66.21770</td>
<td>Core</td>
<td>50</td>
<td>July, 2007</td>
</tr>
<tr>
<td>ASM1</td>
<td>17.95718</td>
<td>-66.21818</td>
<td>Core</td>
<td>63</td>
<td>August, 2008</td>
</tr>
<tr>
<td>PJ1</td>
<td>17.95136</td>
<td>-66.18291</td>
<td>Core</td>
<td>68</td>
<td>August, 2008</td>
</tr>
<tr>
<td>SFM1</td>
<td>17.96098</td>
<td>-66.19993</td>
<td>Core</td>
<td>32</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JB1</td>
<td>17.94667</td>
<td>-66.16761</td>
<td>Core</td>
<td>57</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JB3</td>
<td>17.94181</td>
<td>-66.21340</td>
<td>Core</td>
<td>50</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JB4</td>
<td>17.93211</td>
<td>-66.23317</td>
<td>Core</td>
<td>50</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG1</td>
<td>17.94757</td>
<td>-66.18382</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG2</td>
<td>17.94340</td>
<td>-66.18767</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG3</td>
<td>17.94819</td>
<td>-66.19110</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG4</td>
<td>17.94706</td>
<td>-66.19870</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG5</td>
<td>17.95352</td>
<td>-66.19970</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG6</td>
<td>17.95417</td>
<td>-66.20550</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG7</td>
<td>17.94680</td>
<td>-66.20969</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG8</td>
<td>17.94829</td>
<td>-66.21597</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG9</td>
<td>17.94218</td>
<td>-66.21868</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
<tr>
<td>JBG10</td>
<td>17.92846</td>
<td>-66.22342</td>
<td>Grab sample</td>
<td>N/A</td>
<td>August, 2008</td>
</tr>
</tbody>
</table>

2.2. Sedimentological Analysis

Grain-size analysis was carried out on freeze-dried sediment core sections and surface samples. This analysis was performed using a combination of traditional wet sieving to separate larger fractions and sand (LFS) from finer-grained materials (such as clay and silts). A 63µm mesh sieve was used to separate LFS (>63 µm) from finer-grained material (<63 µm). Material that is less than 63 µm in size was further analyzed using a Micromeritics Saturn DigiSizer ® 5200 to determine the percent silt and clay within samples. Particle size classes were defined as follows: clay size ≤ 4 µm; 4 µm < silt ≤ 63 µm; LFS >63µm. (see Appendix B). Loss on ignition (LOI) was used to determine organic matter and carbonate content using a sequential method. Organic matter content was calculated as ignition loss at 550 °C. Through subsequent ignition of the sample at 950 °C, weight loss was used to estimate carbonate content in each sample.
All analysis was performed on oven-dried at 105 °C to constant weight. (see Appendix A).

2.3. Gamma Analysis

2.3.1. $^{137}$Cs determination

Sediment and soil samples were prepared for gamma analysis using procedures described in Johnson-Pyrtle and Scott (2001). Approximately, one to two grams of samples were sealed in plastic test tubes and were assayed for gamma emitters using two Canberra® high purity germanium well detectors connected to a Canberra Genie multi-channel analyzer, which records the gamma spectra in 4096 channels. These detectors were calibrated using U.S. National Institute of Standards and Technology 4357 Ocean Sediment multiline and Canberra Industries MGS-5 sediment standards. $^{137}$Cs was identified a specific energy peak at 661 keV in every other sample (every 1 cm in top 10 cm and every 2 cm below until depth of 40 cm). A Peruvian soil standard, Columbia River sediment river sediment standard, an Ocean sediment standard prepared by NIST was used to verify detector performance. Since it is believed that $^{137}$Cs has a strong affinity to clays, resulting values for samples were normalized against clay. Total $^{137}$Cs inventories were calculated for each core using the following equation:

\[ I = k \sum \rho_i t_i C_i \]  

**Equation 4.1**

Where $I$ = the $^{137}$Cs inventory (Bq/m$^2$) for each sediment core,

$C_i$ = $^{137}$Cs (Bq/g) measured in each increment

$\rho_i$ = the density for each sediment increment (g/cm$^2$)

$t_i$ = the thickness of each increment (cm)

$k = 10,000$, a constant for converting Bq/cm$^2$ to Bq/m$^2$. 
2.3.2. **Excess** $^{210}$Pb determination

Both well detectors and a planar detector were used to measure $^{210}$Pb activity concentrations in sediment and soil samples. Sealed plastic vials containing 1-2 grams of sample and aluminum tin cans containing approximately five grams of sediments were allowed reach secular equilibrium during a period of no less than 30 days prior to analysis. This time period is assumed to be the equivalent of 8 half lives of $^{222}$Rn, the immediate daughter of $^{226}$Ra. $^{214}$Pb is a daughter product of $^{222}$Rn and a precursor to $^{210}$Pb. The activity of $^{226}$Ra can be determined by determining the activity of $^{214}$Pb. Samples were analyzed for gamma emitters mentioned above for at least 24 hrs. $^{210}$Pb and $^{214}$Pb were identified by specific energy peaks of 46 keV and 351 keV, respectively. Energy and efficiency performance were calculated and factored prior to determining the actual activities (see Appendix E). Excess $^{210}$Pb (also known as unsupported $^{210}$Pb) can be estimated by subtracting $^{214}$Pb ($^{226}$Ra) activity from the total $^{210}$Pb activity. Gamma activities were also normalized with clay (Aalto et al. 2003).

2.4. **Statistical Analysis**

Statistical analysis was performed using SAS Enterprise Guide 4.2. Regression analyses were performed to assess relationship between $^{137}$Cs radionuclide activity concentrations and sediment characteristics (clay, silt, mud fraction (clay + silt) and organic matter). A One-way ANOVA was performed to assess the influence of location on the spatial distribution of radionuclides in bay.

3.0. **Results and Discussion**

3.1. **Grain-size Distribution, O.M. and Carbonate Content in Surface Grab Samples**

For all grab samples with the exception of JBG10 silt and clay-sized particles is the dominant sediment types. In contrast, JBG10 showed a greater distribution of silts
and LFS. The greater distribution of mud fraction (% clay + % silts) in nine of the ten grab samples suggests potential low energy depositional areas for these finer-grained materials. With reference to grab sample JBG10, the evenly distributed LFS and silt-sized particles suggests a higher energy depositional area. This sample was collected at a major opening in bay where there is a potentially greater oceanic (wave) influence. Spatially, all grab samples were collected from the middle portion of the bay, where depths ranged from 6-9 m (Figure 4.1). This area may be a potential area of sediment focusing.

Table 4.2. Grain-size distribution for grab samples collected within the bay.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>Clay (%)</th>
<th>Silt (%)</th>
<th>Larger Fraction + Sand (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>JBG1</td>
<td>26.24</td>
<td>66.07</td>
<td>7.69</td>
</tr>
<tr>
<td>JBG2</td>
<td>24.96</td>
<td>66.37</td>
<td>8.67</td>
</tr>
<tr>
<td>JBG3</td>
<td>27.04</td>
<td>65.53</td>
<td>7.43</td>
</tr>
<tr>
<td>JBG4</td>
<td>26.24</td>
<td>60.95</td>
<td>12.82</td>
</tr>
<tr>
<td>JBG5</td>
<td>24.77</td>
<td>61.46</td>
<td>13.77</td>
</tr>
<tr>
<td>JBG6</td>
<td>24.67</td>
<td>61.36</td>
<td>13.97</td>
</tr>
<tr>
<td>JBG7</td>
<td>25.93</td>
<td>59.79</td>
<td>14.28</td>
</tr>
<tr>
<td>JBG8</td>
<td>26.69</td>
<td>56.19</td>
<td>17.12</td>
</tr>
<tr>
<td>JBG9</td>
<td>22.67</td>
<td>72.64</td>
<td>4.68</td>
</tr>
<tr>
<td>JBG10</td>
<td>6.51</td>
<td>49.53</td>
<td>43.96</td>
</tr>
</tbody>
</table>
Table 4.3 includes estimated O.M. and carbonate content for the grab samples collected within the bay. All grab samples appear to have an abundance of shells (pieces of shells, broken up). Organic matter in these samples is composed of leaves and roots from surrounding mangrove forests, seagrass and macro algae communities. This material may have been washed into some of these locations through tides and currents since most of these grab samples were collected in the inner portions of the bay (Figure 4.3). Grab sample JBG10 has the lowest O.M (4.6 %) while JBG7 has the highest (15.7 %). Surface sample, JBG8 has the highest (15.8 %) carbonate content whereas JBG9 had the lowest (5.7 %).
### Table 4.3. Percent O.M. and carbonate content in surface grab samples.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>O.M. (%)</th>
<th>Carbonates (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>JBG1</td>
<td>12.8</td>
<td>9.1</td>
</tr>
<tr>
<td>JBG2</td>
<td>11.1</td>
<td>12.6</td>
</tr>
<tr>
<td>JBG3</td>
<td>15.3</td>
<td>9.2</td>
</tr>
<tr>
<td>JBG4</td>
<td>15.1</td>
<td>13.7</td>
</tr>
<tr>
<td>JBG5</td>
<td>12.9</td>
<td>13.2</td>
</tr>
<tr>
<td>JBG6</td>
<td>11.5</td>
<td>15.3</td>
</tr>
<tr>
<td>JBG7</td>
<td>15.8</td>
<td>11.0</td>
</tr>
<tr>
<td>JBG8</td>
<td>14.2</td>
<td>15.8</td>
</tr>
<tr>
<td>JBG9</td>
<td>15.4</td>
<td>5.7</td>
</tr>
<tr>
<td>JBG10</td>
<td>4.6</td>
<td>11.4</td>
</tr>
</tbody>
</table>

### 3.2. Grain-size Distribution, O.M. and Carbonate Content in Sediment Cores

#### 3.2.1. AF1 and AF2

Sediment cores AF1 and AF2 were collected within the Jobos Bay watershed. Specifically, these cores were obtained from an intermediate area between an agricultural farm and Mar Negro a protected mangrove forest (a part of the Jobos Bay National Research Reserve). Rationale for collecting sediment cores at this location was to estimate sediment flux in this protected area from the adjacent agricultural lands. Table 4.4 contains averaged grain-size distribution for both AF1 and AF2. Figures 4.4-4.5 shows grain-size profiles for AF1 and AF2, respectively. A simple comparison of these cores indicates that there is some variation in sediment distribution in the cores based on averaged values (Table 4.4). In both cores, silt-sized particles dominate each core of these cores. This pattern is demonstrated both in Table 4.4 and in figures 4.4-4.5. The abundance of silt-sized particles may be reflective of soil types within the JB watershed.
### Table 4.4. Grain-size distribution for all sediment cores.

<table>
<thead>
<tr>
<th>Sample Core</th>
<th>Clay (%)</th>
<th>Silt (%)</th>
<th>LFS(%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>AF1</td>
<td>18.9</td>
<td>71.4</td>
<td>9.8</td>
</tr>
<tr>
<td>AF2</td>
<td>14.3</td>
<td>68.8</td>
<td>16.9</td>
</tr>
<tr>
<td>ACN1</td>
<td>12.4</td>
<td>37.3</td>
<td>50.3</td>
</tr>
<tr>
<td>ASM1</td>
<td>5.7</td>
<td>34.8</td>
<td>59.6</td>
</tr>
<tr>
<td>PJ1</td>
<td>10.5</td>
<td>64.0</td>
<td>25.6</td>
</tr>
<tr>
<td>SFM1</td>
<td>23.2</td>
<td>66.4</td>
<td>10.3</td>
</tr>
<tr>
<td>JB1</td>
<td>12.4</td>
<td>70.1</td>
<td>17.5</td>
</tr>
<tr>
<td>JB3</td>
<td>9.2</td>
<td>24.9</td>
<td>65.9</td>
</tr>
<tr>
<td>JB4</td>
<td>11.6</td>
<td>51.7</td>
<td>36.7</td>
</tr>
</tbody>
</table>

**Figure 4.4.** Grain-size distribution for sediment core AF1.
O.M. content for AF1 exhibits a variable, but decreasing trend with depth (Figure 4.6). Conversely, carbonate content appear shows an increasing trend with depth within the AF1 core (Figure 4.7). The decrease in organic matter with depth may be attributed to a decrease in biomass (mangrove roots) under the surface. As mentioned before, this core was collected in a transitional area between agricultural areas and mangrove forest. The increase in carbonate content with depth may reflective of mixing and burial carbonate material. Both O.M. and carbonate content within AF2 were highly variable with depth. No obvious patterns were visible in the depth profiles for these characteristics. Figures 4.8-4.9 contains the depth profiles for percent O.M. and carbonate content for AF2. Despite the variation in trends for O.M. and carbonate content between the two cores, average O.M. and carbonate content are relatively similar.
in both cores (Table 4.5). This can be expected since these cores were collected relatively close together (Figure 4.1).

Table 4.5. Percent O.M. and carbonate content for all cores.

<table>
<thead>
<tr>
<th>Sample Core</th>
<th>O.M. (%)</th>
<th>Carbonate (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>AF1</td>
<td>15.1</td>
<td>14.4</td>
</tr>
<tr>
<td>AF2</td>
<td>12.2</td>
<td>11.8</td>
</tr>
<tr>
<td>ACN1</td>
<td>9.8</td>
<td>10.4</td>
</tr>
<tr>
<td>ASM1</td>
<td>20.5</td>
<td>8.9</td>
</tr>
<tr>
<td>PJ1</td>
<td>8.4</td>
<td>5.1</td>
</tr>
<tr>
<td>SFM1</td>
<td>11.6</td>
<td>4.4</td>
</tr>
<tr>
<td>JB1</td>
<td>10.8</td>
<td>6.1</td>
</tr>
<tr>
<td>JB3</td>
<td>9.1</td>
<td>18.2</td>
</tr>
<tr>
<td>JB4</td>
<td>11.8</td>
<td>22.2</td>
</tr>
</tbody>
</table>

Figure 4.6. Depth profile of organic matter for AF1 sediment core.
Figure 4.7. Depth profile of carbonate content for AF1 sediment core.

Figure 4.8. Depth profile of organic matter for AF2 sediment core.
3.2.2. Sediment cores collected with the Bay

When comparing average grain-size distribution, we see that the distributions varied among cores (Table 4.4). On average, LFS was the dominant sediment type in ACN1, ASM1 and JB3 (Table 4.4). These cores comprised of 50% or more of LFS. Within these cores the LFS characteristics differed within each core. For instance, in ACN1 the inorganic LFS was composed of white-beach sand while the organic portion comprised of broken shells and corals well as small amounts of leaf litter from adjacent mangrove forest. ASM1 is the most interesting of all cores collected from bay in that most of the LFS fraction consisted of ash, possibly sugar cane ash discharged into the bay during operation of Aguirre Sugar Mill. Broken coral fragments were the dominant LFS in this core. JB3 was collected closest to outline mangrove islands located on the western side of the bay. This part of the bay also has coral communities, which may supply

Figure 4.9. Depth profile of carbonate content for AF2 sediment core.
broken carbonate fragments to this site. Figures 4.10-4.12 show the depth profiles of grain-size for all cores except AF1 and AF2.

Figure 4.10. Grain-size distribution for sediment core ACN1.
Figure 4.11. Grain-size distribution for sediment core ASM1.

Figure 4.12. Grain-size distribution for sediment core JB3.
Silt-sized particles are the dominant sediment type within SFM1, PJ1, JB1 and JB4 (Table 4.4). An evaluation of the depth profiles for SFM1, PJ1 and JB1 supports this statement (Figures 4.13-4.15). These patterns may also be explained by the fact these cores were collected closest to the shorelines of the bay and very close to mangrove forest. These cores were collected from very shallow and low energy making it possible for these “finer” particles to be deposited. The sediment profile for JB4 shows that silt-sized particles are abundant in the surficial layers however, at a depth of around 16 cm, an alternating pattern of silt and LFS particles are intermixed (Figure 4.16). This pattern continues until a depth of 28 cm (Figure 4.16). Below this point, LFS becomes prevalent (Figure 4.16). Variation in sedimentary layers for JB4 may suggest a period where a storm event may have caused LFS from neighboring mangrove island (coarse beach sand and/or coral fragments) to move and be deposited in this area. Since this core is the most seaward (Figure 4.1) but still located in shallow area, wave action may cause resuspension of material in this area, hence the defined mixed layer at 16 cm depth.
Figure 4.13. Grain-size distribution for sediment core SFM1.

Figure 4.14. Grain-size distribution for sediment core PJ1.
Figure 4.15. Grain-size distribution for sediment core JB1.

Figure 4.16. Grain-size distribution for sediment core JB4.
Average O.M. and carbonate content for all sediment cores are recorded in table 4.5. The highest amount of O.M. is 36 % measured in ASM1 at a depth of 20 cm. The lowest O.M. content (< 1%) was measured in PJ1 at a shallow depth of 2 cm. Overall, percent O.M. content was variable in ACN1, ASM1 and PJ1 (Figures 4.17-4.19). Sediments cores SFM1, JB1, JB3, JB4 showed a decreasing trend in organic matter with depth. This may be attributed to sampling locations. These cores were collected close to the shoreline in shallow areas, where the presence of mangroves, seagrass and macroalgae may cause an increase in O.M. deposition on the surface. The reduction in root system with depth may help explain the decreasing trends visible in core profiles (Figures 4.20-4.23).

![Graph of O.M. content versus depth](image.png)

**Figure 4.17.** Depth profile of O.M. for ACN1 sediment core.
Figure 4.18. Depth profile of O.M. for ASM1 sediment core.

Figure 4.19. Depth profile of O.M. for PJ1 sediment core.
Figure 4.20. Depth profile of O.M. for SFM1 sediment core.

Figure 4.21. Depth profile of O.M. for JB1 sediment core.
Figure 4.22. Depth profile of O.M. for JB3 sediment core.

Figure 4.23. Depth profile of O.M. for JB4 sediment core.
Percent carbonate content in sediment cores collected within the bay is highly variable with increased depth (Figures 4.24-4.30). Despite such variability, decreasing trends are somewhat visible in the JB1 and JB3 cores (Figures 4.28-4.29). One may assume that this trend may be a result of a decrease in the carbonate shelled or precipitating organisms at these sites with depth. Conversely, ACN1 (Figure 4.24) and JB4 (Figure 4.30) exhibited increasing trends with depth. A possible explanation for this is that over time sediment accumulated at the surface burying old carbonate material that was buried in the recent past.

**Figure 4.24.** Depth profile of carbonate content for ACN1 sediment core.
Figure 4.25. Depth profile of carbonate content for ASM1 sediment core.

Figure 4.26. Depth profile of carbonate content for SFM1 sediment core.
Figure 4.27. Depth profile of carbonate content for PJ1 sediment core.

Figure 4.28. Depth profile of carbonate content for JB1 sediment core.
Figure 4.29. Depth profile of carbonate content for JB3 sediment core.

Figure 4.30. Depth profile of carbonate content for JB4 sediment core.
3.3. Vertical Distribution of Radionuclides For Sediment Cores

3.3.1. $^{137}$Cs distribution

The depth profiles of $^{137}$Cs for all cores are shown in figures 4.31-4.39. Table 4.6 provides the computed $^{137}$Cs inventories for each sediment core. The vertical profiles of $^{137}$Cs do not reflect the atmospheric inputs function for the area (Figure 3.17). Within the each profile there are multiple subsurface peaks, none of which may be used to accurately determine sedimentation rates for the bay. To derive some meaningful comparisons, $^{137}$Cs inventories were computed for each of the cores in the bay. JB4 had the highest inventory (4.94 dpm/cm$^2$) while JB3 had the lowest inventory (0.84 dpm/cm$^2$) despite SFM1 being the shortest core in length (Table 4.1). Currents in Jobos Bay flow in a south and west direction, JB4 was collected closest to the mouth of the bay (most south and western core). It may be assumed that lateral transport processes within the bay may have deposited $^{137}$Cs laden sediments at this site. The lower amounts of $^{137}$Cs inventories computed for JB3 may be explained by its grain-size distribution. LFS dominate this core. $^{137}$Cs is believed to have a strong affinity to fine-grained particles (He and Wallings, 1996) therefore, the greater amounts of LFS reduces the potential for $^{137}$Cs to be deposited at that sampling site. AF1 and AF2 as stated before are cores collected within the watershed. AF2 had the highest computed inventory when compared to AF1 (Table 4.6). The AF1 core is shorter in length when compared to AF2, which may provide an explanation for differences in inventory. Further evaluation is necessary in order to gain greater insight reported differences in cores.
Table 4.6. \(^{137}\)Cs inventories for all sediment cores.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>(^{137})Cs Inventory (dpm/cm(^2))</th>
</tr>
</thead>
<tbody>
<tr>
<td>AF1</td>
<td>1.3</td>
</tr>
<tr>
<td>AF2</td>
<td>3.1</td>
</tr>
<tr>
<td>ACN1</td>
<td>1.6</td>
</tr>
<tr>
<td>ASM1</td>
<td>2.5</td>
</tr>
<tr>
<td>SFM1</td>
<td>1.2</td>
</tr>
<tr>
<td>PJ1</td>
<td>1.5</td>
</tr>
<tr>
<td>JB1</td>
<td>1.9</td>
</tr>
<tr>
<td>JB3</td>
<td>0.84</td>
</tr>
<tr>
<td>JB4</td>
<td>4.9</td>
</tr>
</tbody>
</table>

Figure 4.31. \(^{137}\)Cs distribution normalized with clay for sediment core AF1.
Figure 4.32. $^{137}$Cs distribution normalized with clay for sediment core AF2.

Figure 4.33. $^{137}$Cs distribution normalized with clay for sediment core ACN1.
Figure 4.34. $^{137}\text{Cs}$ distribution normalized with clay for sediment core ASM1.

Figure 4.35. $^{137}\text{Cs}$ distribution normalized with clay for sediment core SFM1.
Figure 4.36. $^{137}$Cs distribution normalized with clay for sediment core PJ1.

Figure 4.37. $^{137}$Cs distribution normalized with clay for sediment core JB1.
Figure 4.38. $^{137}$Cs distribution normalized with clay for sediment core JB3.

Figure 4.39. $^{137}$Cs distribution normalized with clay for sediment core JB4.
3.3.2. $^{210}$Pb distribution

Similar to the vertical distributions of $^{137}$Cs, the excess $^{210}$Pb profiles showed irregularities in all the sediment cores (Figures 4.40-4.48). These profiles deviated from the simple exponential decline expected for this radionuclide. Irregularities with the excess $^{210}$Pb profiles may suggest irregular sedimentation rates or accelerated sedimentation in certain areas in recent history. Inventories for $^{210}$Pb were also estimated in each of the sediment cores. Similar to $^{137}$Cs calculated inventories, JB4 had the highest inventory (194 dpm/cm$^2$) whereas JB3 had the lowest (0.66 dpm/cm$^2$). As mentioned above, JB3 is dominated by LFS, which does not provide a sufficient surface area for the adsorption of radionuclides to its surface. On other hand, JB4 may be a site of sediment focusing hence the higher inventory of the natural radionuclide of $^{210}$Pb. In addition, JB4 is the southern most sediment core and may be a site of greater oceanic influence when compared to the other cores in the bay. The coastal ocean may be a source of radionuclides to this area.
Figure 4.40. Excess $^{210}$Pb profiles normalized with clay for AF1.

Figure 4.41. Excess $^{210}$Pb profiles normalized with clay for AF2.
Figure 4.42. Excess $^{210}\text{Pb}$ profiles normalized with clay for ACN1.

Figure 4.43. Excess $^{210}\text{Pb}$ profiles normalized with clay for ASM1.
Figure 4.44. Excess $^{210}\text{Pb}$ profiles normalized with clay for SFM1.

Figure 4.45. Excess $^{210}\text{Pb}$ profiles normalized with clay for PJ1.
Figure 4.46. Excess $^{210}$Pb profiles normalized with clay for JB1.

Figure 4.47. Excess $^{210}$Pb profiles normalized with clay for JB3.
3.3.3. Spatial distribution of radionuclides in surface sediments

The $^{137}$Cs and excess $^{210}$Pb activity concentrations in surface sediments are reported in Table 4.7. Figures 4.49-4.50 are maps showing the distributions of $^{137}$Cs and excess $^{210}$Pb in surface samples. Surface samples for ASM1 had the highest level of $^{137}$Cs (1.87 dpm g$^{-1}$ clay). While the following samples reported no $^{137}$Cs in their surfaces or measures were below detection limit: JBG2, JBG9, JBG10. A possible explanation for these low values may be increased cation exchange competition at these sites. In addition, it may be possible that $^{137}$Cs laden sediment may have been moved out of the inner bay area via erosion, waves and/or currents. Additionally, sediment core JB3 has the lowest $^{137}$Cs inventories when compared to all cores (Table 4.6) however, the surface samples of this core seem to be “enriched” with this nuclide. Sediment composition at the surface of this core may have an influence on $^{137}$Cs retention on the
surface of this core however, further testing is necessary to obtain a more conclusive explanation. $^{210}\text{Pb}$ was detected in all surface samples (Table 4.7). The primary input of freshwater to the bay is through ground water. Freshwater entering the bay through this route may serve as a source of natural radionuclides to the bay hence, the detection of $^{210}\text{Pb}$ in all surface samples. High levels of excess $^{210}\text{Pb}$ were measured in JB4 (80.44 dpm g$^{-1}$ clay) and JBG10 (19.96 dpm g$^{-1}$ clay) surface samples (Table 4.7). These samples are the southern-most sites in the bay and may be significantly influenced the coastal ocean. In fact, the coastal ocean may be a major source of $^{210}\text{Pb}$ to these sites. In addition, this observation may support the assumption of sediment focusing occurring at these sampling sites.

**Table 4.7.** Radionuclide activity concentrations in surface sediments. Activity of the first 5 cm of sediment cores were averaged and presented in this table.

<table>
<thead>
<tr>
<th>Sediment ID</th>
<th>$^{137}\text{Cs}$ (dpm g$^{-1}$ clay)</th>
<th>$^{210}\text{Pb}_{ex}$ (dpm g$^{-1}$ clay)</th>
</tr>
</thead>
<tbody>
<tr>
<td>JBG1</td>
<td>0.16</td>
<td>0.94</td>
</tr>
<tr>
<td>JBG2</td>
<td>0.00</td>
<td>6.51</td>
</tr>
<tr>
<td>JBG3</td>
<td>0.45</td>
<td>3.16</td>
</tr>
<tr>
<td>JBG4</td>
<td>0.08</td>
<td>3.83</td>
</tr>
<tr>
<td>JBG5</td>
<td>0.24</td>
<td>5.40</td>
</tr>
<tr>
<td>JBG6</td>
<td>0.57</td>
<td>1.79</td>
</tr>
<tr>
<td>JBG7</td>
<td>0.06</td>
<td>6.80</td>
</tr>
<tr>
<td>JBG8</td>
<td>0.11</td>
<td>3.95</td>
</tr>
<tr>
<td>JBG9</td>
<td>0.00</td>
<td>8.68</td>
</tr>
<tr>
<td>JBG10</td>
<td>0.00</td>
<td>19.96</td>
</tr>
<tr>
<td>ACN1</td>
<td>0.42</td>
<td>1.65</td>
</tr>
<tr>
<td>ASM1</td>
<td>1.87</td>
<td>7.69</td>
</tr>
<tr>
<td>PJ1</td>
<td>0.43</td>
<td>4.27</td>
</tr>
<tr>
<td>SFM1</td>
<td>0.31</td>
<td>3.62</td>
</tr>
<tr>
<td>JB1</td>
<td>0.67</td>
<td>7.97</td>
</tr>
<tr>
<td>JB3</td>
<td>0.24</td>
<td>6.84</td>
</tr>
<tr>
<td>JB4</td>
<td>1.28</td>
<td>80.44</td>
</tr>
</tbody>
</table>
Figure 4.49. Spatial distribution of $^{137}$Cs in surface samples within Jobos Bay

Figure 4.50. Spatial distribution of excess $^{210}$Pb in surface samples within Jobos Bay
3.4 Statistical Analysis

To test the possible influence of location on radionuclide a One-way ANOVA was performed. Results from this statistical test indicate that there is a location effect on $^{137}$Cs distribution with Jobos Bay. The reported F-value (2.43) was significant at a significance level of $\alpha = 0.05$. This result can be used to support the earlier suggestion of sediment focusing in the central and deeper parts of the bay. To further evaluate the effect of sediment characteristics on radionuclide distribution, regression analyses were performed. Resulting $R^2$ values are presented in Table 4.8. Very few statistical significant relationships were determined between $^{137}$Cs and various sediment characteristics. Similar findings were reported by Zaborska et al. (2010) for $^{137}$Cs in the north-western Barents Sea. The results from regression analysis in this study suggest that particle composition may not have a major influence on sedimentary $^{137}$Cs. Activity concentrations of this radionuclide may be driven by physical (transport) and biogeochemical processes.

**Table 4.8.** Statistical relationships between $^{137}$Cs and sediment characteristics.

<table>
<thead>
<tr>
<th>Sample ID</th>
<th>n</th>
<th>Clay (%)</th>
<th>Silt (%)</th>
<th>Mud (%)</th>
<th>O.M. (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>AF1</td>
<td>17</td>
<td>0.28*</td>
<td>0.02</td>
<td>0.26*</td>
<td>0.14</td>
</tr>
<tr>
<td>AF2</td>
<td>20</td>
<td>0.00</td>
<td>0.59*</td>
<td>0.31</td>
<td>0.0</td>
</tr>
<tr>
<td>ACN1</td>
<td>24</td>
<td>0.15</td>
<td>0.14</td>
<td>0.15</td>
<td>0.04</td>
</tr>
<tr>
<td>ASM1</td>
<td>24</td>
<td>0.26</td>
<td>0.10</td>
<td>0.14</td>
<td>0.11</td>
</tr>
<tr>
<td>SFM1</td>
<td>15</td>
<td>0.24</td>
<td>0.26</td>
<td>0.10</td>
<td>0.34*</td>
</tr>
<tr>
<td>PJ1</td>
<td>24</td>
<td>0.07</td>
<td>0.04</td>
<td>0.06</td>
<td>0.05</td>
</tr>
<tr>
<td>JB1</td>
<td>23</td>
<td>0.19*</td>
<td>0.06</td>
<td>0.14</td>
<td>0.38*</td>
</tr>
<tr>
<td>JB3</td>
<td>23</td>
<td>0.02</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
</tr>
<tr>
<td>JB4</td>
<td>22</td>
<td>0.02</td>
<td>0.09</td>
<td>0.09</td>
<td>0.16</td>
</tr>
<tr>
<td>Surface Samples</td>
<td>10</td>
<td>0.11</td>
<td>0.00</td>
<td>0.05</td>
<td>0.02</td>
</tr>
</tbody>
</table>

*Indicates those relationships that were statistically significant at a $P < 0.05$. 

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4.0. Conclusion

Sedimentological analysis of sediment cores and surface samples showed that finer material (silt and clay sized particles) may be “focused” in the inner, deeper portions of the bay with coarser material lining the shores of this estuary. The lack of significant correlations between $^{137}$Cs and clay, silt, mud fraction and O.M. suggests that sedimentology may not have a great influence on this radionuclide therefore, further studies are necessary to explain the spatial and vertical distribution of $^{137}$Cs and $^{210}$Pb in Jobos Bay estuary. The distribution and activities for $^{137}$Cs and $^{210}$Pb were studied in the Jobos Bay estuary. Activity concentration varied spatially in surface samples and vertically within sediment cores. This variability may be attributed to a variety of processes such as lateral transport, resuspension and biogeochemical activities. The erratic profiles for both radionuclides did not allow for the computation of sedimentation rates in this estuary. However, these environmental tracers ($^{137}$Cs and $^{210}$Pb) provided valuable insight into sediment movements within the Jobos Bay Estuary. Additional research must be conducted in order to determine the importance of land areas in acting as a secondary source of $^{137}$Cs to the bay. Biogeochemical and mineralogical analysis must be performed in order to discern and understand processes governing sediment distribution in the bay.
1.0. Introduction
The transfer of material from watersheds to estuaries is complex (Phillips and Slattery, 2007). To link soil erosion to in-stream sedimentation and transitional marine ecosystems such as estuaries, requires an understanding of the magnitude and variation of water mediated soil erosion within and between landscapes as well as the hydrologic and estuarine hydrodynamic processes influencing sediment delivery and deposition.

Needless to say, water mediated soil erosion is a global environmental problem that links terrestrial environments to aquatic ecosystems. Soil erosion not only affects soil quality but consequently reduces crop productivity and biodiversity in the landscape (Lal, 1998). Beyond the on-site impacts of soil erosion, off-site effects include: reduction in reservoir storage, reduction in primary productivity and increased turbidity in aquatic ecosystems (Crossland et al., 2005). A source of major concern for coastal managers in particular, is the potential for sediments to transport nutrients and pollutants to estuarine systems, which may disrupt aquatic ecosystem functions and health (Crossland et al., 2005). Prior to any attempts of linking soil erosion to sediment delivery to estuaries, conceptualizing and characterizing hydrologic responses within watersheds is critical. Linking soil erosion to sediment delivery to estuaries is not necessarily a simple watershed or coastal issue, but a complex hydrological issue that must be considered in conjunction with watershed morphology, land use/land cover (LULC), soils and hydrological principles. Connecting relevant processes such as geomorphology, hydrology and ecology is lacking (Hupp, 2000). While Hupp (2000) focused on the south-eastern coastal plains of the
United States, the importance of making such connections in the tropics particularly of Small Island Developing States (SIDS) in the Caribbean is even more significant.

The ability to link soil erosion to sediment delivery is influenced by hydrological and hillslope connectivity, overland flow (runoff), soil types and LULC. Steep slopes may not be the only supply of sediments to channels while foothills may serve as a sediment source especially if there is a predominance of overland flow in the watershed (Harvey, 2001; Harvey 2002). Given this interrelationship it is imperative that the hydrologic response (including geomorphic characteristics, LULC and soils) of watershed be considered in such studies. Knowledge of the geomorphic characteristics will allow us to discern the buffering capacity of the watershed with reference to storage of sediments in the landscape (Harvey, 2001). LULC is also an important factor in soil erosion and sediment delivery studies. Certain land use practices tend to accelerate soil loss more so than others (Pimentel and Kounang, 1998). For instance, soils covered with dense forest vegetation are less susceptible to the erosive impact of raindrops than agricultural lands or bare soil (Pimentel and Kounang, 1998). Vegetation patterns in the landscape also have an influence of runoff connectivity and subsequently soil loss and sediment transport in the landscape (Mayor et al., 2008). Essentially, to understand the source sink relationship of sediment production to sediment delivery processes from the watershed to in-stream systems, hydrologic principles, geomorphic characteristics, soils and land use data must be combined so our ability to quantify and manage the in-stream impact of soil erosion will improve. The aim of this study was to develop and implement a conceptual framework for linking watershed hydrologic response to sediment
production, sediment transport and sediment delivery. Specifically, the objectives of this study were to:

1. Conceptualize and characterize watershed hydrologic response;
2. Conceptualize and characterize watershed geomorphology;
3. Identify potential sediment source areas in watersheds.

2.0. Study Area
Puerto Rico is the fourth largest island in the Caribbean with an estimated area of 8,895 km$^2$ (Daly et al., 2003). The island is bordered by the Atlantic Ocean to the north and the Caribbean Sea to the south. With an estimated population density of 438 individuals/km$^2$, it is considered to be one of the most densely populated areas in the United States and its territories (Martinuzzi et al., 2007). Geographically, the island is mountainous with extensive coastlines in the north and south (Field, 2003; Boose et al., 2004). Cordillera Central, the mountain range of Puerto Rico, separates the island’s northern and southern coastal plains, the highest peak being Cerro de Punta, measured at an elevation of 1350 m (Malmgren and Winter, 1999).

For this study, two watersheds were selected on mainland Puerto Rico. Figure 5.1. shows the location of study watersheds on mainland Puerto Rico. Rio Espiritu Santo (RES) is located on the north-eastern coast of the island where as Jobos Bay (JB) watershed is located within the south-central part of Puerto Rico. Besides the obvious differences between the sizes of the two watersheds, there is variation in the amount of rainfall received in each watershed in addition to differences in soils, geology and LULC. Table 5.1. provides a comparison of catchment, soil, geology and rainfall for study watersheds.
Table 5.1. Catchment characteristics for study watersheds

<table>
<thead>
<tr>
<th>Variables</th>
<th>RES</th>
<th>JB</th>
</tr>
</thead>
<tbody>
<tr>
<td>Area (km²)</td>
<td>75</td>
<td>167</td>
</tr>
<tr>
<td>Annual Average Rainfall (mm)</td>
<td>3600</td>
<td>1130</td>
</tr>
<tr>
<td>Geologic Types</td>
<td>9</td>
<td>12</td>
</tr>
<tr>
<td>Soil Types</td>
<td>28</td>
<td>29</td>
</tr>
<tr>
<td>LULC Types</td>
<td>13</td>
<td>11</td>
</tr>
</tbody>
</table>

Figure 5.1. Shows the location of study watersheds, mainland Puerto Rico.

3.0. Methods

3.1. GIS Data Sources

The primary data input layers for this analysis were obtained from various sources and at various resolutions. The DEM were obtained from the United States Geological Survey at a 30 m resolution. The vector soil data were obtained from the United States Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS).

The Parameter-Elevation on Independent Slopes Model (PRISM) rainfall dataset was
used to derive erosivity factor (R). Refer to chapter 3 for more detail of R-factor derivation. Land use and Land cover data was obtained from National Land Cover Database (NLCD). Table 5.2 contains detail of data source and resolution for the generated raster layer.

Table 5.2. Data sources and resolution used in the analysis

<table>
<thead>
<tr>
<th>Primary Data</th>
<th>Secondary Data</th>
<th>Source</th>
<th>Resolution</th>
</tr>
</thead>
<tbody>
<tr>
<td>DEM</td>
<td>Profile Curvature</td>
<td>USGS</td>
<td>30m</td>
</tr>
<tr>
<td>LULC</td>
<td>C-Factor</td>
<td>NLCD</td>
<td>30m*</td>
</tr>
<tr>
<td>Rainfall</td>
<td>R-Factor</td>
<td>PRISM</td>
<td>225m*</td>
</tr>
<tr>
<td>Soils</td>
<td>Hydrologic Soil Group</td>
<td>SSURGO</td>
<td>30m</td>
</tr>
</tbody>
</table>

* This raster layer was resampled to a 30 m resolution for consistency dem resolution and spatial analysis.

3.2. Development of Conceptual Framework

A conceptual framework is often used as a problem-solving tool to make connections between processes that influence an outcome. For this study, a conceptual framework was developed to link processes involved in production, availability and transport of sediment from watersheds to sediment delivery to aquatic systems. The key variables in this framework are: rainfall, soil type, LULC and geomorphology. These variables interact in complex ways producing a wide range of responses depending on the various combinations of these variables on the landscape. To conceptualize and characterize these responses and their influence on sediment production and transport the following questions were explored (Figure 5.2): 1) what is the role of geomorphology, soils, LULC and the terrestrial hydrologic cycle in soil erosion processes, sediment production, availability and transport that leads to in-stream sedimentation? and 2) what factors control sediment transport and delivery within the watershed and ultimately sediment loading to streams and estuaries?
The first question: ‘what is the role of geomorphology, soils, LULC and the terrestrial hydrologic cycle in soil erosion processes that leads to in-stream sedimentation?’ is essentially asking how does geomorphology, soil, LULC and hydrology influence sediment supply and availability? The amount of sediment available (supply) within the watershed will provide an estimate of sediment yield to stream channels and to estuaries. In general, within a watershed an abundance of convex or erosional slopes will indicate “active” source areas and supply of sediment available for transport processes (Figure 5.3). The second question is related to the first in that if there
is sediment available for transport then, what is the watershed’s ability to transport available sediments to adjacent stream networks and eventual delivery to estuaries? The underlying question being: Is this watershed transport-limited?

The relationship of sediment supply and sediment availability for transport may be complex depending on the spatial occurrence and adjacency of concave and convex slopes. It was assumed that optimum combination of these two processes (sediment supply and the presence of transport processes) can and will facilitate sediments produced within the watershed to reach in-stream and estuarine system. In other words, a watershed can have ample sources of sediments, but an absence of transport processes to facilitate transport will not deliver sediments to aquatic system conversely, or it may have abundant transport processes but limited sediment supply available for transport. In a supply-limited watershed, total area available to produce sediment (a function of slope and non-protective LULC) is limited therefore, very low or no sediment is available for transport. The abundance of concave slopes can also cause supply-limiting environment by actively providing depositional areas for sediments (Figure 5.4).

To evaluate, sediment transport limitation within a watershed, interactions between rainfall, geomorphology, soil type and LULC must be understood. The representation and inclusion of runoff in the conceptual model is important because it is the primary mechanism by which sediment can be transported from one zone to the next. Mechanisms for runoff generation include: Hortonian overland flow (HOF), subsurface storm flow (SSSF) and saturation overland flow (SOF) (Montgomery et al., 1997). In a transport-limited system, soil infiltration rate is assumed to be high despite LULC type resulting in a low runoff potential (Brooks et al., 2003) (Figure 5.5). Sediment transport
is not limited in a watershed where runoff potential is high. A high runoff potential may be a result of: (1) low infiltration rates of soil units or (2) in the event that rainfall intensity exceeds the infiltration rates of soils (Figure 5.6). In either case, surface overland flow provides a means by which sediments can be moved from one area to another.

![Sub-model showing the conceptualization of a watershed where sediment supply is not limited.](image)

**Figure 5.3.** Sub-model showing the conceptualization of a watershed where sediment supply is not limited.
Figure 5.4. Sub-model showing the conceptualization of a watershed that is sediment supply-limited.
Figure 5.5. Sub-model showing the conceptualization of a sediment transport-limited watershed.
Figure 5.6. Sub-model showing the conceptualization of a watershed where sediment transport is not limited.

The questions that are addressed in this conceptual model are basic, but an understanding of these dynamic processes as posed by these basic questions will help address a wide range of complex environmental interactions and facilitate accurate modeling of processes involved in sediment delivery to an aquatic system. This conceptual model will initiate a systematic way of relating the potential impact of soil erosion to sediment delivery to estuaries. Also, such a conceptual framework guided by
GIS, GPS and remote sensing will allow users to gather basic environmental information systematically within the context of each unique watershed and their hydrologic response.

3.3. Identification of Potential Sediment Source Areas

Potential source areas within the conceptual framework were defined as areas where non-protective LULC were present on convex (erosional) slopes. The rationale for this decision is that convex slopes are zones where flow accelerates. Acceleration of water flowing through the landscape can provide a source of erosive power, which can potentially remove and transport large amount of material. It is also recognized that vegetated convex slopes can limit the amounts of sediments produced and available for transport (sediment supply-limited). In this study, the distribution of non-protective covering on convex slopes was analyzed in order to identify potential source areas of sediment in the watershed. ArcGIS 9.2. Spatial Analyst Combinational AND tool was used to intersect protective and non-protective areas to differentiate potential source area from potential supply-limited areas (See details below).

3.4. Sediment Availability, Transport and Delivery

Sediment availability in the conceptual model is represented as the difference between sediment source areas (convex slopes) and sediment storage areas (concave slopes). The assumption here is that if the sediment source areas are more abundant in the landscape when compared to storage areas then there is “excess” sediment that is available for transport (supply not-limited). This sediment may be stored temporarily in the watershed for a short period before it is transported to adjacent stream networks and subsequent delivery to estuaries. In the event where sediment source areas are not prevalent in the watershed and storage areas dominates there may be limited amounts of
sediments available for transport therefore this system is supply-limited. It is important
to note at this point that while a system may be supply-limited, this does not necessarily
mean that there is no potential for delivery. Supply-limited simply implies that sediment
loading will not be as much as what may be expected or predicted for a system where
sediment supply is not limited.

Sediment supply is not the only limitation that influences delivery to a stream or
estuary. A watershed’s ability to channel runoff within the watershed influences
sediment transport through the watershed. A watershed may have sediment available for
transport, but if there is no mechanism to remove and transport these sediments away
from it source then sediment delivery and loading will be affected. In this conceptual
model, transport capacity was represented as runoff potential. Areas within the
watershed that has a low runoff potential may exhibit a low transport capacity whereas
zones with high runoff potential has the capacity to remove and transport a large amount
of sediments. For sediment transport to occur there must be both sediment available for
transport and a mechanism for sediment delivery.

In this study, to evaluate the watershed’s potential for sediment transport and
delivery, the following variables were combined in the model: rainfall (R-factor), runoff
potential (Q), LULC (protective and non-protective ground cover) and Geomorphology
(convex and concave slopes). In ArcGIS 9.2 Spatial Analyst the R-factor was reclassed
into four major categories: low, moderate, moderately high and high. These classes
represent the rainfall-erosivity potential as one move from the coastal plains region (low)
to the steep slopes in the headwaters. Runoff potential was also reclassed into 4
categories (low, moderate, moderately high and high). LULC was originally classed and
separated into protective ground covering from non-protective covering. ArcGIS 9.2 Spatial Analyst, Curvature tool was used to describe watershed geomorphology and classify slopes as convex and concave. Using ArcGIS 9.2 Spatial Analyst, Combinational AND tool. These variables were combined providing a variety of scenarios, which describes sediment transport potential within each of the study watersheds. The scenarios were then reclassified into 4 major groups. Sediment transport was defined a having a low, moderate, moderately high or high potential for sediment transport. Categories were represented spatially and areal distributions were calculated. (See details below)

3.5. Characterization and Conceptualization of Watershed Sediment Source Areas

Sediment source areas are identified as areas where non-protective LULC occur on convex slopes. These areas are assumed to be vulnerable to soil erosion therefore, sediment production increases providing a supply of sediments available for transport. To identify potential sediment source areas within the study watersheds, the distribution of non-protective covering on convex slopes was analyzed. Watershed geomorphology was analyzed and slope morphology was used to distinguish convex slopes from concave slopes. LULC was also divided into protective and non-protective covering since the occurrence of latter can enhance sediment production processes. Identifying sediment source areas within each of the study watersheds will help differentiate between those areas that sediment supply-limited from those that are not.

3.5.1. Geomorphology

Geomorphology was included in this conceptual model because of its potential effect on sediment supply within a watershed. In its simplest form, watershed
geomorphology can be described by curvature analysis (the main categories being planform and profile curvature) and slope morphology (the main categories being concave and convex slopes). Slope morphology was used to identify source areas. However, it should be noted that profile curvature relates to slope morphology and acceleration and deceleration of flow. Convex slopes are identified as areas of erosion, while concave slopes are considered depositional areas. Convex slopes or erosional areas in this conceptual framework are considered to be potential sediment source areas, while concave slopes are identified as potential storage areas in the watershed. Convex slopes are areas where water flow is accelerated, initiating erosion and supplying sediments to adjacent areas. When convex slopes dominate the terrain, sediment is then available for transport. However, if concave slopes are prevalent then watershed’s depositional areas are increased thus reducing the availability of sediments for transport consequently, this system become sediment supply-limited.

To analyze landscape geomorphology, a digital elevation model (DEM) for the island of Puerto Rico (DEM) was obtained from the USGS (Table 5.2). Using this DEM, profile curvature maps were created for each watershed. Profile curvature maps were created in ArcGIS 9.2 Spatial Analyst, using the Curvature tool. Note that profile curvatures, which are measures of convexity and concavity, are negative and positive, respectively. The erosional areas (profile convex slopes with a negative value) were extracted from each watershed in this study.

3.5.2. Reclassification of LULC- sediment source areas
Land cover data for the year 2001 for the Commonwealth of Puerto Rico was acquired from NLCD established by the Multi-Resolution Land Characteristics
Consortium (Table 5.2; USGS, 2003). The land cover data distinguishes between 16 classes. An additional nine classes are available in coastal areas (USGS, 2003). Land cover within each watershed was extracted using the watershed vector shapefile as a mask in ArcGIS 9.2. Within the RES watershed, a total of 13 different LULC types were identified. A total of 11 LULC types were identified in the JB watershed. The resulting layer was then reclassified based on established C-factor values (Cox and Madramootoo, 1998 and Jabbar et al., 2005). C-factor values were then further reclassified into two major categories, protective and non-protective. By reclassing LULC into protective and non-protective areas, potential sediment producing areas (non-protective covering) were separated from non-producing areas (protective covering). C-factor ranges from 0 to 1, values close to zero represents areas of maximum ground cover (i.e. dense forest) while values close to 1 had minimal covering (i.e. bare soils). In this study, LULC types that provided minimal ground cover (i.e. agricultural lands, pastures, bare ground) are considered non-protective and are identified as sediment producing zones (sediment supply is not limited). Areas with dense canopies such as forests protect the soil from the erosive impacts of raindrops and are regarded as zones that do not produce sediments (sediment supply is limited).
Table 5.3. Reclassification values for selected land cover within study watersheds

<table>
<thead>
<tr>
<th>Land Cover Type</th>
<th>C-factor Values</th>
<th>Information Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Open Water</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, Open Space</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, Low Intensity</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, Medium Intensity</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Developed, High Intensity</td>
<td>0.0001</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Barren Land</td>
<td>1.0000</td>
<td>Jabbar et al., 2005</td>
</tr>
<tr>
<td>Evergreen Forest</td>
<td>0.001</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Shrub/scrub</td>
<td>0.01</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Herbaceous</td>
<td>0.01</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Cultivated Crops</td>
<td>0.12</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Woody Wetlands</td>
<td>0.001</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Emergent Herbaceous Wetlands</td>
<td>0.001</td>
<td>Cox and Madramootoo, 1998</td>
</tr>
<tr>
<td>Pasture</td>
<td>0.01</td>
<td>Jabbar et al., 2005</td>
</tr>
</tbody>
</table>

3.6. Characterization and Conceptualization of Watershed Hydrologic Response and Sediment Transport Capacity

Evaluating connectivity in terms of sediment production, transport and delivery within a watershed is critical to minimize effects of sediment yield and loading in aquatic system. This determines the ability of sediments to be transported between compartments within the landscape therefore, conceptualizing relationships between geomorphology, soils, land use and hydrologic response is the first step in this framework. With the advancement of computing and geospatial technologies, the potential for making connections between compartments and related complex processes such as soil erosion and sediment delivery has also increased. To characterize and conceptualize watershed hydrologic response with regards to sediment transport, soil hydrologic groups (HSG) was mapped for each watershed in this study. Curve numbers (CN) were assigned to LULC classes within each of the study watersheds. Storage capacity (S) and soil runoff potential (Q) was also computed and displayed spatially.
Runoff is assumed to be the primary mechanism by which sediment is transported through the study watersheds. Computing runoff potential within each of the watershed will identify areas and help differentiate those areas that are transport-limited from those that are not.

### 3.6.1 Hydrologic Soil Group (HSG)

HSG describes the infiltration capacity of a particular type of soil. The infiltration capacity of soils is important because it influences where and how runoff is generated in a particular area. In general, soils with a high infiltration rate have a low runoff potential, while soils with low infiltration rates have a high runoff potential (NRCS, 1986). Table 5.4 provides a brief description of the various hydrologic groups. Soils with high runoff potential (low infiltration rates) such as groups C and D may generate runoff via surface overland flow or saturated overland flow. Both processes in this case can transport detached soil particles from one zone in the watershed to the next, hence these soil groups do not promote a transport-limited system.

Soil mapping units within each of the watershed was classified into one of the four hydrologic groups. Individual vector layers showing the spatial distribution of HSG was generated for each of the study watersheds. These vector layers were then converted into a raster layer of 30 m resolution. The HSG layers were then used to assign curve numbers to the various LULC types within the study watersheds. The assignment of curve numbers is a critical step in the computation of potential discharge for the study watersheds.
Table 5.4. Hydrologic soil groups and relevant properties

<table>
<thead>
<tr>
<th>HSG</th>
<th>Soil Texture</th>
<th>Water Transmission (inch/hr)</th>
<th>Infiltration Class</th>
<th>Runoff Potential</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>Sand, loamy sand, or sandy loam</td>
<td>&gt; 0.30</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>B</td>
<td>Silt loam or loam</td>
<td>0.15-0.30</td>
<td>Moderate</td>
<td>Moderate</td>
</tr>
<tr>
<td>C</td>
<td>Sandy clay loam</td>
<td>0.05-0.15</td>
<td>Low</td>
<td>Moderately High</td>
</tr>
<tr>
<td>D</td>
<td>Clay</td>
<td>0.0.05</td>
<td>Very Low</td>
<td>High</td>
</tr>
</tbody>
</table>

* Data from table was adapted from information provided by (NRCS, 1986)

3.6.2 Curve Number (CN) and Storage Capacity (S)

The original Soil Conservation Service –Curve Number (CN) method was developed for the purpose of modeling rainfall-runoff from ungauged watersheds (Lyons et al., 2004). CN is related to LULC and HSG. In assigning a CN to an area, the user is describing the effect of LULC on runoff for a given soil type. Generally, dense forested areas will have a low CN while urban areas will have very high CN. In the context of sediment transport capacity, a forested system, with a low CN value may limit sediment transport whereas in an urban system or even on some agricultural lands, sediment transport is not limited. A forested system with a low CN value represents other hydrologic processes such as: stem flow, evapo-transpiration and interception, which indirectly influence runoff potential. These processes work together reducing the amount and intensity of precipitation reaching the forest floor. Runoff is typically not observed in forest systems hence, sediment transport is limited. On the other hand, an agricultural system may be assigned a high CN due to a lack of protective covering over compacted soils. As a result, a greater amount of precipitation intercepts the soil surface generating runoff.
To evaluate the rainfall-runoff in the study watersheds, CNs were assigned to the various land use types in each watershed. This was accomplished in ArcGIS 9.2 using Spatial Analyst. The HSG raster layers generated previously were combined with the LULC rasters to determine the HSG associated with a particular land use type. Once this was completed, CN was assigned to this combine land use/HSG layer based on established CN values. Lower CN values suggest a lower runoff potential.

**Table 5.5. Curve Number and LULC relationship within RES watershed**

<table>
<thead>
<tr>
<th>Land use/ Land cover</th>
<th>Hydrologic Group</th>
<th>Curve Number</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Developed-open spaces</td>
<td>A, B, C, D</td>
<td>39, 61, 74, 80</td>
</tr>
<tr>
<td>Developed-low intensity</td>
<td>A, B, C, D</td>
<td>49, 69, 79, 84</td>
</tr>
<tr>
<td>Developed -Medium intensity</td>
<td>A, B, C, D</td>
<td>49, 69, 79, 84</td>
</tr>
<tr>
<td>Developed -High Intensity</td>
<td>C, D</td>
<td>86, 89</td>
</tr>
<tr>
<td>Barren Lands</td>
<td>A, C</td>
<td>77, 91</td>
</tr>
<tr>
<td>Evergreen Forest</td>
<td>A, B, C, D</td>
<td>30, 55, 70, 77</td>
</tr>
<tr>
<td>Shrub/Scrub Vegetation</td>
<td>C, D</td>
<td>70, 77</td>
</tr>
<tr>
<td>Herbaceous</td>
<td>A, B, C, D</td>
<td>30, 71, 78</td>
</tr>
<tr>
<td>Cultivated Crops</td>
<td>A, C, D</td>
<td>69, 84, 79</td>
</tr>
<tr>
<td>Hay/ Pasture</td>
<td>A, C, D</td>
<td>43, 77, 82</td>
</tr>
<tr>
<td>Woody Wetlands</td>
<td>A, B, C, D</td>
<td>36, 60, 73, 79</td>
</tr>
<tr>
<td>Emergent Herbaceous Wetlands</td>
<td>C, D</td>
<td>72, 79</td>
</tr>
</tbody>
</table>
Table 5.6. Curve Number and LULC relationship within JB watershed

<table>
<thead>
<tr>
<th>Land use/ Land cover</th>
<th>Hydrologic Group</th>
<th>Curve Number</th>
</tr>
</thead>
<tbody>
<tr>
<td>Developed-open spaces</td>
<td>A, B, C, D</td>
<td>39, 61, 74, 80</td>
</tr>
<tr>
<td>Developed-low intensity</td>
<td>A, B, C, D</td>
<td>49, 69, 79, 89</td>
</tr>
<tr>
<td>Developed -Medium intensity</td>
<td>A, B, C, D</td>
<td>49, 69, 79, 84</td>
</tr>
<tr>
<td>Developed -High Intensity Barren Lands</td>
<td>A, B, D</td>
<td>68, 79, 89</td>
</tr>
<tr>
<td>Evergreen Forest</td>
<td>A, B, C, D</td>
<td>77, 86, 91, 94</td>
</tr>
<tr>
<td>Shrub/Scrub Vegetation</td>
<td>A, B, C, D</td>
<td>30, 55, 70, 77</td>
</tr>
<tr>
<td>Herbaceous</td>
<td>A, B, D</td>
<td>35, 56, 70, 77</td>
</tr>
<tr>
<td>Cultivated Crops</td>
<td>A, B, C, D</td>
<td>30, 58, 71, 78</td>
</tr>
<tr>
<td>Woody Wetlands</td>
<td>A, B, D</td>
<td>36, 60, 79</td>
</tr>
<tr>
<td>Emergent Herbaceous Wetlands</td>
<td>A, D</td>
<td>32, 79</td>
</tr>
</tbody>
</table>

The assigned CN values were used to determine the storage capacity ($S$) for soils within watersheds. The storage capacity was computed in ArcGIS 9.2 Spatial Analyst raster calculator. The major input layer being the derived CN layer created previously. The storage capacity is found using the following equation:

$$S = \frac{25400}{CN} - 254$$  \hspace{1cm} \text{Equation 5.1}

Where:

$S$ = storage capacity (mm)

$CN$ = Curve number

3.3.3 Runoff potential ($Q$)

The determination of discharge potential ($Q$) was completed using the following equation:

$$Q = \frac{(P - 0.2S)^2}{(P + 0.8S)}$$  \hspace{1cm} \text{Equation 5.2}

Where:

$Q$ = runoff (mm)

$P$ = precipitation (mm)

$S$ = Storage capacity and relates to antecedent moisture conditions of the soil (mm)
A 10-year frequency, 24-hour duration storm (P) was used in the calculation of Q for each watershed. A precipitation event with a 10-year recurrence interval estimates the probability for a storm of this magnitude occurring in any given year (10% chance a storm of this magnitude will occur in a given year). Precipitation values for a 10-year frequency, 24-hour duration storm was obtained from the National Oceanographic and Atmospheric Administration, National Weather Service (south-eastern region) was acquired for watershed. The value of P varied between the two watersheds. Estimated rainfall for this magnitude storm were 262 mm and 220 mm for RES and JB watersheds, respectively (NWS-NOAA, 2010). The runoff potential estimated for watersheds in this study represents the combined effect of soil type and LULC both of which are important factors in runoff generation. The value computed and the map generated from this analysis identifies those areas in the landscape that are transport-limited as well as those that are not, which is critical to sediment delivery.

4.0. Results and Discussion

In chapter 2, the RUSLE model was used to estimate soil loss within the study watersheds. Results from this model suggested that soil loss within the two watersheds was relatively low with very sporadic incidences of erosional “hotspots”. Besides this, no additional information could be obtained using RUSLE, regarding the identification of factors having the greatest influence on soil loss. In addition, it was impossible to identify potential sediment source areas, determine sediment availability and transport capacity of sediments within the watershed. To link sediment soil loss and sediment delivery to aquatic systems, identifying sediment source areas and determining sediment availability and its transport capacity is imperative before undertaking management
plans. To gather more information on the processes affecting soil loss, sediment
availability and sediment transport capacity a conceptual model was developed for the
two study watersheds. In this conceptual model, terrain complexity
(geomorphology/soils), runoff potential, LULC and rainfall were included in the
prediction of sediment availability and transport within the watersheds.

4.1. Characterization and Conceptualization of Watershed Sediment Source Areas

4.1.1. Slope morphology

Slope analysis for RES indicates that the amount of concave (depositional) slopes
and convex (erosional) slopes are evenly distributed in the landscape (Table 5.7; Figure
5.7). In the context of assessing the ability of the watershed to generate (supply)
sediment, terrain analysis shows that the RES watershed has the potential to supply a
large amount of sediments available for transport. Additionally, the “equal” prevalence
of concave slopes in the landscape also suggests that the watershed has the capacity to
store sediments. In the RES watershed, the configuration of convex and concave slopes
in the landscape will be an important factor influencing the ability of runoff to transport
available sediments through the landscape.

Less than 30% of the JB watershed is comprised of convex slopes therefore one-
third of this watershed are potential sediment source areas. JB has a predominance of flat
areas or an extensive coastal plains area, which potentially serve as major storage area in
this watershed (Table 5.7; Figure 5.8). The implications of these results are that JB may
be able to buffer itself against accelerated soil erosion. Comparing both watersheds,
sediments eroded from the landscape may be redistributed within various compartments
in these landscapes with very little sediment actually reaching a stream channel.
Assuming that landform (geomorphology) does not change significantly over short time periods, it is suggested that other external factors may play a significant role in controlling soil erosion and the transport of sediment through watersheds.

Table 5.7. Distribution of concave and convex slopes

<table>
<thead>
<tr>
<th>Curvature</th>
<th>Area Distribution (km$^2$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>RES</td>
</tr>
<tr>
<td>Erosion (Convex)</td>
<td>32.0</td>
</tr>
<tr>
<td>Flat</td>
<td>9.0</td>
</tr>
<tr>
<td>Depositional (Concave)</td>
<td>34.0</td>
</tr>
<tr>
<td></td>
<td><strong>75</strong></td>
</tr>
</tbody>
</table>

Figure 5.7. Slope morphology map for RES watershed.
4.1.2. LULC and sediment source areas

LULC was reclassed into areas with protective ground covering and non-protective ground covering. Based on the estimated distributions, a large portion (79.6%) of the RES watershed is protected (Table 5.8). In the JB watershed, over 63% of the total areas are categorized as non-protective (Table 5.8). The difference between the two watersheds may be due to the fact that the headwaters of RES are part of the federally protected forest system (Caribbean National Forest). Also, the steep terrain in the watershed, does not allow for establishment of extensive residential communities, which can reduce natural vegetation. Jobos Bay does not have the constraints of having a protected headwater system. However, the difference in vegetation types due to differences in climate may explain distribution of protective and non-protective ground cover in each of the watershed. In addition, the JB watershed has a greater distribution of agricultural lands as compared to RES. Given the various distribution between protective
and non-protective land cover in the watersheds it may be assumed that a large part of the RES watershed may be sediment supply-limited. In contrast, the JB watershed may not be supply-limited. Figures 5.9-5.10 show maps of protective and non-protective areas in RES and JB, respectively.

Table 5.8. Distribution of protective and non-protective covering in RES and JB bay watershed

<table>
<thead>
<tr>
<th>LULC Classification</th>
<th>RES</th>
<th>JB</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>%</td>
<td>km²</td>
</tr>
<tr>
<td>Protective</td>
<td>79.6</td>
<td>59.7</td>
</tr>
<tr>
<td>Non-Protective</td>
<td>20.4</td>
<td>15.3</td>
</tr>
<tr>
<td>%</td>
<td>100</td>
<td>75</td>
</tr>
</tbody>
</table>

Figure 5.9. LULC separated into protective and non-protective zones in RES watershed.
4.1.3. Identification of Potential Sediment Source Areas

To identify potential sediment source areas LULC was combined with convex (erosional slopes). Figures 5.11 and 5.12 are maps showing potential source areas of sediment (supply not-limited). On the RES map, most of the convex slopes are classed as supply limited (green colored). Potential sediment source areas (red areas), are areas where sediment supply may not be limited (Figure 5.11). In JB, sediment sources areas (red areas) have a higher spatial coverage when compare to sediment supply limited areas within this watershed (Figure 5.12). This implies that most slopes within the JB watershed act as sediment source areas, which can increase sediments available for transport.

Figure 5.10. LULC separated into protective and non-protective zones in JB watershed.
Figure 5.11. Potential sediment source areas within the RES watershed.

Figure 5.12. Potential sediment source areas within JB watershed.
4.2. Characterization and Conceptualization of Watershed Hydrologic Response and Sediment Transport Capacity

4.2.1. Hydrologic Soil Group (HSG)

Table 5.9 reports the percent and areal distribution of the individual soil groups within each of the study watersheds. The dominant HSG within RES is group C (57%). Soils within this hydrogroup have low infiltration rates with moderately high runoff potential. In the JB watershed, most of the soils were categorized as group D. This hydrogroup comprise approximately 65% of the JB watershed. In general, soils that are in hydrogroup D tend to have low infiltration rates, which results in a high runoff potential. Based on these distributions, the hydrologic response of most soil types within RES and JB does not promote sediment transported-limited systems. Figures 5.13 and 5.14 are maps of HSG in RES and JB, respectively. A visual assessment of the figure 5.13 shows the hydrogroup C occupies a large portion of the headwaters in RES watershed, whereas group D tend to be prevalent in the coastal plains. The locations of sediment cores collected from the adjacent riverine-estuarine system in RES were added to the map (Figure 5.13). It is clear from this map that sampling sites are surrounded by soils with moderately high to high runoff potentials. The implications of these results are that the surrounding areas may have the potential to transport material (via runoff) from adjacent land areas to the riverine/estuarine system. Figure 5.14 is a map showing the occurrence of the various hydrogroups within the JB watershed. In the case of JB watershed, it is very easy to see that hydrogroup D is the group with the highest spatial coverage within this watershed. This group is present both at high elevations as well as on the extensive coastal plains. Figure 5.14 shows location of sediment cores for the JB watershed. The majority of cores for this study were collected in the bay with the
exception of two cores. These two cores were collected within the Jobos Bay watershed (Figure 5.14). Both cores were collected adjacent to areas that have a high runoff potential (group D).

**Table 5.9.** Distribution of HSG within each of the study watersheds.

<table>
<thead>
<tr>
<th>Hydrogroup</th>
<th>RES %</th>
<th>RES km²</th>
<th>JB %</th>
<th>JB km²</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>5.2</td>
<td>3.9</td>
<td>3.4</td>
<td>5.7</td>
</tr>
<tr>
<td>B</td>
<td>4.5</td>
<td>3.4</td>
<td>23.8</td>
<td>39.7</td>
</tr>
<tr>
<td>C</td>
<td>57.2</td>
<td>42.9</td>
<td>6.3</td>
<td>10.5</td>
</tr>
<tr>
<td>D</td>
<td>32.5</td>
<td>24.4</td>
<td>65.2</td>
<td>108.9</td>
</tr>
<tr>
<td>N/A</td>
<td>0.7</td>
<td>0.5</td>
<td>1.4</td>
<td>2.3</td>
</tr>
</tbody>
</table>

**Figure 5.13.** Spatial representation of HSG for RES watershed.
4.2.2. Curve Number (CN) and Storage Capacity(S)

Curve number, as mentioned before relates to soil type and LULC types. In this study CN, was assigned to LULC types identified in each of the study watersheds. There was no distinct pattern in CN occurrence within both watersheds (Figures 5.15 and 5.16). The uniqueness of this pattern may simply be a result of the combined effect of LULC and hydrologic group. In tables 5.5 and 5.6, all LULC types occurred on more than one hydrogroup and as a result have more than one CN values attributed to it. Using the location of sediment cores within the RES watershed as points of references, it is apparent that soils surrounding each core possess various curve number values (Figure 5.15). A visual assessment of CN values within the JB watershed shows that soils along the shoreline also has a variety of CN values, all of which may contributes to ability of sediments to be transported to the bay from the associated watershed (Figure 5.16).
CN values were then used to compute the storage capacity (S) throughout each watershed. The unit of measure for storage capacity is mm. High S values indicate greater storage potential of precipitation prior to runoff generation. Figures 5.17 and 5.18 are maps showing the storage capacity in each of the watersheds. The range of S values for RES watershed is 25 mm to 592 mm. In the JB watershed, values for S range from 16 mm to 592 mm. In the headwater region of RES watershed, the estimated S value is approximately 108 mm. This value is relatively high and demonstrates the role of vegetation in influencing the sediment transport capacity in the watershed. The headwaters of RES are covered by dense tropical forest, which may increase the infiltration rates of soils (root system) allowing for increased storage of precipitation in this portion of the watershed (Figure 5.17). As a result, the sediment transport capacity in headwaters of this watershed may be limited. In the coastal plains region of this watershed, the areas around the sampling sites, possess relatively low S values (Figure 5.17). This suggests that the areas surrounding these sites is unable to store large amounts of precipitation therefore, the runoff of potential may be moderately high to high. Within the headwaters of JB watershed at the highest elevations, these areas have an estimated S value of approximately 75 mm. Compared to S value estimated for the headwaters of RES watershed, JB has a smaller storage capacity (Figure 5.18). It may be assumed from this observation that vegetation and soils in this region (JB watershed) are less effective at retaining (storing) precipitation. As a result, runoff may be initiated faster in the JB watershed when compared to RES. In the coastal plain region of the JB watershed, S values are very diverse; this diversity may influence the sediment transport capacity and sediment delivery to the bay. Diversity of S values in this region may
influence hydrologic connectivity. For example, if an area has a low S value then runoff potential is high. If there is another zone with a higher S value (a lower runoff potential) adjacent this area with lower S value then the transport of sediments through the watershed may be affected. Having these two zones located close to each other may result in a transport-limited system.

Figure 5.15. Spatial representation of curve number (CN) for RES watershed.
Figure 5.16. Spatial representation of curve number (CN) for JB watershed.

Figure 5.17. Spatial representation of storage capacity (S) for RES watershed.
4.2.3. Runoff potential ($Q$)

Runoff potential was computed in ArcGIS 9.2 Spatial Analyst Raster Calculator. Table 5.10 reports the percent and areal distributions of runoff potential in each watershed based on the established classes (low, moderate, moderately high and high). Over 50% of the RES watershed has a moderate runoff potential. In the JB watershed, approximately 63% of its total area has a moderately high runoff potential. The results reported in this table suggest that for most areas within each of the watershed, sediment transport capacity may not be limited. Figures 5.19 to 5.20 are maps of runoff potential within each watershed. Figure 5.19, is a map of runoff potential for RES watershed, in the headwaters most of the areas have a moderate runoff potential. These areas flow into a coastal plains region that experience moderately high to high runoff potential. The implication of this observation is that hydrologic connectivity may be efficient in this
watershed, which can promote sediment transport through the watershed. With reference to JB, the majority of watershed has a moderately high runoff potential (Figure 5.20). Similar to the RES watershed, hydrologic connectivity between zones within JB watershed may not impede sediment transport from one area to the next hence, sediment transport may not be a limiting factor in sediment transport from watershed and delivery to the adjacent bay.

Table 5.10. Runoff potential categories and their spatial distribution within each watershed.

<table>
<thead>
<tr>
<th>Runoff Potential</th>
<th>RES</th>
<th>JB</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>%</td>
<td>km²</td>
</tr>
<tr>
<td>Low</td>
<td>5.2</td>
<td>3.9</td>
</tr>
<tr>
<td>Moderate</td>
<td>56.5</td>
<td>42.4</td>
</tr>
<tr>
<td>Moderately High</td>
<td>27.0</td>
<td>20.3</td>
</tr>
<tr>
<td>High</td>
<td>8.2</td>
<td>6.1</td>
</tr>
<tr>
<td>N/A</td>
<td>3.0</td>
<td>2.3</td>
</tr>
<tr>
<td></td>
<td>100</td>
<td>75</td>
</tr>
</tbody>
</table>
Figure 5.19. Spatial representation of runoff potential (Q) for RES watershed

Figure 5.20. Spatial representation of runoff potential (Q) for JB watershed
4.3. Sediment Availability and Transport

A sediment supply must be available for transport in order for there to be sediment delivery to aquatic ecosystems. Based on the results from analysis of watershed geomorphology, LULC and identification of source areas, a large portion of RES watershed may be sediment supply-limited. The JB watershed may be not be considered a supply-limited watershed. The results from computing runoff potential in each watershed suggest that none of these watersheds are transport-limited. However, to truly map the sediment transport capacity in each of the watersheds, the estimated sediment availability and predicted transport capacity were combined. The result is a map for each watershed showing the sediment transport capacity of various zones in each watershed. Sediment transport capacity as defined by the conceptual model is the relationship between sediment availability and runoff potential within the landscape. Sediment transport capacity can also be defined as the soil loss potential from a watershed to aquatic systems. It may be regarded as such because the term loss refers to some level of deprivation. Figures 5.21 and 5.22 are maps of sediment transport capacity (soil loss potential) in RES and JB, respectively.

Within the RES watershed, of the slopes considered to be erosion prone (convex slopes), approximately 73% were classified as moderate and less than 20% were predicted to experience moderately high soil loss (Table 5.10). None of the areas within the watershed was estimated to have a high soil loss potential (sediment transport capacity). Taking into consideration the results from the previous analysis, it would appear that despite potential limitation in sediment supply, the predicted runoff potential for this watershed has the capacity to transport sediment out of the watershed. Areas of
moderate and moderately high soil loss potential are clearly visible on the map (Figure 5.21).

In the JB watershed, of the slopes considered to be erosion prone, almost 62% of these areas were estimated as having a moderately high soil loss potential (Table 5.11; Figure 5.22). A fairly significant amount of those slopes may also experience moderate (28%) soil loss potential (Table 5.11; Figure 5.22). For this watershed, neither sediment supply or transport was considered limiting however, only a small portion (4%) of the watershed has a high soil loss potential. JB, runoff potential may have a great influence on sediment transport capacity in this watershed.

Table 5.11. Sediment available for transport from watershed to aquatic systems in the RES watershed.

<table>
<thead>
<tr>
<th>Sediment Available for transport</th>
<th>Areal Distribution (%)</th>
<th>Areal Distribution (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>8.3</td>
<td>2.7</td>
</tr>
<tr>
<td>Moderate</td>
<td>72.5</td>
<td>23.2</td>
</tr>
<tr>
<td>Moderately High</td>
<td>19.2</td>
<td>6.1</td>
</tr>
<tr>
<td>High</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>100</strong></td>
<td><strong>32</strong></td>
</tr>
</tbody>
</table>
Figure 5.21. Sediment available for transport from watershed to aquatic systems in RES watershed.

Table 5.12. Sediment available for transport from watershed to aquatic systems in the JB watershed

<table>
<thead>
<tr>
<th>Sediment available for Transport</th>
<th>Areal Distribution (%)</th>
<th>Areal Distribution (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low</td>
<td>6.1</td>
<td>3.0</td>
</tr>
<tr>
<td>Moderate</td>
<td>28.3</td>
<td>13.9</td>
</tr>
<tr>
<td>Moderately High</td>
<td>61.6</td>
<td>30.2</td>
</tr>
<tr>
<td>High</td>
<td>4.0</td>
<td>2.0</td>
</tr>
<tr>
<td></td>
<td><strong>100</strong></td>
<td><strong>49</strong></td>
</tr>
</tbody>
</table>
Figure 5.22. Sediment available for transport from watersheds to aquatic systems in JB watershed.

The differences observed in this watershed when compared to RES may be explained by the fact that the JB watershed is located in southern Puerto Rico, where the microclimate is drier due to a rain shadow effect. Less rain dries out the soil, which makes it more vulnerable to being eroded and transported from its point of origin. Also, in the JB watershed more of its land is used for agricultural and farming purposes (pasture) when compared to RES. It has been established by many studies that certain land use practices such as agriculture has the potential for soil erosion thus acting as a significant source of sediments to aquatic systems (Pimentel and Kounang, 1998). In both watersheds, the dominance of neutral areas or depositional/flat areas comprise the majority of each watershed (RES = 43 km$^2$ and JB = 118 km$^2$), hence sediment transport and delivery of sediments to adjacent stream networks may be influenced by the spatial configuration (connectivity) of hillslopes (specifically convex slopes) and channels.
4.4. Sediment Delivery: Making the Connection

Conceptualization of sediment delivery to downstream ecosystems is often represented as a simple process (Phillips, 1991). From the initial stages of sediment production to sediment transport and eventually sediment delivery is complex. This research attempted to establish a link between potential soil loss from the source areas in the landscape, transport and sediment delivery to RES riverine-estuarine system and consequent sediment deposition. As mentioned in Chapter 3, four sediment cores were collected from the RES riverine-estuarine system (Figure 3.2). The locations of these cores were then overlaid on the watershed soil loss potential maps for this study watershed (Figure 5.21). Based on the sediment composition data obtained for each core and presented in table 5.12, cores in the upper portions of the estuaries (ESC1 and ESC3) had higher amounts of clay and silt-sized particles when compared to cores located at the mouth of the estuaries (ESC2 and CCB4). In light of this data, it may be assumed that cores located in the upper reaches (ESC1 and ESC3) of the estuary may be influenced mostly by upslope LULC and adjacent to sampling sites.

Furthermore, the $^{137}$Cs inventories for all of the sites were compared (Table 5.13). ESC1, the sediment core collected furthest upstream had the highest $^{137}$Cs (dpm cm$^{-2}$) and excess $^{210}$Pb (dpm cm$^{-2}$) inventories when compared to other sample cores. A major source of excess $^{210}$Pb in the environment comes from weather rock material in the terrestrial landscape. A One-way ANOVA was performed to determine the effects of location on $^{137}$Cs distributions within the RES watershed. The results of statistical analysis indicated that location had an effect on $^{137}$Cs distributions measured in all cores. The analysis was significant at a level of p <0.0001 ($r = 0.26$). Given that after the 1960s nuclear weapons testing in the atmosphere were banned, the supply to $^{137}$Cs to the
atmosphere had ceased. It may be assumed that the watershed may be a major source of 
$^{137}\text{Cs}$ delivered to the estuary therefore, it is expected that the sediment core collected the 
highest upstream will have the greatest $^{137}\text{Cs}$ distribution and in Zaborska et al. 2010 
observed that distance from radionuclide source areas in the north-western Barents Sea 
had an influence on its distribution. ESC1 was collected furthest upstream and is located 
along the main stem of the river. To support the point, of location being a potential 
influence on radionuclide distribution; in figure 5.13, ESC1 is surrounded by areas with a 
low to moderately high supply of sediments available for transport. Even if those areas 
with low amounts of sediments available for transport and do not supply the channel with 
a lot of sediment, other adjacent areas may supply sediments to the channel. Therefore, 
the sedimentological characteristics and radionuclide inventories may be reflective of the 
inputs from watershed. Figures 5.23 and 5.24 are maps showing the $^{137}\text{Cs}$ inventories 
overlaying sediment available for transport map as well as an Orthophoto. These maps 
offer some possible support for the suggestion that watershed may serve as a source of 
sediment ($^{137}\text{Cs}$) to this riverine-estuarine system.
Figure 5.23. Map showing areas with sediment available for transport overlaid by $^{137}$Cs inventories estimated from sediment cores in RES watershed.

Figure 5.24. Orthographic map overlaid by $^{137}$Cs inventories estimated from sediment cores in RES watershed.
Table 5.13. Radionuclide inventories and sedimentological characteristics for sediment cores collected from RES watershed

<table>
<thead>
<tr>
<th>Sediment Core</th>
<th>$^{137}$Cs Inventories (dpm cm$^{-2}$)</th>
<th>$^{210}$Pb Inventories (dpm cm$^{-2}$)</th>
<th>Clay (%)</th>
<th>Silt (%)</th>
<th>LFS (%)</th>
<th>O.M. (%)</th>
<th>Carbonates (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Upstream</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>ESC1</td>
<td>7.98</td>
<td>99.4</td>
<td>21.7</td>
<td>51.1</td>
<td>27.1</td>
<td>14.0</td>
<td>2.6</td>
</tr>
<tr>
<td>ESC3</td>
<td>7.73</td>
<td>34.9</td>
<td>18.9</td>
<td>61.0</td>
<td>20.1</td>
<td>15.8</td>
<td>4.3</td>
</tr>
<tr>
<td>ESC2</td>
<td>1.22</td>
<td>2.59</td>
<td>8.7</td>
<td>49.7</td>
<td>41.6</td>
<td>48.7</td>
<td>7.8</td>
</tr>
<tr>
<td><strong>Downstream</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CCB4</td>
<td>6.86</td>
<td>72.3</td>
<td>15.8</td>
<td>48.7</td>
<td>35.8</td>
<td>20.5</td>
<td>5.2</td>
</tr>
</tbody>
</table>
To extend this discussion further, river discharge (cms) and total dissolved solids (tons/day) was obtained from USGS gaging station 50063800 located in the municipality of Rio Grande, Puerto Rico. This was the only gage within this watershed with “extensive” data for total dissolved solids and river discharge. Precipitation data was acquired from NOAA-NWS South-eastern region (NOAA-NWS, 2010). The precipitation data used in this analysis are 30-year monthly averages for this area. This raingage is located in Rio Espiritu Santo watershed (Figure 5.13).

The average daily load for the RES watershed is 1.66 tons/day. This value was estimated from measured TDS loading from a period of 1967 to 2004. Figures 5.25-5.27 are plots of discharge, precipitation and total dissolved solids. From these plots we see that both discharge and total dissolved solids follow precipitation. Also, discharge and total dissolved solids appear to follow a trend of increased discharge increases the amount of solids transported with it. It may be assumed that precipitation is a major factor influencing discharge and total dissolved solids. Considering these trends, the question remains: how does this effect sediment delivery in this watershed? To answer this question, all of the $^{137}$Cs distribution from the sediment cores was plotted on a single graph to determine any trends (Figure 5.28). Three major pulses of $^{137}$Cs were detected in all of the cores. Since $^{137}$Cs is transported via adsorption to sediment particles, it has the potential to act as a tracer. It may be assumed that these pulses of $^{137}$Cs may indicate fluxes of sediment being supplied to these areas possibly during flooding events. In general, these pulses of $^{137}$Cs through out all cores indicates that there is sediment available for transport and that the system is not transport-limited, hence there is sediment delivery to the river and sediment loading occurring in the estuary. Since, the
profiles for both $^{137}$Cs and excess $^{210}$Pb were irregular in all cores for the RES watershed, it was not possible to acquire sediment age and sedimentation rates for the cores. Consequently, it was impossible to acquire solid dates to verify that pulses observed in the $^{137}$Cs profiles are a result of flooding events.

**Figure 5.25.** Shows trend between discharge and total dissolved solids from gauge located within the RES watershed.
Figure 5.26. Shows trend between precipitation and total dissolved solids from gauge located within the RES watershed.

Figure 5.27. Shows trend between precipitation and discharge from gauge located within the RES watershed.
Figure 5.28. $^{137}$Cs pulses in all sediment cores collected from RES watershed.

In addition, this gage is not located on the main channel of this river, hence discharge and sediment delivery to the associated estuary may be underestimated (Figure 5.21). Due to data limitations for JB, it was not possible to make such a connection for JB watershed. However, averaged $^{137}$Cs data in surface sediments for sediment cores and surface grab samples were overlaid on the sediment available for transport map (Figure 5.29). From the map, it can be observed that averaged $^{137}$Cs distribution for surface samples for sediment cores JB1, PJ1, SFM1 and ASM1 have higher distribution of $^{137}$Cs when compared to the grab samples collected in the inner middle of the bay. It is possible that sediment transported from the watershed (possibly with $^{137}$Cs adsorbed unto the surface) may be deposited along the coastline of the bay hence the higher values
measured at these sites. Figure 5.30 is a orthographic map showing the $^{137}$Cs distribution in surface samples within Jobos Bay.

**Figure 5.29.** Averaged $^{137}$Cs distributions for surfaces samples of sediment cores and surface grab samples within Jobos Bay overlaying layer highlighting sediment available for transport.
Figure 5.30. Orthographic map with averaged $^{137}$Cs distributions for surfaces samples of sediment cores and surface grab samples within Jobos Bay.

5.0. Limitations

As stated previously, linking sediment availability, transport from watersheds to sediment delivery and eventually sediment deposition is very challenging. The challenge stems from the fact that the processes being modeled vary in time and space. Additionally, obtaining the data necessary to validate this method in particular was a major short coming in this research. For instance, to validate the assumption that the watershed may be serving as a potential source area of sediment and $^{137}$Cs to the associated estuary, soil samples acquired from those areas with moderate to moderately high sediment available for transport was necessary. The soil samples would have been analyzed to determine bulk inventories of $^{137}$Cs, mineralogy and geochemical composition. A lower inventory of $^{137}$Cs in the watershed indicates the occurrence of soil erosion. The mineralogical and geochemical composition of the soil samples when
compared to those of sediment samples would have provided sediment fingerprints allowing for linking soils in watershed to sediment deposited in the estuary. Funding constraints did not allow for geochemical analysis and re-sampling of sites within the watershed.

Another major limitation of the study was not being able to obtain data from gaging stations along the major channel of the Rio Espiritu Santo river system therefore, extensive and more accurate discharge, total dissolved solids and precipitation was not obtained for this site. The distribution of radionuclides in the sediments proved to be an unreliable tool for dating sediments and determining sedimentation rates within these dynamic systems. Sediment ages would allow for establishing stronger links between sediment delivery and sediment depositions in estuaries. The key to linking watershed processes to sediment deposition rates in estuaries the following information must be available: (1) quantity of soil loss, (2) quantity of soil delivered and (3) quantity of soil deposited. If any of these links are missing or compromised, making the connection between soil loss to sediment delivery and sediment deposition will remain a challenge and an enigma.

6.0. Conclusion and Future Directions

Through analysis of geomorphology and LULC with consideration of the hydrologic principles, potential sediment source areas were identified in watersheds. Watershed connectivity was assessed through slope analysis, vegetation patterns and analysis of runoff potential. Results from the conceptual model suggest that the JB watershed is neither transport-limited nor sediment supply-limited. Analysis also suggested that RES is potentially supply-limited, but not necessarily transport-limited.
The vertical $^{137}\text{Cs}$ distribution suggests that flux of sediment to the estuary is event driven. As a result, the primary routes by which sediment leaving the watershed may enter stream channels may also be identified, which can aid in soil conservation efforts. It is believed that both conceptual and process-based models can be constructed using this framework. Such a framework was developed to linking landscape processes to sediment dynamics in estuaries. This framework can also be applied to policy development and implementation. To strengthen the implied link made to soil and sediment delivery in watersheds, flow paths within in the watersheds will be mapped in order to potential sediment routing areas in the landscape. Longer sediment cores (> 60 cm) will also be collected from study areas. In addition, soil samples will be collected from potential source areas of sediments in the watershed and along flow paths. Mineralogical and geochemical analyses will be performed on both soil and sediment samples to establish sediment fingerprints. These fingerprints can also serve to identify erosion hotspots within the watershed. In addition, this method will be applied to other estuaries to test its applicability in other regions.
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Appendix A: Loss On Ignition Protocol (LOI)

Loss on ignition (LOI) is a cost efficient and reliable way to determine percent organic matter (% O.M.) and soil organic carbon (SOC) (Konen et al., 2002). In addition to advantages stated before, using LOI to determine organic matter does not required specialized equipment and skills or is neither time consuming (Beaudoin, 2003). In this study, LOI procedure was developed based on Dean (1974) and Heiri et al. (2001). In order to conserve sediments for geochemical analysis, LOI was performed on every other sample within a core. Specifically, every odd numbered sample was used in this analysis. LOI was also performed on surface sediment and watershed soil samples.

In preparation for sample LOI analysis, previously freeze dried samples sediment and soil samples were finely ground and homogenized by stirring with spatula or shaken in a closed container. Approximately, 1 gram of homogenized sediment was placed in a pre-weighed crucible at oven dried at 105 degrees Celsius for a time period between 12-24 hrs to extract moisture from samples. Upon leaving the oven, samples were cooled at room temperature in desiccators. After cooling, samples were weighed again to determine dry weight for sediment or soil ($DW_{105}$) in grams. For the determination of % O.M. in samples, samples were placed in a furnace at 550 °C for 4 hrs. Samples were once again cooled in desiccators and weight loss was determined at 550 °C ($DW_{550}$). Computation of % O.M. was completed using equation below in the Microsoft Excel.

Total carbonate content was estimated by returning soil and sediment samples to furnace at 950 °C for 1.5 hrs. Samples were cooled in dessicators and % Carbonate content was determined using equation below.
% O.M. = \left( \frac{DW_{105} - DW_{550}}{DW_{105}} \right) \times 100 \quad \text{(Equation A1)}

Where:

\( DW_{105} \): Dry weight of samples after drying in oven at 105 °C for 12-24 hrs
\( DW_{550} \): Dry weight of samples after burning in furnace at 550 °C for 4 hrs

% Carbonate = \left( \frac{DW_{550} - DW_{950}}{DW_{550}} \right) \times 100 \quad \text{(Equation A2)}

Where:

\( DW_{550} \): Dry weight of samples after burning in furnace at 550 °C for 4 hrs
\( DW_{950} \): Dry weight of samples after burning in furnace at 950 °C for 1.5 hrs
Appendix B: Grain-size Determination

Approximately 3 grams of freeze-dried of even numbered sediment samples were weighed out in beakers (the same weight of soil samples were weighed out). Samples were then dispersed with 10 ml of Sodium Tripolyphosphate solution. A dispersant was used with sediments in order to breakdown colloids in each soil and sediment samples and prevent re-clumping of particles after wet-sieving and prior analysis using Micromeritics Saturn DigiSizer. Dispersed sediments were wet-sieved using a 63µm mesh sieve. This process separated the larger fraction and sand-size (LFS) particles from the smaller grains such as silts and clays. To obtain percent of LFS that was contained in each sample analysis, this fraction was dried in a beaker overnight at 105 °C and weight was recorded.

To determine the percent silt and clay-size particles in each sample, the material that passed through the sieve (<63 µm) was then analyzed using the DigiSizer. No additional chemical preparation was necessary to perform this analysis. A light scattering technique employed by the Micromeritics DigiSizer was used to measure the particle size. Based on the how the particles react to the light helps estimates the size of the particle. Particles with an estimated diameter of < 4 µm was categorized as clays. Silt-sized particles considered to have diameters ≤ 63 µm, but greater than 4 µm. Figure A1 is a screenshot of report generated after analysis. The DigiSizer is set to run three test on each sample. Results from the third test is saved and used as final grain-size analysis.

The Digisizer provides a raw percentage for each of the particle class. These percentages are obtained and subtracted from the original mass of sediment in order to
determine actual percent composition for silt and clay-sized particles. For certain samples particles larger than 63 µm were measured by the instrument. These particles were considered to be LFS and were added to the initial sieved mass in order to obtain the total percent of LFS present in a given sample.

**Figure A1.** Screenshot of grain-size report after analysis with DigiSizer.
Appendix C: Graphs of Radionuclide Activity Not Normalized With Clay

$^{137}$Cs distributions for all sediment cores

Figure A2. $^{137}$Cs distribution not normalized with clay. Measurements for ESC1.
Figure A3. $^{137}$Cs distribution not normalized with clay. Measurements for ESC2.

Figure A4. $^{137}$Cs distribution not normalized with clay. Measurements for ESC3.
Figure A5. $^{137}\text{Cs}$ distribution not normalized with clay. Measurements for CCB4

Figure A6. $^{137}\text{Cs}$ distribution not normalized with clay. Measurements for AF1
Figure A7. $^{137}$Cs distribution not normalized with clay. Measurements for AF2

Figure A8. $^{137}$Cs distribution not normalized with clay. Measurements for ACN1.
Figure A9. $^{137}$Cs distribution not normalized with clay. Measurements for ASM1.

Figure A10. $^{137}$Cs distribution not normalized with clay. Measurements for SFM1.
Figure A11. $^{137}$Cs distribution not normalized with clay. Measurements for PJ1.

Figure A12. $^{137}$Cs distribution not normalized with clay. Measurements for JB1.
Figure A13. $^{137}\text{Cs}$ distribution not normalized with clay. Measurements for JB3.

Figure A14. $^{137}\text{Cs}$ distribution not normalized with clay. Measurements for JB4.
Excess $^{210}\text{Pb}$ distributions for all sediment cores

Figure A15. $^{210}\text{Pb}_{\text{ex}}$ distribution not normalized with clay. Measurements for ESC1.

Figure A16. $^{210}\text{Pb}_{\text{ex}}$ distribution not normalized with clay. Measurements for ESC2.
Figure A17. $^{210}\text{Pb}_{\text{ex}}$ distribution not normalized with clay. Measurements for ESC3.

Figure A18. $^{210}\text{Pb}_{\text{ex}}$ distribution not normalized with clay. Measurements for CCB4.
Figure A19. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for AF1.

Figure A20. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for AF2.
Figure A21. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for ACN1.

Figure A22. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for ASM1.
Figure A23. $^{210}\text{Pb}_{\text{ex}}$ distribution not normalized with clay. Measurements for SFM1.

Figure A24. $^{210}\text{Pb}_{\text{ex}}$ distribution not normalized with clay. Measurements for PJ1.
Figure A25. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for JB1.

Figure A26. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for JB3.
Figure A27. $^{210}$Pb$_{ex}$ distribution not normalized with clay. Measurements for JB4.
Appendix D: Gamma Spectrometry Calibration

Well efficiencies are plotted in Figure A28. Table A1 Shows tabulated version of graphed data. Figure A28 simply show that well detector efficiency is greater at low energy levels.

**Figure A28.** Gamma calibration was performed using NIST 4357 Standard Multiline
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<th>Energy keV</th>
<th>Efficiency (%)</th>
<th>Error</th>
<th>Computed keV</th>
<th>Error</th>
<th>Difference</th>
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Appendix E: Self –Adsorption and Weight Correction

A subset of samples was analyzed in two gamma detectors with different configuration in order to validate radionuclide activities determined for this study. Both the gamma well and planar detectors were used to obtained radionuclide activity concentrations; the same calibration standard was used to calibrate the well and planar detector. Activity concentrations for $^{137}$Cs and $^{210}$Pb were determined either by well detector or planar detector. This data is reported in chapters 3 and 4 of the dissertation document.

Low levels of radioactivity are typical for environmental samples therefore, it is important to determine the detector efficiency especially for those samples with gamma energies of 100 keV and below. Determination of self absorption characteristics and efficiencies for both detectors allowed for the verification of $^{137}$Cs activity concentrations for sediment and soil samples analyzed in this study. Figures A29. shows the self absorption trend and weight correction factors for samples analyzed using the planar detector. The full energy peak efficiencies are affected by the high self-absorption of the gamma rays emitted, which strictly depend on the energy of the gamma-ray considered as well as on the composition and apparent density of the analyzed sample. As the total weight of the sample increase self-adsorption chances will increase. Figure A30. shows that correlation and how using samples in a range of weights that were from 1gram to 11grams, will allow us to correct for the transmission received by planar detector as a function of sediment weight.
Adsorption is a function of sample density and as a result it influences sample counts (Cochran et al., 1998). Environmental samples are comprised of various materials such as organic matter, mineralogy, pH, grain-size and other materials. Recognizing this, environmental samples must be normalized in order to eliminate variation in sample densities, which affects activity transmission to detectors (Cochran et al., 1998). This normalization is performed using the same geometry, realizing that although the geometry and volume of a sample can be controlled, the density of a sample varies
according to the composition of the sample. As stated in Appendix D, which presents the efficiency for well detectors; efficiencies are greater at low energy levels. Sediment samples from the island of Vieques, Puerto Rico were analyzed using both the planar and well detector. Consistencies between the two detectors were observed and serve to validate the acceptable usage of both detectors for gamma analysis of samples.
ABOUT THE AUTHOR

Nekesha Bernadette Williams was born in the Republic of Trinidad and Tobago. She has a Bachelor of Science degree in Environmental Studies with a concentration in Biology from the State University of New York, College of Marine Science. In addition, she has obtained a Master of Science degree in Natural Resources with a focus on Watershed Hydrology. As a doctoral student, Nekesha has conducted research that integrated geospatial technology such as geographic information system (GIS) with radiogeochemistry. She is a recipient of a National Science Foundation-sponsored Florida-Georgia Louis Stokes Alliance for Minority Participation - Bridge to Doctorate (LSAMP-BD) Fellowship, an Alfred P. Sloan Minority Doctoral Scholarship and an University of South Florida (USF) Diverse Student Success Fellowship. She has received specialized training in GIS via an USF Graduate Certificate Program and the GIS Summer School which was convened in Vespucci Institute, Florence Italy.