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Sustainable natural resource management in regional ecosystems : case studies in semi-arid and humid regions

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SUSTAINABLE NATURAL RESOURCE MANAGEMENT IN REGIONAL ECOSYSTEMS: CASE STUDIES IN SEMI-ARID AND HUMID REGIONS

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ABSTRACT

Sustainability calls for policies that meet current societal needs without compromising the needs of future generations; thus, a dual relationship between human and natural resources is required. The main goal of the current research was to introduce up-to-date environmental techniques for sustainable natural resource utilization in semiarid and humid ecosystems in short and long term. To achieve this goal, two studies were implemented. First, sustainable land use management was evaluated in a newly reclaimed, semiarid region in the *Bustan 3* area (341.27 km^2), Egypt. To achieve sustainable management in this agro-ecosystem; detection of land cover change, assessment of the most sensitive areas to desertification, and evaluation of land capability for agricultural use were required. Using multi-temporal remotely-sensed data in the *Bustan 3,* the results indicated that this area had been drastically changed from 100% barren desert land to 79% agricultural land, due to successful land reclamation efforts in the 1990s. Although 70% of this area had a good capability for agricultural production, ~89% of the *Bustan 3* area was critically sensitive to desertification. By applying suitable land management scenarios, the land capability for agricultural use could be increased. Second, a natural resource conservation program was examined by studying the effects of compost/mulch, as a best management practice, for soil erosion control on highway roadsides in Louisiana, USA (a humid region). Louisiana is plagued by widespread impairments to surface water quality. Total suspended solids (TSS) and associated turbidity in runoff water are considered the most problematic nonpoint source pollutant of Louisiana surface waters. At the plot scale, the effects of compost/mulch on soil and water resources were evaluated. Research results showed that the use of compost/mulch without tillage incorporation successfully conserved the topsoil on the roadsides, increased soil moisture retention, moderated soil temperature, and reduced the TSS, soil loss, runoff, and water flow rate. Tillage incorporation is not recommended since it

decreased the compost/mulch effectiveness in reducing runoff and sediment losses. While the two studied areas, in semiarid and humid regions, were disparate in their characteristics, sustainable natural resource management was successfully achieved by using appropriate management practices in each case.

CHAPTER 1. INTRODUCTION/LITERATURE REVIEW

1.1 SUSTAINABILITY DEFINITIONS

Climate change, energy and fuel, material resource scarcity, water scarcity, population growth, urbanization, wealth, food security, ecosystem decline and deforestation; are a set of ten global sustainability megaforces that will affect the environmental change and every business for next two decades (KPMG, 2012). Particularly, population growth and urbanization are key drivers of the demand for energy, water, and food, and the resulting degradation and depletion of natural resources (Hecht *et al.*, 2012).

Over the past 50 years, global “sustainability” of natural resources has become increasingly important. The goal of achieving a sustainable planet, one that will accommodate the basic needs of its present inhabitants while preserving the resources that will enable future generations to flourish, has gained increasing acceptance (NRCS, 2012). According to the Environmental Protection Agency (EPA, 2012a), sustainability is

> Everything that we need for our survival and well-being depends, either directly or indirectly, on our natural environment. Sustainability maintains the conditions that permit fulfilling the social, economic and other requirements of present and future generations.

Lankford and Beale (2007) stated that sustainability is

> The management of natural resources to ensure their continued capacity to be productive in both agricultural and environmental capacities.

This statement utilizes the threshold limit, where below it, the natural resource is unable to generate, renew or protect itself, or it moves from self-renewing to exhaust.

Sustainability concept is a dynamic concept that varies with respect to social, economic and political factors. It evolves based on the perspectives of the public and private sectors. From a public perspective, sustainability should meet basic economic and social needs for now and in the future without undermining the environmental quality of natural resources. From a business

perspective, the goal of sustainability is to increase long-term shareholder and social value, while decreasing industry's use of materials and reducing environmental degradation (Hecht *et al.*, 2012). Sustainability goal is to merge the knowledge of all involved sectors; farmers, workers, and scientists…etc., to gain a broader perspective on the constraints and potential of natural resources management systems. This involvement will provide more realistic, efficient and acceptable strategy for the decision-makers (Lefroy *et al.*, 2000).

The term sustainable development is widely used as exchangeable term for sustainability. Sustainable development was introduced by the World Commission on Environment and Development (Brundtland Commission, 1987) as

> Development that meets the needs of the present without compromising the ability of future generations to meet their own needs.

Globally, sustainability has characterized as resting on three pillars: social well-being, economic prosperity, and environmental protection (Hecht *et al.*, 2012). In other words, sustainability involves three main components: social, economy, and environment (Figure 1.1). For the purpose of the current study only the environmental issue will be considered.

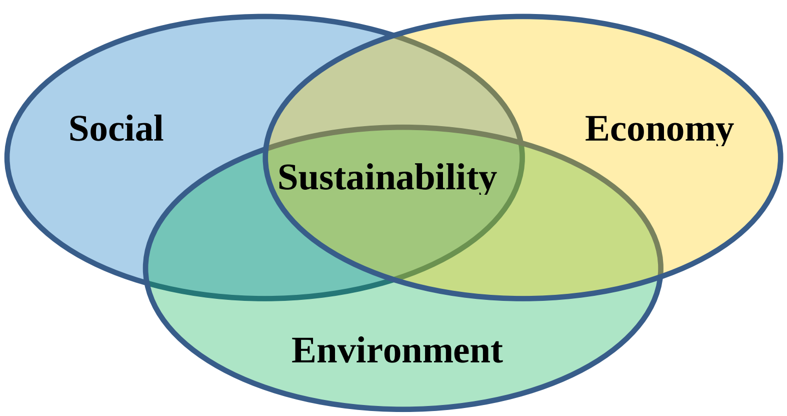

Figure 1.1. Schematic presentation of the sustainability components.

1.2 NATURAL RESOURCE MANAGEMENT

Growing pressures on natural resources are a worldwide issue that deteriorates the environmental systems, increases the risk of state changes, and finally reduces ecosystem resilience (Guerin, 2007). Natural resources represent the materials that occur naturally and are valuable in their unmodified form and can be renewable or non-renewable. The renewable resources can restock themselves when used in a sustainable manner (Jegatheesan *et al*., 2009).

Natural resource management (NRM) concerns the human impacts on the natural environment, the productivity of land and water bodies, and ecosystem services and qualities. The NRM refers to maintaining the quality of life and ethical values related to sustainable management. With increasing human population, effective management is becoming essential at all scales: local, regional, national, and global (Ostendorf, 2011). Rist *et al*. (2007) reported that the aim of a management is to define regulations, procedures, and technologies, which govern the relationship between humans and nature. This governance can be achieved by supporting multi-disciplinary work for a goal of accomplishing sustainable development (Hurni and Wiesmann, 2004). With the inherent complexity of interactions between socio-cultural, economic, and biophysical system components, management is difficult, so, information at spatial and temporal levels, is needed (Ostendorf, 2011).

Currently, there is a broad concern about developing of NRM practices that conserve soil and water resources and sustain environmental quality (Sahrawat *et al*., 2010). The degradation of soil and water resources has clearly shown that these resources are finite. Although soil mismanagement could benefit individual landholders in the short term, it negatively affects surface- and ground-water quality on a long term (Lal, 2007). Land resource management is a core of NRM process in general and in agriculture sector specifically. Land resources management is needed to keep the soil at an acceptable level of productivity and to reduce land

degradation. Thus, there is increasing concern on defining specific criteria for land management under various land uses.

1.3 SUITABLE LAND MANAGEMENT UNDER AGRICULTURAL LAND USE

Based on FAO (2012), sustainable land management (SLM) is defined as a knowledge-based procedure that helps integrate land, water, biodiversity, and environmental management to meet rising food and fiber demands while sustaining ecosystem services and livelihoods. Improper land management can lead to land degradation and significant reductions in productivity. Cowie *et al*. (2011) defined the SLM as

> The management of land to meet present needs without compromising the ability of future generations to meet their own needs.

Dumanski and Smyth (1994) stated that SLM is combined procedures that achieve productivity, security, protection, viability, and acceptability for both socioeconomic principles and environmental concerns. To keep the ecosystems functioning under dynamic conditions, they must be resilient (Chapin *et al*., 2010). A major objective of SLM, under agricultural land use, is to impart resilience to agro-ecological systems (Cowie *et al*., 2011). SLM encourages an integrated, holistic perspective on land management (Schwilch *et al*., 2011 and 2012).

Soil is one of the oldest natural resources that have been utilized by humans, so it is essential component in natural-human relationship. Although, soil can be replaced, it considers a non-renewable resource as the rate of replacement is much slower than the rate of utilization (Jegatheesan *et al*., 2009). Sustainable use of soil was defined by Hannam and Boer (2002) as

> The use of soil in a manner that preserves the balance between the processes of soil formation and soil degradation, while maintaining the ecological functions and needs of soil.

Agricultural activities, as the common land use, positively or negatively impacts soil and water quality at the watershed. Although, agricultural practices can improve or maintain soil

quality and sustain productivity, they may cause many problems such as; soil loss via water and wind erosion, loss of organic matter, waterlogging, salinization/alkalization of the soil, and the contamination of water resources. These adverse effects could be occurred when farming systems are intensified, without considerations of conserving soil and water resources (Sahrawat *et al*., 2010). Additionally, more negative effects occur when land in the sensitive ecosystems with porous soils, e.g. in semi-arid and arid regions, are cultivated disregarding soil and water conservation practices (Lilburne *et al*., 2004). Agro-ecosystems could be sustained when agricultural practices have been achieved in a sustainable manner. An effective sustainable agriculture system involves the development of farming strategy considering all factors that maintained crop productivity without negatively impacting the environment (Rao *et al*., 2000). Based on American Society of Agronomy (ASA, 1989):

> Sustainable agriculture is one that, over the long-term, enhances environmental quality and the resource base on which agriculture depends; provides for basic human food and fiber needs; is economically viable; and enhances the quality of life for farmers and society as a whole.

Complexity of the agro-ecosystem patterns makes the management decisions difficult to implement. Using advanced technologies (e.g. remote sensing (RS), geographic information system (GIS), modeling, rapid measurements in-situ, and best management practices (BMPs) are highly valuable to dealing with the interaction relationships of land resources management. RS as well as GIS technologies are powerful tools for detecting, studying, investigating, and interpreting land resources as they are capable to study soils at spatial and temporal domain with a cost effective manner. Additionally, RS and GIS provide up-to-date and archive information on land resources, land degradation, and land use/land cover (LULC) changes, which are essential for sustainable land use planning. Although, the dynamics of land-use changes is critical key for sustainability, it is not alone sufficient to achieve sustainability. Decision-makers have to have

information on the system performance, the driving forces, current status of soil resource (e.g. land capability), and the potential change in soil over time (change detection) to sustain agricultural land use (Sharmaa *et al*., 2006). Monitoring LULC can provide early warning of adverse impacts, if any, and identify the most critically affected areas through environmental sensitivity analysis (Bindraban *et al*., 2000). Based on those procedures; change detection, environmental sensitivity analysis, and land capability are needed for identifying the appropriate land management practices that can achieve sustainable use of the land resources.

1.3.1 The Bustan 3 area, Egypt

Agriculture is a key sector of the Egyptian economy (IFAD, 2005). Egyptian agricultural land can be divided geographically into *Upper* and *Lower* Egypt, where *Upper* Egypt comprises the Nile Valley from Giza to the south and *Lower* Egypt comprises the Nile Delta from Cairo to the north. These lands can be further divided into "*Oldlands*" and "*Newlands*." *Oldlands* are found in the Nile Valley as well as the Nile Delta and include the lands that have been intensively cultivated for thousands of years. *Newlands* include lands that have been reclaimed relatively recently (post-1950) or are in the process of being reclaimed (Figure 1.2). *Newlands* are less fertile, but with time and good management of water and cropping patterns, their productivity can improve (UNDP, 2003). The total area of Egypt is around one million square kilometers. Approximately 95% of population lives on only 4% of the Egypt land. Since the 1980s, the Egyptian government has advocated policies aimed at extending cultivated land and maximizing production of the existing agricultural lands. Thus, determination of the trend and rate of land cover conversion are required for the development sustainable land use planning (Shalaby and Tateishi, 2007). Land reclamation in the Egyptian context means converting desert areas into agricultural land by extending water canals into the desert, enhancing soil fertility, and providing infrastructure for new village construction (Adriansen, 2009).

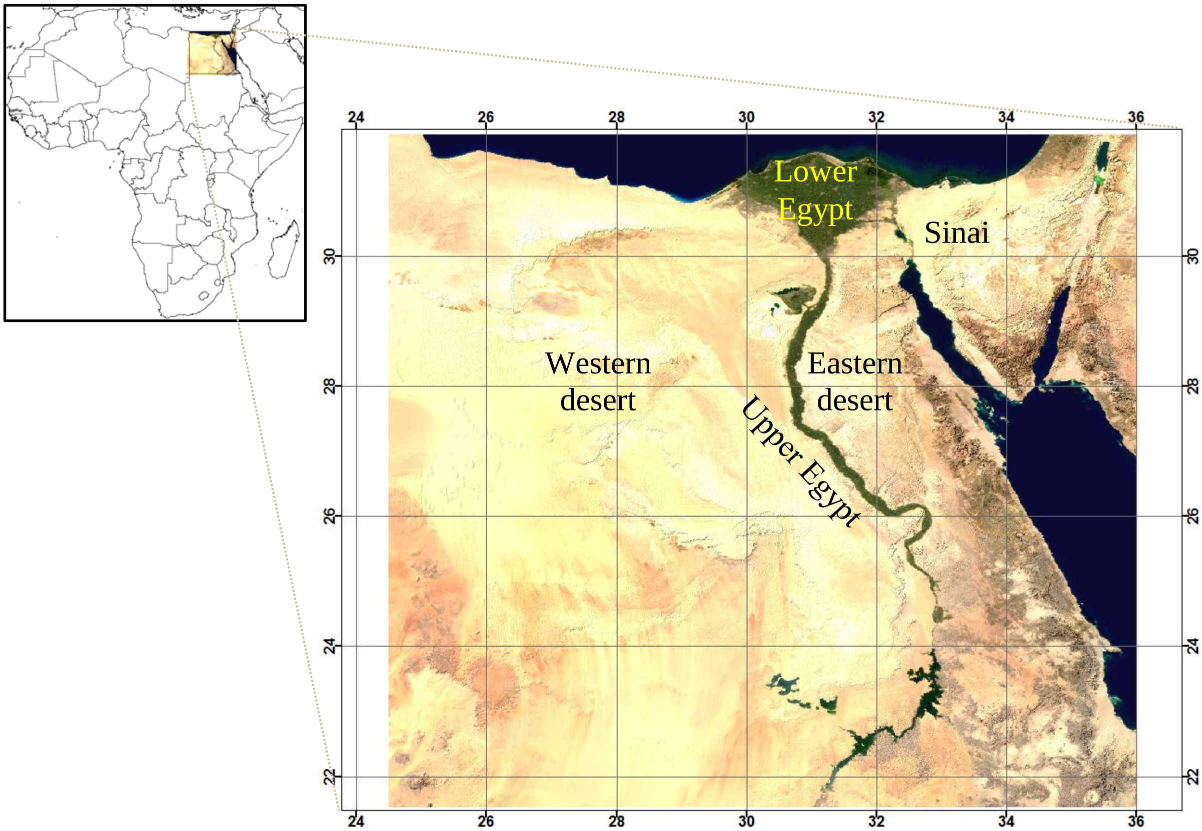

Figure 1.2. General location and main geographical divisions of Egypt soils.

Land reclamation remains high on the agenda of the Egyptian Government and is supported directly or indirectly by international agencies (Bush, 2007). The desert region of the West Delta in Egypt includes a total land reclamation area of 2346 km^2, with 823.2 km^2 reclaimed prior to 1978. The Egyptian government plans for 60% of the reclaimed area to be auctioned to investors and 40% to be distributed on concessionary terms to small farmers and unemployed graduates. The *Newlands* Agricultural Services Project (NLASP) area comprises 789.6 km^2 of recently reclaimed land which was allocated in 0.021 km^2 parcels to selected settlers in three separate localities: *West Nubaria*, *El Bustan*, and *Sugar Beet* areas. The reclamation process includes the installation of irrigation and drainage systems, the construction of roads, houses, community buildings, a supply of portable water, and electricity (IFAD, 1992).

The *Bustan 3* area (Figure 1.3) is one of the *Newlands* in the western Nile Delta, Egypt that targeted to the reclamation processed during 1990s (IFAD, 1992).The geographical location is in UTM zone 36 between latitudes 3368500 to 3392000 N (30º 26´ to 30º 39´ N) and longitude 226000 to 255500 E (30º 80´ 30" to 30º 27´ E), occupying around 341.28 km^2 (34,128 ha).

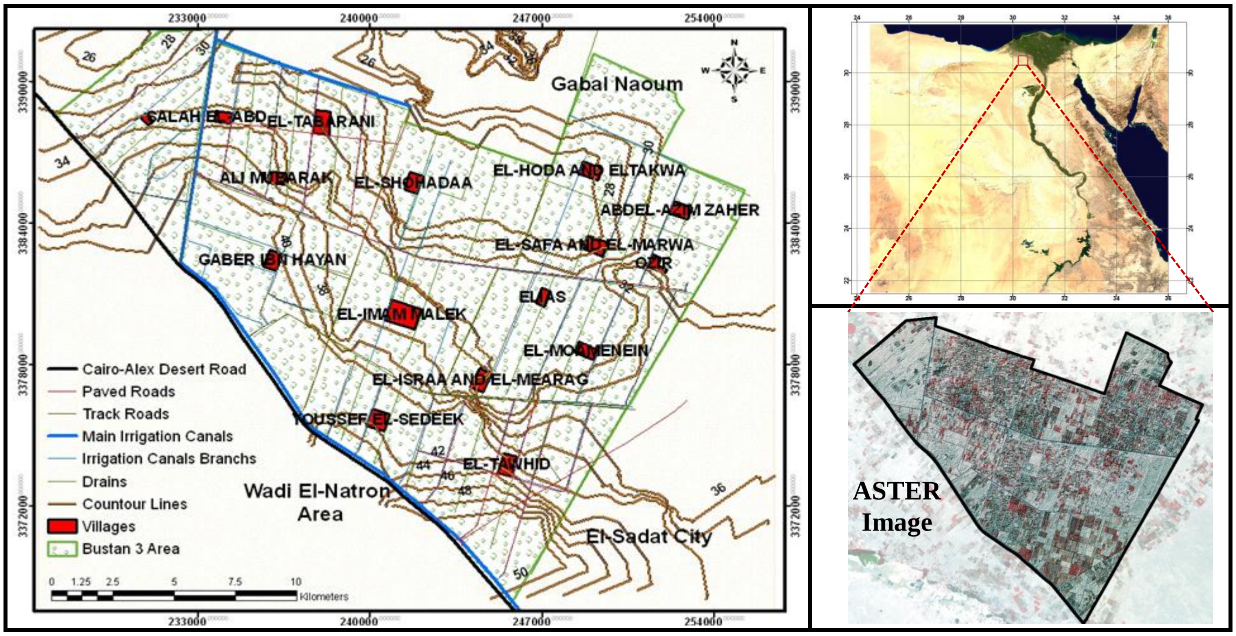

Figure 1.3. Location and main infrastructure of the *Bustan 3* area, Egypt.

The study area is characterized by a Mediterranean climate and can be considered semiarid. Table 1.1 shows the average of 30-yr period climatic parameters collected from the Tahrir meteorological station, the nearest station to the study area (FAO, 1993). Two main landforms of the *Bustan 3* area; the relatively low altitude landform that characterized by an undulating land form of coarse sand, and sandy plain landforms which are sandy, nearly level sediments of the deltaic stage of river terraces (Sadek, 1993). The *Bustan 3* soils are classified as Typic Torripsamments based on Soil Survey Staff (2010). Sadek (1993) reported that this area contains desert geomorphic units such as sand dunes and sandy plains. The geological deposits represent the Pliocene, Holocene, and Pleistocene eras. Figure 1.4 shows the reclamations

processes in the *Bustan 3* area, Egypt, starting from bare soils to a successful agricultural production.

Table 1.1. Average climatic data (over 30 years) collected from Tahrir meteorological station, Egypt. (Source: Bakr *et al.*, 2009).

Month	Temperature Maximum	Minimum	Rainfall	Relative Humidity	Wind Speed	Sunshine
	------------°C------------		---mm---	----%----	---km/d---	Hours
January	19.6	7.8	10.0	80.0	268.0	7.0
February	20.5	8.0	7.0	79.0	311.0	7.9
March	24.0	10.1	1.0	76.0	328.0	8.6
April	28.0	12.5	1.0	68.0	311.0	9.6
May	31.7	15.1	1.0	66.0	311.0	10.9
June	34.3	18.8	0.0	68.0	285.0	12.0
July	34.5	20.3	0.0	71.0	259.0	11.7
August	34.8	20.8	0.0	72.0	216.0	11.1
September	32.5	18.7	0.0	74.0	207.0	10.3
October	30.2	16.5	2.0	73.0	207.0	9.2
November	25.7	13.2	5.0	77.0	216.0	8.1
December	21.5	9.5	8.0	78.0	259.0	7.0
Average	28.1	14.3	35.0	73.5	264.8	9.4

Figure 1.4. Agricultural production development in the *Bustan 3* area, Egypt. The three photos up, indicate the area while it was barren land then the insulation of irrigation system. The three photos bottom, show the successful agricultural production under different vegetation intensity.

1.3.2 Land use/land cover change

Change detection is the process of identifying differences in the state of an object by observing it at different times (Singh, 1989). Two primary categories of change detection exist. One focuses on detection of detailed change trajectories, called 'from-to'; post classification comparison is a common example of this approach. The second focuses on the detection of binary change/non-change features, such as vegetation index differencing (Lu *et al*., 2004). Remotely sensed satellite imagery is the most appropriate source of information to determine LULC change (Currit, 2005), as it offers the opportunity to assess the effects of reclamation processes and provide the data needed for the development of national agricultural strategies (Pax Lenney *et al.*, 1996). Landsat satellite data is the most widely used data type for land cover mapping as it has provided earth observation data since 1972 (Williams *et al*., 2006) with relatively high spatial resolution (Cohen and Goward, 2004; Wulder *et al*., 2008) and free access by the Geocover dataset under USGS website (Knorn *et al*., 2009). Two most common classification methods have been used for studying LULC change; unsupervised and supervised classifications, which are considered as pixel-based classification (Moreno and De larriva, 2012). The hybrid classification technique which is a composite technique from unsupervised and supervised classifications could be used to increase the accuracy of the final classified map (Castellana *et al.*, 2007; Bakr *et al*., 2010).

According to Richards and Jia (2006), supervised classification is performed by a series of steps: 1) decide the ground cover classes, 2) identify the training sets, 3) create the signature for each class, 4) classify the pixels, 5) produce thematic maps, and 6) assess the accuracy of those maps. After verifying the location of a specific land cover type via ground truth points, different supervised signatures, for each class, are developed. Separability analysis can be performed on the training data to estimate the expected error in the classification (Swain and

Davis, 1978; Landgrebe, 2003). Based on the separability cell array, different signatures per class can be merged together (Jensen, 2004). Finally, the probability was normalized for all signatures. The maximum likelihood decision rule is the most common parametric rule algorithm in supervised classification (Richards and Jia, 2006). The maximum likelihood classifier is a conventional statistical classification technique that allocates each pixel to the class that has the highest likelihood or probability of membership (Schowengerdt, 2007; Mather, 1999). The basis of this algorithm is the probability density function (PDF), which may be derived from Eq. 1.1:

$$p(x_k|i) = \frac{1}{\sqrt{2\pi}\sqrt{|M_i|}} \exp\left(-\frac{1}{2}D^2\right) \tag{1.1}$$

Where, $p(x_k|i)$ represents the PDF for pixel k with data vector x_k as a member of class i, M_i is the variance – covariance matrix for class i and D^2 is the Mahalanobis distance between the pixel k and the mean vector (v_i) of the pixel's class i. D^2 may be calculated from Eq. 1.2:

$$D^2 = (x_k - v_i)^T M_i^{-1}(x_k - v_i) \tag{1.2}$$

When pixels are classified incorrectly, thresholding analysis could be performed. These pixels are identified statistically, based upon the distance image file and classified raster image (Swain and Davis, 1978).

The unsupervised classification approach is an automated classification method that creates a thematic raster layer from a remotely sensed image by letting the software identifies statistical patterns in the data without using any ground truth data (Lillesand *et al*., 2008). The iterative self-organizing data analysis technique (ISODATA) clustering method uses spectral distance as a sequential method (Tou and Gonzalez, 1974). ISODATA is iterative, so it repeatedly performs an entire classification and recalculates statistics. Pixels belonging to a particular cluster are therefore spectrally similar. The most frequently similar matrix encountered is Euclidean distance (Richards and Jia, 2006). If x_1 and x_2 are two pixels whose similarity is to

be checked and *N* is the number of spectral components, the Euclidean distance between them is given in Eq. 1.3:

$$d(x_1, x_2) = \|x_1 - x_2\| = \left\{(x_1 - x_2)^t (x_1 - x_2)\right\}^{\frac{1}{2}} = \left\{\sum_{i=1}^{N} \left(x_{1_i} - x_{2i}\right)^2\right\}^{\frac{1}{2}} \tag{1.3}$$

The normalized difference vegetation index (NDVI) is the most widely used index in the processing of satellite data (Myneni *et al*., 1995; Tucker, 1979). The NDVI is defined in Eq. 1.4 as (Rouse et al., 1974):

$$NDVI = \frac{(\rho NIR - \rho R)}{(\rho NIR + \rho R)} \tag{1.4}$$

Where, ρNIR and ρR are spectral bidirectional reflectance factors at near-infrared and red wavelengths, respectively (Rouse *et al*., 1973; Bannari *et al*., 1995). The NDVI values range from −1 to +1. Values of < 0, ~ 0, and > 0 are non-vegetated, water, and vegetation, respectively (Krishnaswamy *et al*., 2009). The NDVI values of arid, semiarid, or Mediterranean areas during the dry summer season are strongly dependent on plant water availability in preceding months (Maselli, 2004). For Landsat satellite images, the red and near-infrared wavelengths are presented in band 3 and 4, respectively.

The US Geological Survey (USGS) defines spatial data accuracy as: “The closeness of results of observations, computations, or estimates to the true values or the values accepted as being true” (US Geological Survey, 1990). Accuracy results are expressed in tabular form; often known as error, confusion, or contingency matrix. Different measures can be derived from the values in an error matrix, and user's and producer's accuracy (Janssen and van der Wel, 1994; Banko, 1998), depending on whether the calculations are based upon the matrix’s row or column (Campbell, 2011; Story and Congalton, 1986). The user's accuracy is a measure of the reliability of the map. The wrong classes are referred to as errors of commission.

User's accuracy (%) = 100% − error of commission (%)

The producer's accuracy is derived by dividing the number of correct pixels in one class by the total number of pixels as derived from reference data. It includes the error of omission.

Producer's accuracy (%) = 100% − error of omission (%)

Kappa coefficient is commonly used as a measure of map accuracy (Hudson and Ramm 1987; Congalton and Green 1999) and was developed by Cohen (1960). Kappa coefficient has become a widely used measure for classification accuracy and was recommended as a standard by Rosenfield and Fitzpatrick-Lins (1986). According to Bishop *et al*., (1975) Kappa coefficient can be calculated by Eq. 1.5:

$$\mathrm{K}^\wedge = \frac{\mathrm{N}\sum_{i=1}^{r} x_{ii} - \sum_{i=1}^{r} x_{i+}x_{+i}}{N^2 - \sum_{i=1}^{r} x_{i+}x_{+i}} \tag{1.5}$$

Where, r is the number of rows and columns in the error matrix, N is the total number of observations, X_{ii} is observations in row i and column i, X_{i+} is the marginal total of row i, and X_{+i} is the marginal total of column i. Large classes tend to be represented by a larger number of sample points than the smaller classes. Some very small classes may not be represented at all.

1.3.3 Environmental sensitivity area to desertification

Arid and semi-arid regions are characterized by limiting water resource that precludes the ecosystem functionality on such areas. Land degradation is a serious problem in such regions due to their fragility and vulnerability to degradation processes. Land degradation is defined as

> Reduction or loss of the biological or economic productivity and complexity of agricultural land, forests and woodlands (United Nations, 1994).

In order to determine the sustainable land use in fragile ecosystem, there is an urgent need to identify the driving forces that leading to land degradation. Land degradation is a global problem that involves climate, soil, vegetation, economic, and population and used to describe an

environmental phenomenon affecting dry lands (Salvati and Zitti, 2009; Salvati *et al*., 2011). Dryland areas are environmentally fragile and susceptible to degradation, severe degradation is blamed for the disappearance of around 5–10 million ha of agricultural land annually (Gao and Liu, 2010). Desertification is the label for land degradation in arid, semiarid, and dry sub-humid areas, collectively called drylands (Adamo and Crews-Meyer, 2006). Desertification was defined by UNCCD (1999) as a process of land degradation in arid, semiarid, and dry sub-humid areas that is the result of several factors, including human activities and climate variation. Considering the complex and interrelated processes among the set of natural and anthropogenic factors, explaining the susceptibility of land to degradation or desertification is difficult. To assess sustainable land use, addressing the degradation and risks associated is required by using proper methods according to the locally dominant degradation-related processes (Contador *et al*., 2009). Land degradation could be studied via several methods, such as field visits and remote sensing. Remote sensing method is cost-effective and time-efficient compared with the field method (Gao and Liu, 2008). Remotely sensed data are effective in identifying and mapping land degradation risks (Lu *et al*., 2007). Monitoring of the long-term trend of land degradation requires consistent and repeatable data that are available for many years; multi-temporal remote sensing data are the perfect source for this application. Land degradation severity and its process can be efficiently monitored from multi-temporal satellite images (Collado *et al*., 2002).

In agronomy, there is a difference between degradation and desertification: the former is not necessarily an irreversible process and can be controlled and stabilized with appropriate technical intervention, while the latter is a permanent, practically irretrievable, situation with an almost total loss of biological potential. Although soil degradation is largely induced anthropically, via agricultural activities, natural events can also contribute to this phenomenon (Basso, 2000). Controlling land degradation on agricultural land is important to the objectives of

sustainable growth and increasing the welfare of the many people who depend on agriculture for their livelihoods. The resulting land use practices will affect the level of production, the quality of the land, or even can lead to land degradation. The on-site effects of land degradation on agricultural land are a major source of concern, since they threaten the sustainability of agricultural production. On the other hand, expansion of agriculture into new areas may mask the effects of degradation on existing agricultural land. Continued expansion will be increasingly difficult, and will bring into use more marginal land (Pagiola, 1999). Agricultural land use has changed over time; factors affecting these changes include increasing agricultural productivity and intensification, changes in population density, industrialization, urbanization, tourism, agricultural mechanization and use of agrichemicals (Tanrivermis, 2003). Salvati and Zitti (2008) explored the temporal variation (1990–2000) of a synthetic index of vulnerability to land degradation on the whole Italian territory. Santini *et al*. (2010) developed a GIS-based software tool for the qualitative assessment of desertification risk at Sardinia Island (Italy). Six driving factors of desertification (overgrazing, vegetation productivity, soil fertility, water erosion, wind erosion and seawater intrusion) were model-simulated over two time periods to investigate the spatio-temporal evolution pattern of desertification-prone areas.

Rubio and Bochet (1998) stated that indicators are integrated, simplified, and synthetic information that provide data on status and evolution of relevant physical, chemical, biological and anthropogenic processes that related to complex processes such as desertification. Additionally, indicators can be easily used by decision-makers and imported to GIS to determine the causes and effects at a spatial geographical extension. Heink and Kowarik (2010) explained that indicators can be distinguished as; descriptive versus normative indicators and indicators as measures of ecological attributes versus as ecological components.

They also stated that an indicator in ecology and environmental planning is

> A component or a measure of environmentally relevant phenomena used to depict or evaluate environmental conditions or changes or to set environmental goals (Heink and Kowarik, 2010).

Environmental indicators (EI) play vital role in environmental reporting as they provide an important source of information for policy makers and help to guide decision-making (Niemeijer and de Groot, 2008). Based on Jackson *et al*. (2000), EI are important because they provide ‘‘a sign that relays a complex message in a simplified and useful manner.” Environmental sensitivity can be defined as the response of the environment to a change in one or more external factors. The relationships between the cause of the change and the effect are complex because different environmental components respond directly but they are affected indirectly. Degradation occurs when the response is considered deleterious to the ‘health’ of the environment (Basso *et al*., 2000). An environmentally sensitive area to degradation could be considered as a spatially delimited area in which some key aspects related to its sustainability are unbalanced and not sustainable for a particular environment (Basso *et al*., 2000). This is linked to interactions among elementary factors that are directly or indirectly related with key processes involved in the degradation phenomenon (Contador *et al*., 2009).

1.3.4 Scenario analysis

In agro-ecosystems, the management of soil resources is critically needed. A balance between sustaining the high level of agricultural production and preventing environmental degradation is a challenge for decision-makers. Sustainable soil management should maintain soil functionality, keep diversify of agro-ecosystem, and account for all options to increase crop production (Robert *et al*., 1993). Accordingly, there is an increasing need to determine soil properties, produce soil characterization maps, and develop practical recommendations based on scientific principles for land evaluation of different land uses. Studying soil types,

physiochemical properties of soil and their spatial distribution, type of land use, and land capability is required. Land capability reflects the physiochemical properties of the soil as well as the climatic conditions (De la Rosa, 1992). Integrating the sustainability concept with computerized models for evaluating land resources could be a very beneficial tool for achieving multi-scenarios based on inherent characteristics of the soil. According to Intergovernmental panel on climate change (IPCC, 2012), scenario in natural science was defined as

> A coherent, internally consistent and plausible description of a possible future state of the world. It is not a forecast; rather, each scenario is one alternative image of how the future can unfold.

For achieving sustainable land use planning, there are many factors affecting the selection of land use scenarios (Chen *et al*., 2003). Based on KPMG (2012),

> Scenario analysis helps to identify systemic risks that may emerge from the interactions of sustainability megaforces and provides a context for identifying growth opportunities before they become mainstream.

1.4 SUITABLE LAND MANAGEMENT ON ROADSIDES

Soil erosion is an environmental concern as it leads to loss of topsoil and sedimentation of water bodies (Pieri *et al*., 2007). Thus, it has on-site and off-site impacts on soil and water resources (Girmay *et al*., 2009). On-site, soil erosion affects physicochemical soil properties by loss of nutrient-rich topsoil, decreasing fertility and productivity, which results in land degradation (Ebisemiju, 1990). Off-site impacts of soil erosion include increased sedimentation and turbidity, increased levels of nutrients and pollutants in the waterbodies, and siltation of dams and irrigation channels (Hopmans *et al*., 1987; Ji, 2008). Besides human intervention, natural factors such as steep topography, erosive soil types, and high rainfall intensity can lead to soil erosion (Hartanto *et al*., 2003). Rainfall plays an important role in determining the magnitude of runoff and soil loss. The worst case scenario exists when steep hill slopes are placed in high-rainfall areas. While there are various sources of impacts, skid trails and logging

roads have been identified as major sources of sediment from anthropogenic activities (Hartanto *et al*., 2003). As erosion is a natural process it cannot be completely eliminated. However, best management practices (BMPs) can be used for controlling and managing sediment loading (U.S. Environmental Protection Agency, EPA, 2005). One effect of urbanization is an increase of the area of impermeable surfaces. This in turn has numerous consequences for some city infrastructure and surrounding environment. Decreased infiltration increase surface water runoff and stress on existing stormwater infrastructure (Berndtsson, 2010). As cities grow, new highways are constructed for transportation and development. Consequently, the streams ecosystems within highway corridors are susceptible to impacts from construction activities (Chen *et al*., 2009; Berndtsson 2010). With increasing mechanization, road construction impacts on soil have dramatically increased especially in humid areas, where high rainfall exacerbates the risk of soil erosion (Hartanto *et al*., 2003). Although construction is not a source of water pollution, the sedimentation processes due to soil disturbance during construction activities are considered a major nonpoint source (NPS) of pollution (Houser and Pruess, 2009). Road construction impacts on soil are significantly increased in humid areas, where high rainfall exacerbates soil erosion risk and amplifies this risk on steep hill slopes in such areas. Houser and Pruess (2009) explained that sediment is problematic for water quality since turbid water can restrict sunlight, consequently affecting aquatic life. Additionally, suspended particles often contain adsorbed pollutants (organics/heavy metals) on their surfaces.

In the United States, about 19% of the total land area has been directly affected by the public roads system (Forman, 2000). Chen *et al*. (2009) stated that, as one of the major NPSs of water pollution, the construction of new highways can have short- and long-term effects on stream biotic and abiotic conditions (Barton, 1977; Stout and Coburn, 1989; Wellman *et al*., 2000; Hedrick *et al*., 2007). These effects mainly result from sedimentation, habitat degradation,

changing of leaf processing, and inputs of toxins from construction materials (Barton, 1977; Stout and Coburn, 1989; Eldin, 2002). Keller and Sherar (2003) pointed out that roads are to blame for approximately half of the erosion from logging operations, and most erosion occurs during the first rainy season after disturbance.

Various BMPs have been developed and implemented to prevent environmental impacts of human activities. Along highways, numerous BMPs have been used to impound runoff and control soil erosion such as: vegetated buffers and mulches, porous pavement materials, retention or detention basins and ponds, silt fence, seeding, and natural riparian wetlands have been implemented as BMPs to treat runoff and control soil erosion (Han *et al*., 2005; Li *et al*., 2006; Hogan and Walbridge, 2007; Houser and Pruess, 2009). However, the effectiveness of some implemented BMPs on water quality protection is still unclear (Easton *et al*., 2008). Land use and soil cover are considered the most important factors affecting the intensity and frequency of overland flow and surface wash erosion (García-Ruiz, 2010; Kosmas *et al*., 1997). The amount of bare soil on a site is generally a good indicator of the soil's vulnerability to erosion and degradation. Good soil coverage is an essential element in soil conservation programs. Vegetation protects the soil from eroding in various ways. Rainfall interception by the plant has two main consequences, the most important being that it reduces the erosive power of impacting raindrops. It also reduces the volume of water reaching the soil surface (Nunes *et al*., 2011).

Monitoring of soil loss, using runoff plots, is cost-effective and provides valuable information about soil erosion risks caused by runoff. Runoff plots clearly demonstrate site disturbances where the plots are located. Monitoring allows for direct linkages to be made between management practices and their impacts on runoff and soil erosion, thereby enabling decision makers to identify problems and take appropriate preventive measures to improve their management practices (Hartanto *et al*., 2003).

1.4.1 Site specification and plot establishment

Louisiana is in the south central part of the U.S. (Figure 1.5). Based on the Louisiana water quality inventory report (Louisiana Department of Environmental Quality, LDEQ, 2010), Louisiana land covers 43,562 miles^2 (112,825.06 Km^2) which represents 84% of the entire state area while 16% of Louisiana is covered with water (8,277 miles^2, 21,437.33 km^2). As the state has many river systems, alluvial sediment covers most parts of the state territory. Pleistocene terraces and Holocene alluvium are the main geological features exist in Louisiana (Weindorf, 2008). Five physiographic regions exist in Louisiana; Costal Marsh, Mississippi Alluvial Valley, Red River Valley, Terraces, and Hills. Louisiana territory is mostly flat and the elevation gradually decreases from northwest to south. With the plenty of water bodies in Louisiana, many of them remain impaired for the designated use of aquatic life. Turbidity, total suspended solids (TSS), and biochemical oxygen demand (BOD), were mainly associated with most impaired water bodies and they can be related to NPSs of pollution by the runoff from agricultural fields; forestry areas; construction sites; and urban areas.

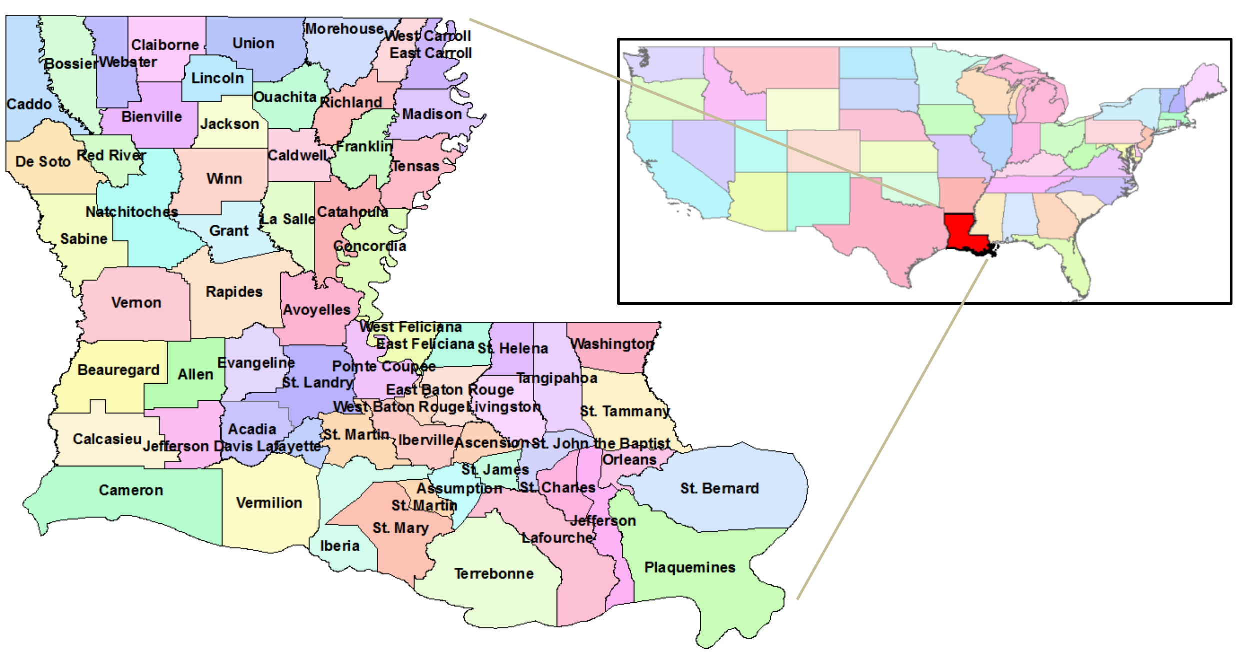

Figure 1.5. General location of Louisiana State, USA.

Louisiana climate is a humid subtropical associated with warm, moist air from the Gulf of Mexico in south/southeast direction (LDEQ, 2010). As high annual precipitation is expected from such areas, Louisiana average annual precipitation (Figure 1.6) varies from 47 to 71 inches (119.4 to 180.3 cm) from northwest to southeast, respectively (Weindorf, 2008).

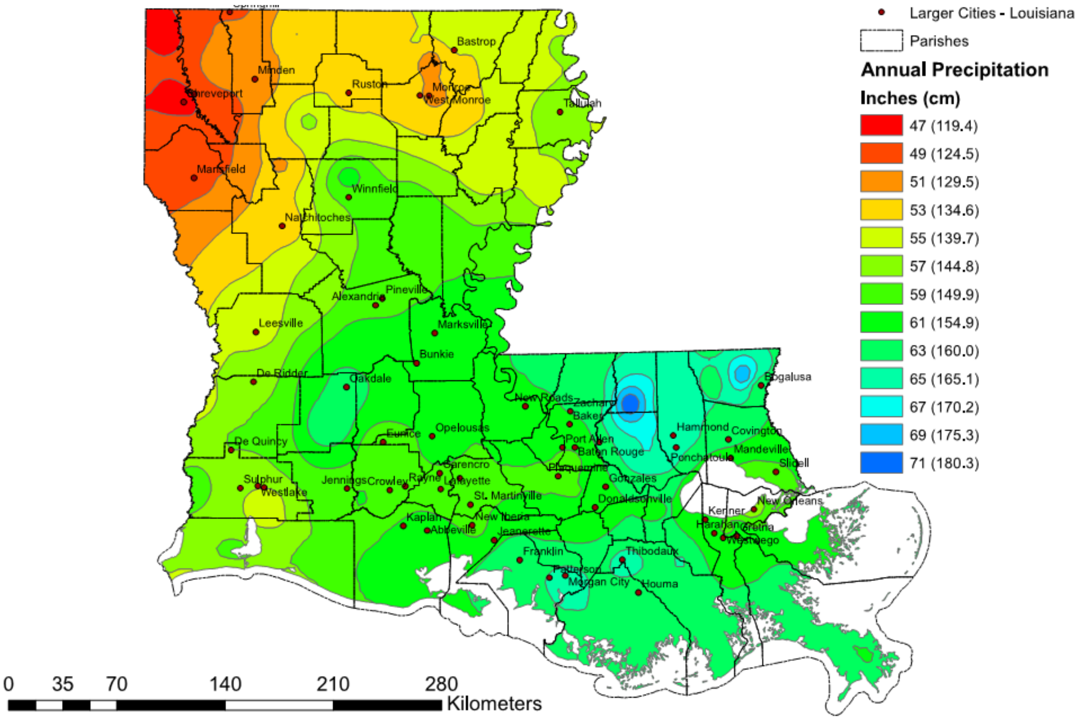

Figure 1.6. Average annual precipitation in Louisiana, USA. (source: Weindorf, 2008)

In the current study two locations on highways right-of -ways in Louisiana, USA, were chosen. The first location was adjacent to the northbound lane of US Highway 61, around 8 km away from St. Francisville city in west Feliciana parish, and had one site (site 1, S1). Site 1 had the steepest slope of 34% and was an active construction area during 2010/2011. The second location on the roadside of IH-49, about 20 km from Bunkie city in Rapids parish, and had three sites; site 2, site 3, and site 4, as S2, S3, and S4, respectively (Figure 1.7). IH-49 roadside is prone to erosive undercutting in many areas. Site 2 was an erosive “blowout” area adjacent to the northbound lane with slope of 25%. Site 3 was an erosive “backcut” area adjacent to the southbound lane with 15% slope. Site 4 was an erosive “blowout” area in the center median of

IH-49 with the lowest slope of and 10%. Two plots (side-by-side) were constructed at each site. The plots were established in February, 2010 and the experiment was wrapped up by May, 2012. All plots had a fixed size of 4.0 m X 4.0 m. Heavy gauge steel was used for edging the plots from three sides. The downslope side directed runoff from the plots into 0.305 m depth H-flumes. At each site, one of the two plots was lightly tilled at the beginning of the experiment and one kept non-tilled (Bakr *et al*., 2012). Compost/mulch mixture was used with different rate at each plot. The mulch materials were locally harvested (70% hardwood and 30% pine trees). Compost blended into the mulch was a double-ground, screened, recycled wood fiber material, also harvested locally (Bakr *et al*., 2012). The compost/mulch was applied at soil surfaces in different thickness (10 cm, 5 cm, and no-compost/mulch (as a control)) to evaluate the effectiveness of these materials as BMPs to control soil erosion and conserve water quality at the adjacent water bodies (Figure 1.8). Previously, Hartanto *et al*. (2003) concluded that at the plots scale, the presence of organic materials is very important in preventing soil detachment and providing surface roughness, which reduces runoff and soil particle movement.

A Refrigerated Isco® Model 6712 auto-sampler was utilized for each plot to collect the water samples after each rainfall event. Isco® auto-samplers were programmed for uniform 24-h composite samples with 5-min frequency time intervals. The H-Flumes, size of 1.00 ft (0.305 m), were designated for this experiment since they are capable for monitoring flow over a wide range with a high accuracy (Grant and Dawson, 1997). Isco auto-samplers had a capability to record and store temperature, rainfall, and the levels data. Level data indicates to the depth of water inside the flume (m), which can be used to calculate the flow rate and the volume of water using Isco Flowlink 4.15 software (ISCO, 2002).

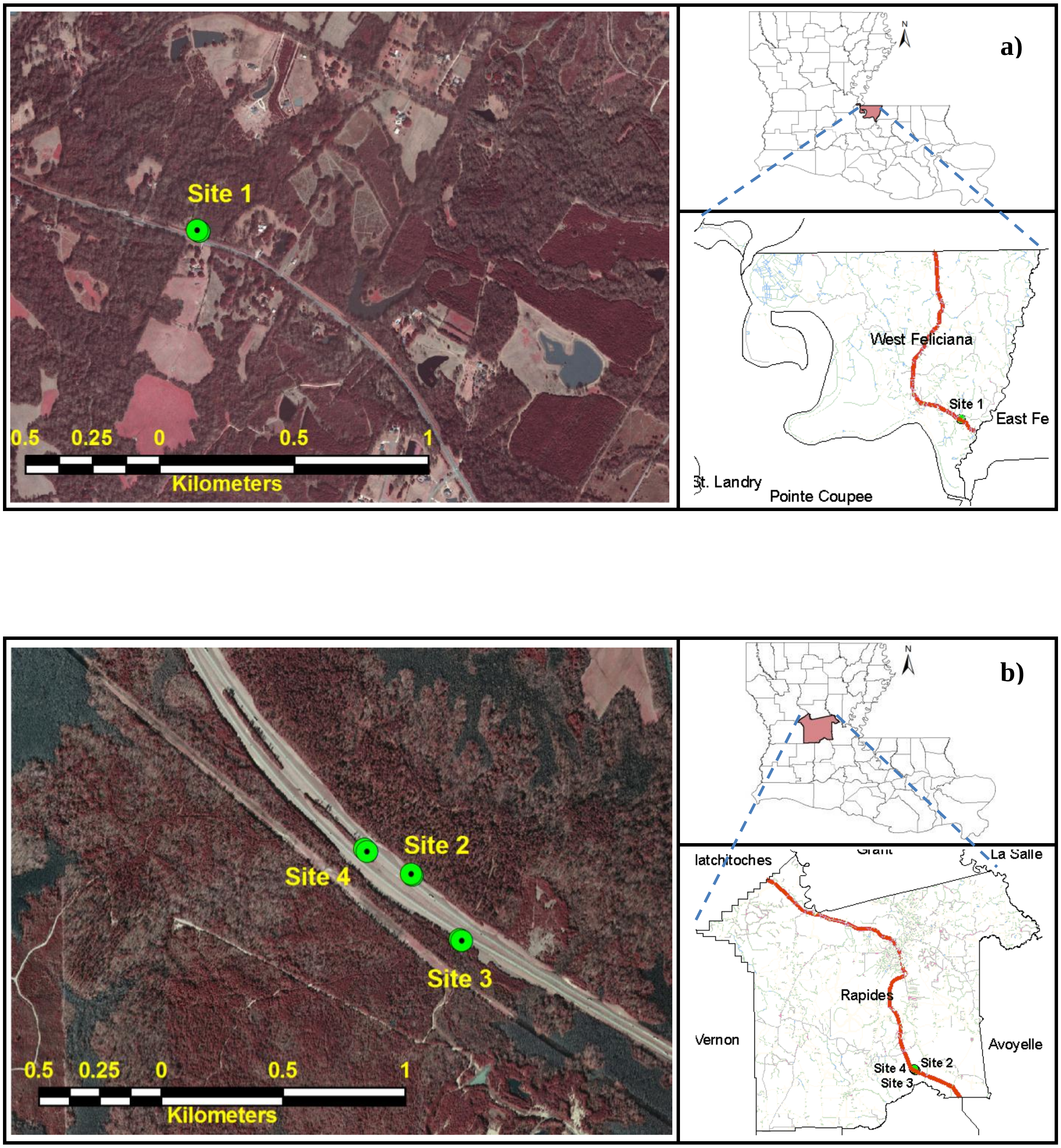

Figure 1.7. The studied sites along two different highways in Louisiana, USA. a) Site 1 along US Highway 61, b) Sites 2, 3, and 4 along IH-49.

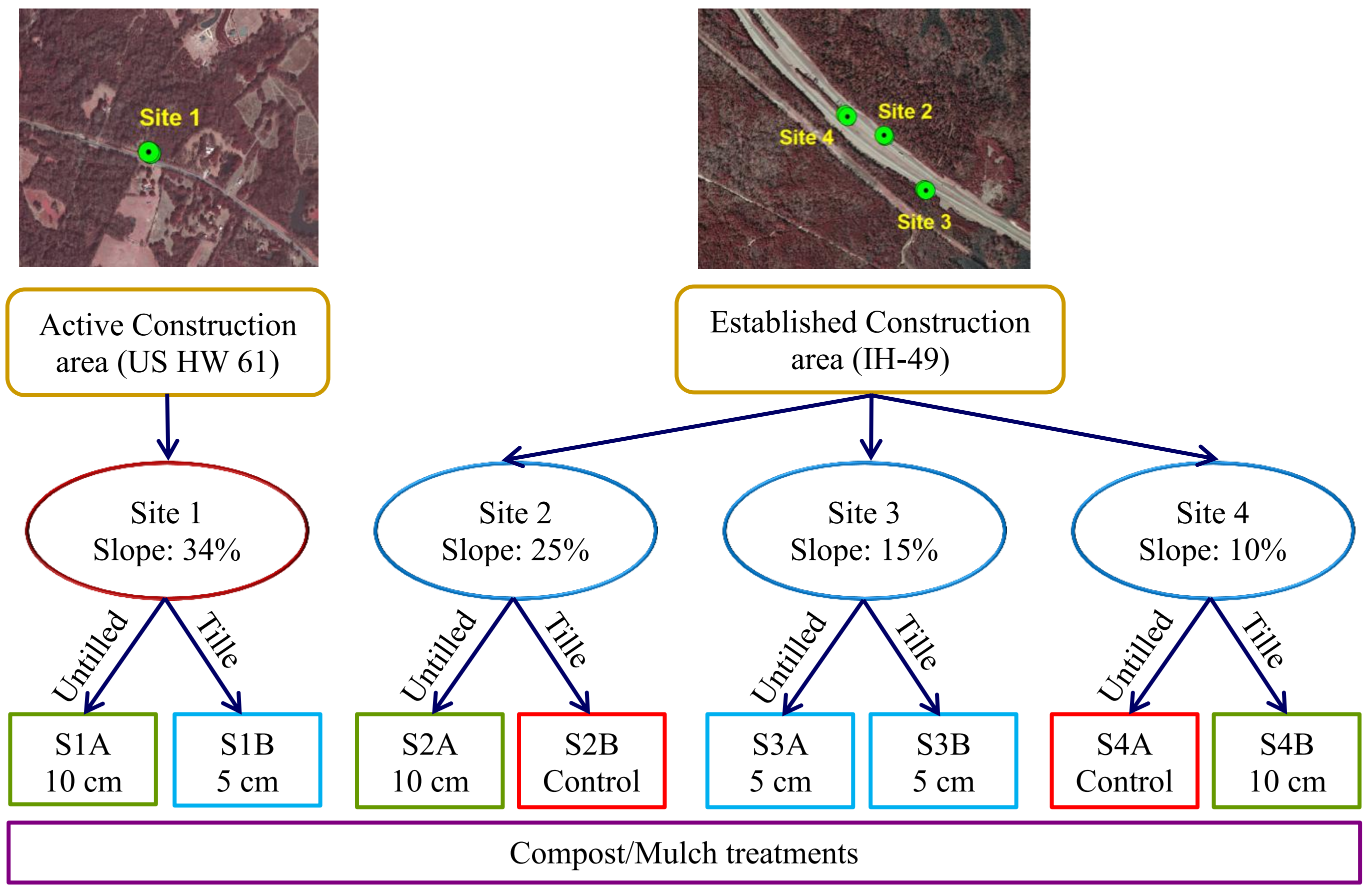

Figure 1.8. The scheme chart of the experiment design for the current study in Louisiana, USA.

Each plot was also supported by HOBO® Micro Stations (H21-002) Data Logging (Onset Computer Corporation, Bourne, MA, USA) that have four outlets. Two 12-Bit Temperature Smart Sensors (S-TMP-M006) and two Soil Moisture Smart Sensors(S-SMx-M005) were used to record surface soil temperature and moisture every 2 min and the average of 10 min were read. Each pair for soil temperature/moisture was placed up and down slope within each plot. The four sites with all instruments are presented in Figure 1.9.

Figure 1.9. The final shape for each site with the supported instrumentation. a) site 1, b) site 2, c) site 3, and d) site 4, Louisiana, USA.

1.4.2 Compost/mulch impacts on soil properties

Organic residues as well as tillage practices have strong potential to alter the physiochemical soil properties such as; bulk density, soil moisture and temperature, heat and solute movement through the soil (Dahiya *et al*., 2007). Tillage practices affect surface roughness, bulk density, and porosity of soil. Those three features impacts different soil properties, such as; water storage, infiltration, and surface runoff (Mwendera and Feyen, 1994). Organic residues used as mulching known to reduce soil evaporation, increase soil water,

decrease diurnal soil temperature variations and increase saturated soil hydraulic conductivity, allow for better water infiltration, improve surface soil stability, and enhance water use efficiency (Dahiya *et al*., 2003; Cook *et al*., 2006; Sarkar and Singh, 2007; Bunna *et al*., 2011).

According to Smets *et al*. (2008), a mulch cover has some important advantages regarding soil conservation and productivity. Mulch cover has been confirmed as a very effective management practice to decrease and control soil erosion by water, enhance soil physiochemical properties, and reduce the destructive effect of the raindrops. Mulch can conserve moisture, prevent surface compaction or crusting, reduce runoff and erosion, and to establish desired plant cover, on slopes of 3% or greater (LDEQ, 2012). Application of 3-inch (7.6 cm) layer of leaf mulching in New Jersey was improved soil aggregation and increased soil water holding capacity (Kluchinski *et al*., 2002). Ramakrishna *et al*. (2006) studied the effects of polythene mulch, straw mulch, and chemical mulch on soil temperature and moisture. Their results indicated that straw mulch was most desirable type economically and environmentally.

Udeigwe *et al*. (2007) studied nine different Louisiana cultivated agricultural soils, and the results indicated that the higher clay contents resulted in higher TSS and higher particulate phosphorus (PP) in the runoff with the relation

$$TSS = 0.01e^{0.01(Clay)} \qquad R^2 = 0.91 \qquad (1.6)$$

$$PP = 2.53e^{0.005(Clay)} \qquad R^2 = 0.87 \qquad (1.7)$$

However, soil EC inversely related to runoff TSS with the following relationship

$$TSS = 0.02(EC)^{-3.95} \qquad R^2 = 0.70 \qquad (1.8)$$

As erosion control, compost has been introduced to the eroded soils in different ways; it can be incorporated with the topsoil by tillage, applied as compost blanket, or implemented as a filter beam (ridge that is used to control sediment transport). Compost has a positive influence on the physical properties of soil (Arthur *et al*., 2011; Weindorf *et al*., 2006). Compost has been

successfully used to reduce the soil loss and soil erosion by water (Faucette *et al*., 2004; Persyn *et al*., 2004; Birt *et al*., 2007). Arthur *et al*. (2011) showed that there was a significant increase by 21% in the total carbon content due to compost application on the loamy sand soils. Weindorf *et al*. (2006) studied the effect of compost on the coefficient of linear extensibility (COLE), water content, and infiltration rate in different soils in Dallas. The results indicated that the compost incorporation significantly reduced the COLE and increased soil water content while the infiltration rate was not significantly affected by compost.

1.4.3 Compost/mulch effectiveness to sustain water quality

The national water quality handbook (NRCS, 2003), defined water quality as,

> The physical, chemical, and biological composition of water as related to its intended use for such purposes as drinking, recreation, irrigation, and fisheries.

Sanders et al. (1983) defined water quality management as,

> The management of the physical, chemical, and biological characteristics of water.

LDEQ (2010) stated that water quality criteria are;

> Elements of state water quality standards expressed as constituent concentrations, levels, or narrative statements representing the quality of water supporting a particular designated use. When criteria are met, water quality will protect the designated use.

According to Parparov and Gal (2012), the use of water resources that was once concerned with quantity of supplied water, has shifted to include water quality criteria. This subsequently led to development of the sustainable water management term (Kates *et al*., 2001; Kemp and Martens, 2007). Recently, water resources management conceptualization is evolving from focusing on inflows and /or outflows to include the holistic view of the entire ecosystems. Additionally, the water management tools that regulate the use of nutrient and toxicant pollutants have been alerted in respect to the increase of aquatic system resilience (Parparov and Gal, 2012; WFD, 2000; Carpenter and Cottingham, 1997; Folke *et al*., 2005).

Clean water is one of the major challenges that humanity faces it in the 21st century. Unsafe water has not only led to fish kills, adversely impacts on the aquatic life, and deteriorated water quality, but also it could cause major human diseases. Two main sources of pollution are responsible for water quality degradation, point and nonpoint sources. NPS means any source of water pollution that does not meet the legal definition of "point source" in section 502(14) of the Clean Water Act. NPS pollution comes from many diffuse sources as it caused by rainfall or snowmelt that runoff over and through the ground carrying natural and human-made pollutants then depositing them into surface and ground waters (EPA, 2012b). In 1987, the congress enacted Section 319 of the Clean Water Act (CWA) for establishing a national NPS management program to control NPS of water pollution (EPA, 2012c). Under Section 319(a), all states and territories have addressed, assessed, and identified NPS pollution problems for the water quality problems. Under Section 319(b), all states have adopted management programs to control NPS pollution. Since 1990, congress has annually appropriated grant funds to states under Section 319(h) to help them to implement those management programs (EPA, 2003). The main goal of applying those types of management programs on the highway roadsides, is to sustain the infrastructure while maintaining the quality of runoff water at an acceptable level.

LDEQ (2010) provided a list of the suspected causes of impairment to water quality in Louisiana. Low dissolved oxygen, mercury, Coliform bacteria, turbidity, total dissolved solids, nitrate/nitrite, phosphorous, and total suspended solids, were the most suspected cause of impairment in Louisiana’s water bodies (Table, 1.2).

According to LDEQ (2010), the NPS pollution represented the largest percentage of the reported suspected sources (444) of impairment. NPS pollution consists of those forms of pollution caused by the runoff of stormwater from land such as agricultural fields, forestry areas, construction sites, and urban areas. Additionally, construction activities involve clearing land and

moving soils prior to and during construction. Consequently, the major water pollutant that is generated is sediment which adversely affects soil and water quality. The erosion rates from construction sites are higher compared to the rates from cultivated lands. Soil loss from new development can range from 20 to 150 tons per year, whereas the national average for cropland is 8 tons per year (LDEQ, 2010).

Table 1.2. Total number of water bodies impacted by each suspected cause of impairment, Louisiana, USA. (Updated from: 2010 Louisiana Integrated Report assessment).

Suspected causes of impairment	Total number of water bodies
Dissolved oxygen	190
Mercury in fish tissue	103
Fecal Coliform bacteria	96
Turbidity	88
Total dissolved solids (TDS)	66
Nitrate/Nitrite	57
Phosphorous (P)	55
Total suspended solids (TSS)	51
Non-native aquatic plants	43
Sulfates	42
Sedimentation/Siltation	35
Chloride	33

Sediment has two main forms in the surface water; it can be suspended in the water column or settle on the bottom of a waterbody. Sediment transport process includes three steps; eroding from one place, carrying in the flow, and depositing in another place (Ji, 2008). The TSS concentration is determined as a dry-weight of sediment in the water samples and expressed as mg L^{-1}. The Environmental Sciences Section (ESS) Method 340.2 has been commonly used for TSS (EPA, 1993). The visual effect of high sediment concentrations in water samples is turbidity that blocks light penetration inon the water body and adversely affects the recreational water activities and aquatic life. Turbidity measurements are commonly achieved using a turbidimeter which gives reading by Nephelometric Turbidity Units (NTUs). Dissolved oxygen (DO) is considered as the most important parameter of water quality in Louisiana (LDEQ, 2010). The BOD is usually used to measure the amount of oxygen consumed by biochemical activity in

water bodies. The membrane-cover polarographic electrod (probe methods) has been commonly used to measure DO consumed during a specific incubation time (mainly 5 days, BOD_5) at a specific temperature. BOD_5 concentration (mg L^{-1}) is determined by the difference between the DO concentrations before and after incubation time based on Method 5210 B in APHA (2005).

As land management practices strongly impact soil erosion and sediment yield, sediment and erosion control programs are needed as BMPs. Using mulch as a BMP for a construction sites, is recommended by LDEQ (2012). The types of materials that are suitable for mulching include; wood waste and shredded residues, upholster's Burlap, wood cellulose fiber (Hydromulching), straw or hay, and commercial mulch. Based on Louisiana standard specifications for roads and bridges (DOTD, 2006), mulch shall consist of either tacked vegetative mulch or an approved fiber mulch product. Tacked vegetative mulch consists of pine straw, stems or stalks of oats, rye, rice, or other approved straws, or hay. Fiber mulch products consist of organic fiber mulches. Mulch effectiveness depends on many factors, such as; slope gradient, soil type, rainfall erosivity, type of mulch materials, rate of mulch application, and plot size. According to Smets *et al*. (2008), the relationship between mulch cover and the erosion rate has been expressed by many authors as

$$SL = ae^{(-bC)} \tag{1.9}$$

where, SL is the soil erosion rate, C is the mulch cover (%), a and b are constant with b is a coefficient describing the effectiveness of a given mulch cover in reducing SL and has range between 0.01-0.1 (Brown *et al.*, 1989). The ratio of SL during the presence of mulch to SL without mulch (control) is defined as a mulch factor (MF) (Smets *et al.*, 2008),

$$MF = e^{(-bC)} \tag{1.10}$$

Composting is one component in USEPA’s hierarchy of integrated solid waste management. It involves the aerobic biological decomposition of organic materials to produce a

stable humus-like product. The composting process should be viewed as an environmentally sound and beneficial means of re-cycling organic materials, not a means of waste disposal (EPA, 1995). Although erosion is a natural process, road building and new construction activities can aggravate it. When the construction activities have been initiated, all vegetation and topsoil is removed, leaving the soil vulnerable to erosion. On highways roadsides, compost can be an effective BMP to reduce erosion. Based on roadside slope, a 2- to 3-inch (5- to 7.6 cm) layer of compost could be placed on top of the soil to control erosion. Because of its ability to retain moisture, compost also helps protect soil from wind erosion and during droughts (EPA, 1997).

Storey *et al*. (1996) conducted a project entitled "The use of compost and shredded brush on right-of-way for erosion control" in Texas, USA. The results indicated that the use of compost/mulch to control erosion from the highway right-of-ways was most effective economically and environmentally. The material cost of compost was three times the cost of average mulch. To minimize the cost, the mulch, which was mainly wood chip that has been taken from right-of-way cleaning operations, was successfully used.

1.4.4 Compost/mulch impacts on water runoff, flow rate, and soil loss

Intensive rainfall is a frequently event in the southern part of the U.S. Associated with subtropical climate in such area, this event could cause many environmental problems; e.g. flooding, tropical storm, and water erosion. Erosion rates are a function of rain splash and runoff and are related to slope. With low slopes, rain splash is the dominant factor causing erosion, and with higher slope runoff is the dominant factor (Battany and Grismer, 2000). Evaluation of runoff and flow rate caused by water erosion in the field is usually expensive and/or time-consuming. When the rainfall intensity exceeds the infiltration rate of soil, the overland flow or runoff could occur (Ji, 2008; Battany and Grismer, 2000). Runoff, flow rate, as well as soil loss may be assessed on a study plot at different scales. Also, simulated rainfall has been widely used

to evaluate soil infiltration rate, sediment loss, and rainfall–runoff relationships in different regions around the world. Sharpley and Kleinman (2003) studied the effect of plot length on overland flow and phosphorus (P) transport using simulated rain. The results indicated that plot length influences hydrology, sediment discharge, and concentration of P in overland flow.

On road systems, large volumes of high velocity runoff may be produced and moved to down streams. Roadsides, mainly when associated with steep slopes, are usually susceptible to hydraulic erosion processes, and may contribute substantially to stream sedimentation, even during low magnitude rainfall events (Ziegler *et al*., 2001). In order to reduce the runoff velocity and allow the rainfall to penetrate soil surface, use of surface coverage is required. Grismer and Hogan (2004) used a portable rainfall simulator for assessing the impacts of revegetation/mulch, as a BMP, on soil infiltration, runoff, and sediment yields from disturbed road-cut soils. Their results indicated that the infiltration rates were increased and the runoff rates were reduced with mulch treatment.

1.5 OBJECTIVES

The main goal for this research was to provide the most recent advances in environmental techniques for sustainable utilization and protection of vulnerable natural resources, namely soil and water resources, in semiarid and humid ecosystems during the short and long term. This goal was achieved through two main studies. First, sustainable land use management was evaluated for a newly reclaimed area in semiarid region in the *Bustan 3* area, Egypt. Second, soil and water conservation programs on the roadsides were examined by studying the effect of compost/mulch as a BMP for soil erosion control on two highways in Louisiana, USA, as a humid region. One question needs to be answered form presented research; how did the management strategies conserve our natural resources?

In order to achieve the first part of our project three objectives were carried out, they aimed to: 1) Monitor the changes in LULC in response to human-induced changes and management practices in the Bustan 3 area, Egypt, 2) Detect the most vulnerable areas to desertification within the study area, and 3) Examine the land capability for agricultural use and implement different management scenarios that help the decision-maker to choose the most appropriate management practices for sustaining natural resources in the Bustan 3 area, Egypt. For the second part of this project, three objectives were accomplished aiming to evaluate the effect of compost/mulch on: 1) soil physiochemical properties, 2) runoff water quality, and 3) total runoff, flow rate, and soil loss under a simulated rainfall, on the roadsides in Louisiana, USA. With the hypothesis of using the compost/mulch as a BMP can improve soil properties, control water quality of the runoff, and decrease the runoff rates.

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CHAPTER 2. MONITORING LAND COVER CHANGES IN A NEWLY RECLAIMED AREA OF EGYPT USING MULTI-TEMPORAL LANDSAT DATA[1]

2.1 INTRODUCTION

Changes in the earth's surface can be related to natural dynamics or human activities and can occur either suddenly or gradually (Coppin *et al.,* 2004). Timely and accurate change detection of Earth's surface features provides a better understanding of the interactions between human and natural phenomena to better manage and use resources (Lu *et al.,* 2004). Two of the most common uses of satellite images are mapping land cover via image classification and land cover change via change detection (Song *et al.,* 2001). Landsat satellite data is the most widely used data type for land cover mapping and has provided earth observation data to meet a wide range of information needs since 1972 (Williams *et al.*, 2006). The availability of Landsat data in the Geocover dataset and the United States Geological Survey's (USGS) decision to provide free access to all Landsat data holdings offer opportunities for land cover classifications using Landsat imagery (Knorn *et al.,* 2009).

Change detection can be performed by supervised or unsupervised approaches (Singh, 1989). A supervised technique requires ground truth points to derive training sets containing information about the spectral signatures of the changes that occur in the considered area between two dates. An unsupervised technique performs change detection without any additional information besides the raw images considered; however, it is also fraught with some critical limitations (Bruzzone and Prieto, 2002). As these basic approaches have limited utility independently, a hybrid classification method was used in order to obtain both high change accuracy and efficiency (Schowengerdt, 2007).

[1] Reprinted from Applied Geography, Vol. 30 (4), Bakr, N., D.C. Weindorf, M.H. Bahnassy, S.M. Marei, and M.M. El-Badawi, Monitoring land cover change in a newly reclaimed area of Egypt using multi-temporal Landsat data, 592-605, 2010, with permission from Elsevier.

Vegetation index differencing is often regarded as an effective method to enhance the difference among spectral features (Lu *et al.*, 2005). The normalized difference vegetation index (NDVI) is often used to monitor vegetation dynamics (e.g. Julien *et al.*, 2006; Myneni *et al.*, 1997; and Zhou *et al.*, 2001). It has been shown to be highly correlated with plant health, vegetation density, and cover (Ormsby *et al.*, 1987). The NDVI can be used as a general indicator of vegetation cover and vigor, however, in a single NDVI image; barren fields are indistinguishable from temporarily fallow, yet healthy fields. Moreover, immature crops with low density cover could be confused with poor crops (Wallace *et al.*, 1993). Furthermore, in a two-image change detection analysis, the effects of crop rotation could be erroneously identified as land cover change (Pax Lenney *et al.*, 1996). Finally, NDVI appears to be a poor indicator of vegetation biomass if it is low, as is common in arid and semi-arid regions (Huete and Jackson, 1987). Various researchers have discussed land use/land cover changes (e.g. Dewan and Yamaguchi, 2009; Gao *et al.*, 2006; Kaiser, 2009; Li *et al.*, 2001; Serra *et al.*, 2008; and Siren and Brondizio, 2009). Particularly, numerous studies have discussed land cover changes in agricultural lands in arid and semi-arid regions. Pilon *et al.* (1988) identified agricultural expansion as one type of land use change in Nigeria using a combination of post classification comparison (PCC) and spectral comparisons between two multispectral scanner (MSS) images. Castellana *et al.* (2007) presented a new approach to perform change detection analyses based on a combination of supervised and unsupervised techniques. Julien and Sobrino (2009) presented a new method for monitoring vegetation by using NDVI and land surface temperature data. In Egypt, Kishk (1986) and Metz (1991) indicated that there is little consensus among real estimates of cultivated lands in Egypt due to the difficulty in: 1) identifying the actual areal extent of these lands, and 2) determining the quality of these lands. Sadek (1993) mapped the expansion of deltaic agricultural lands into the adjacent deserts by tracing the boundaries of

cultivated lands on satellite imagery to monitor the reclamation process. Pax Lenney *et al.* (1996) used field calibrated multi-temporal NDVI features derived from ten Landsat TM images from 1984 to 1993 to assess the status of agricultural lands in the Nile Delta, adjacent Western Desert, and coastal regions, in Egypt. Shalaby and Tateishi (2007) used maximum likelihood supervised classification and PCC change detection techniques to map land cover changes on the Northwestern coast of Egypt using Landsat images acquired in 1987 and 2001, respectively.

The total area of Egypt is around one million square kilometers. Approximately 95% of population lives on only 4% of the land. Since the 1980s, the Egyptian government has advocated policies aimed at extending cultivated land and maximizing production of the existing agricultural lands. Thus, determination of the trend and rate of land cover conversion are required for the development of rational land use policy (Shalaby and Tateishi, 2007). Satellite remote sensing offers the opportunity to assess the effects of these processes and provide the data needed for the development of national agricultural strategies (Pax Lenney *et al*., 1996).

The objectives of this study were to: 1) provide a recent perspective for different land cover types in the *Bustan 3* area, Egypt, and 2) monitor land cover changes that have taken place from 1984 to 2008 in the *Bustan 3* area, using the hybrid classification technique and NDVI.

2.2 STUDY SITE

The desert region of the West Delta in Egypt includes a total land reclamation area of 2346 km^2, with 823.2 km^2 reclaimed prior to 1978. The Egyptian government plans for 60% of the reclaimed area to be auctioned to investors and 40% to be distributed on concessionary terms to small farmers and unemployed graduates. The Newlands Agricultural Services Project (NLASP) area comprises 789.6 km^2 of recently reclaimed land which was allocated in 0.021 km^2 parcels to selected settlers in three separate localities: *West Nubaria*, *El Bustan*, and *Sugar Beet* areas. The reclamation process includes the installation of irrigation and drainage systems, the

construction of roads, houses, community buildings, a supply of portable water, and electricity (IFAD, 1992). The *El Bustan* extension area (*Bustan 3* area) is one of the Newlands in the West Delta reclamation zone, Egypt (Figure 2.1). It is located in UTM zone 36 between 226,000 m and 255,500 m Easting (Longitude: 30° 80´ 30" to 30° 27´ E) and 336,8500 m–339,2000 m Northing (Latitude: 30° 26´ to 30° 39´ N), and occupies 341.27 km^2.

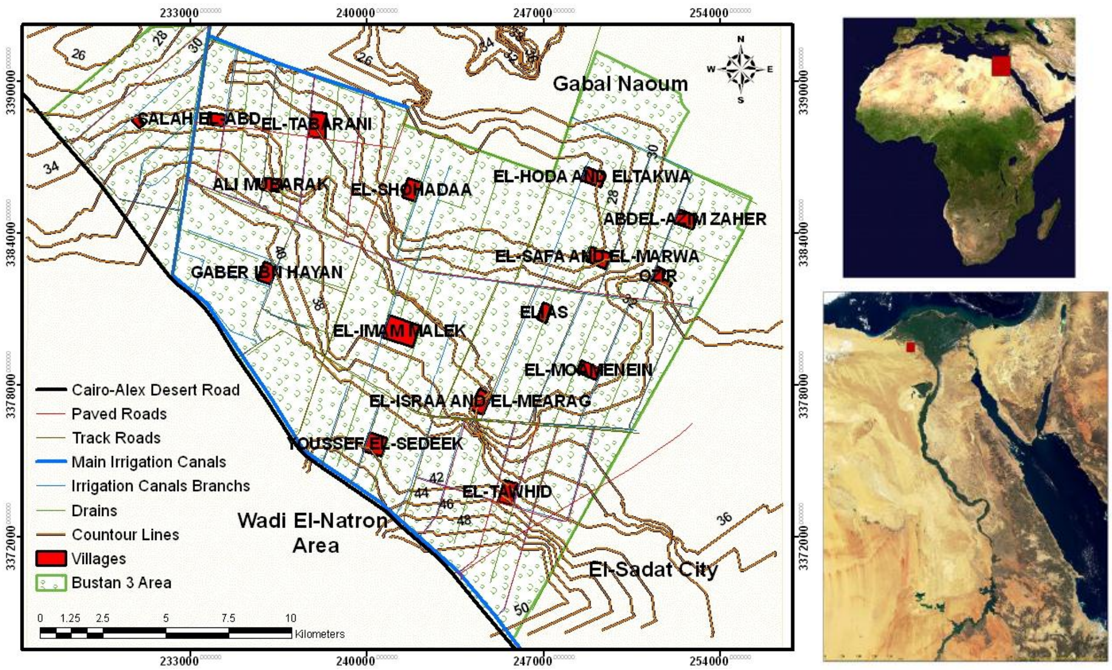

Figure 2.1. General location and main infrastructure of the *Bustan 3* area, Egypt.

The *Bustan 3* area is characterized by a semi-arid, Mediterranean climate. Climate data was collected from the Tahrir meteorological station (latitude: 30º 39´ N, longitude: 30º 42´ E, elevation: 16 m). Average climatic data across thirty years show that the maximum and minimum temperatures occur in August (35ºC) and January (8ºC), respectively. Rainfall occurs during the winter months, from October to March, with total precipitation around 35 mm y^{-1}. Relative humidity varies from 6 6% in May to 80% in January with an average of 73.5%. The average of wind speed and sunshine are 264.8 km d^{-1} and 9.4 h, respectively (FAO, 1993). The *Bustan 3* soils are mostly classified as Typic Torripsamments (Soil Survey Staff, 2006). Sadek

(1993) reported that this area contains desert geomorphic units such as sand dunes and sandy plains. The geological deposits represent the Pliocene, Holocene, and Pleistocene eras.

Bakr *et al.* (2009) reported that the general land capability classes of *Bustan 3* area range from class S2, land with good capability for agriculture practices, to class N, marginal lands. About 70% of this area has a good capability for agricultural production and could be increased to 96% when best management practices are applied.

2.3 DATA SETS

Landsat 4, 5, and 7 satellites that maintain near polar, sun-synchronous orbit were used for this research. Images are acquired nominally at 9:30 am local time on a descending path. The orbit altitude is 705 km and provides a 16-day, 233-orbit's cycle. The swath width is 185 km and the image size is 185 km X 170 km. Landsat 4 and 5 carry TM and MSS sensors. The MSS was the principal sensor on Landsat 1, 2, and 3. The instantaneous field of view (IFOV) of the MSS on Landsat 4 and 5 have been modified to 81.5 m and 82.5 m, respectively, although the pixel center spacing of 56 m has been retained. Additionally, bands have been renamed: bands 1, 2, 3 and 4, correspond to bands 4, 5, 6 and 7 from earlier missions. Landsat TM data has 30 m resolution in six spectral bands ranging from blue to middle infrared and 120 m resolution in one thermal-infrared band (Markham *et al.*, 2004). Landsat 7 was launched in April 1999, carrying the previous TM sensors with the new ETM^+ instrument. The latter sensor is similar to TM, but has an additional grayscale (panchromatic) band which collects imagery at 15 m resolution, and also a thermal-infrared band with resolution improved to 60 m. Despite the scan line corrector (SLC) failure, the USGS maintains delivery of data from Landsat 7 (Cohen and Goward, 2004). Five Landsat 4, 5, and 7 images were selected to support the time series analysis in this research: 1984, 1990, 1999, 2004, and 2008 (Table 2.1). All data scenes were acquired under clear

atmospheric conditions during the dry season, a time when the weather is generally cloud-free (during July and early August for each year).

Table 2.1. Satellite image information for data acquisitions corresponding to path 177/row 39.

Imagery date	Spatial resolution	Satellite/sensor	No. of bands	Scene identifier	Format
07/09/1984	60 m	Landsat-5 MSS	4	LM51770391984191AAA03	GeoTIFF
08/03/1990	30 m	Landsat-4 TM	7	LT41770391990215AAA03	GeoTIFF
07/11/1999	30 m	Landsat-7 ETM^+	7	L71177039_03919990711	FAST-L7A
07/08/2004	30 m	Landsat-7 ETM^+	7	LE71770392004190ASN01	GeoTIFF
07/03/2008	30 m	Landsat-7 ETM^+	7	LE71770392008185ASN00	GeoTIFF

Three topographic maps sheets at a scale of 1:50,000 cover the *Bustan 3* area, Egypt. These maps were digitized from paper maps to produce the infrastructure map of the *Bustan 3* area (Figure 2.1). The digitized map was used also to perform the geometric correction of the satellite images and to confirm ground truth information. Forty-eight ground truth points were collected from the *Bustan 3* area during July 2007 to provide information about the land cover types at that time. These points were used to examine the training sets, during the supervised classification (essential for the 2008 image).

2.4 METHODS

2.4.1 Data pre-processing

All satellite imagery data was geometrically corrected to the projection: UTM zone 36 and Datum: D_Egypt_1907. Geometric correction was done using both digitized topographic map and ground control points (GCP) to register the 2008 image. Other images were co-registered using this image. The root mean-square error (RMSE) between the 2008 image and other images was around 0.3 pixel which is acceptable according to Lunetta and Elvidge (1998) who reported that the accepted RMSE between any two dates should not be more than 0.5 pixel. Atmospheric correction is not required for some remote sensing applications such as in change

detection and also image classification with a maximum likelihood classifier using a single date image. As long as the training data and the image to be classified are on the same relative scale (corrected or uncorrected), atmospheric correction has little effect on classification accuracy. Thus atmospheric correction for a single date image is often equivalent to subtracting a constant from all pixels in a spectral band (Song *et al*., 2001). In addition, all images were acquired during the summer season with 0% cloud coverage.

According to Scepan *et al.* (1999), the most useful band combinations in Landsat for discrimination of land cover categories are bands 4-5-3, 4-3-2, and 3-2-1 assigned as red, green, and blue, respectively. Band 5 of Landsat is sensitive to variations in vegetative water content and soil moisture. It is also provides a good contrast between different types of vegetation. Therefore, a combination of bands 4 (NIR), 5 (MIR), and 3 (Red) is good for the analysis of soil moisture and vegetation conditions for this area, where new irrigated areas can be found and different vegetation densities exist. All images were subset and masked to the boundary of the *Bustan 3* area. All images use bands 4-5-3; except for the MSS 1984 image because it contains only four bands, therefore a band combination of 4-3-2 was used (Figure 2.2). All processing was completed using ERDAS IMAGINE 9.2 (Leica Geosystems, 2008) software.

The Landsat 7 ETM$^+$ experienced a failure of its SLC on May 31, 2003 and is now permanently disabled. Beginning in May 2004, USGS began providing the first in a series of data products to help make the SLC-off data more usable. SLC-off data are composited products based on two or more SLC-off scenes acquired within a short period of time, within or during a month (Markham *et al.,* 2004). Filling the scan gap first requires precise knowledge of what pixels are valid in an image and which are to be filled. The gap filling was achieved using a spatial modular for both ETM$^+$ 2004 and ETM$^+$ 2008 images, using an ETM$^+$ July 24, 2004 image and an ETM$^+$ June 1, 2008 image, respectively.

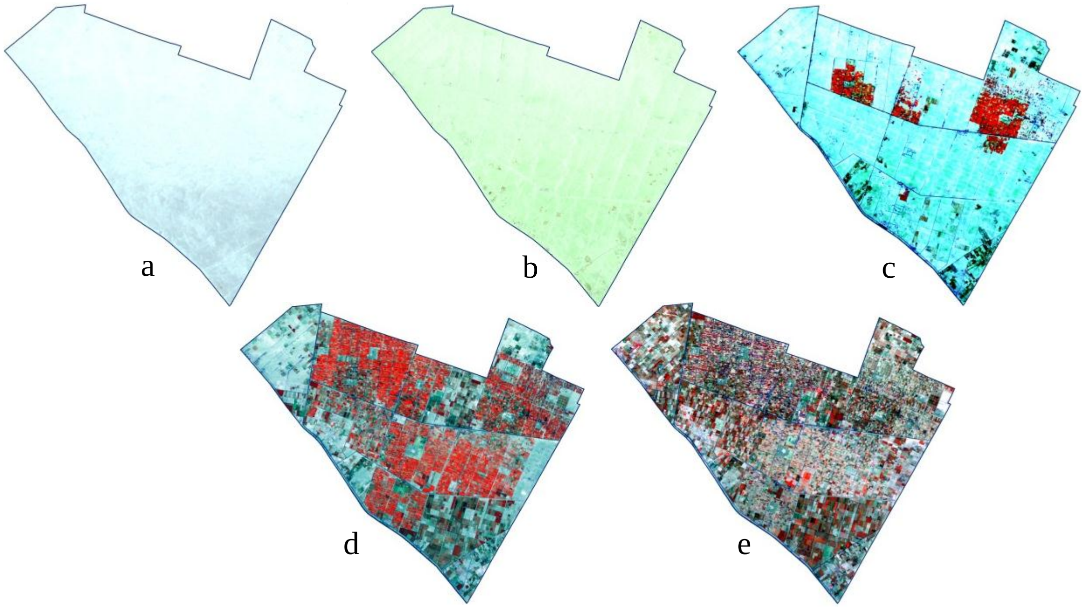

Figure 2.2. Landsat images for the Bustan 3 area, Egypt. (a) MSS 1984, bands 4-3-2 were assigned as RGB; (b) TM 1990; (c) ETM$^+$ 1999; (d) ETM$^+$ 2004; and (e) ETM$^+$ 2008: all images use bands 4-5-3 assigned as RGB.

2.4.2 Hybrid classification methodology

2.4.2.1 Supervised procedure

A hierarchical land cover classification system (Anderson *et al.,* 1976) was used to detect the different land cover classes in the *Bustan 3* area, Egypt. According to this system, four land cover categories exist in this area: 1) urban or built-up land, 2) agricultural land, 3) water, and 4) barren land. Different training sets were delineated for each land cover class and verified through a digital topographic map, ground truth points, and the visual interpretation of different images. Using the training sets, various spectral signatures for each class were developed and evaluated using separability analysis to estimate the expected error in the classification for various feature combinations (Landgrebe, 2003). Using a separability cell array, different spectral signatures in each class were merged together (Jensen, 2004).

The maximum likelihood decision rule, the most common supervised classification method used with remotely sensed imagery data (Richards and Jia, 2006), was used as a

parametric rule. The basis of the maximum likelihood classifier is the probability density function (PDF), which depends on the Mahalanobis distance between each pixel and the centroid of the belonging class. When the maximum likelihood classification was performed, a distance image file, Mahalanobis distance, was produced in addition to thematic layer output. Those outputs were used to create the threshold image, which was accomplished for identifying the pixels that are the most likely to be classified incorrectly and placed in class zero. The 'water class' for canals makes manual training set delineation difficult (Buchheim and Lillesand, 1989) so, it appeared in threshold images as unclassified data. Consequently, the unclassified mask was created to add the 'water class' to the threshold image.

2.4.2.2 Unsupervised procedure

The unsupervised classification approach is commonly called clustering, because it is based on the natural groupings of pixels in image data. After the classification is complete, the analyst employs posteriori knowledge to label the spectral classes into information classes (Thapa and Murayama, 2009). The iterative self-organizing data analysis technique (ISODATA) clustering method (Tou and Gonzalez, 1974) was used to accomplish the unsupervised classification. In this clustering method, pixels belonging to a particular cluster are spectrally similar, and the most frequently similar matrix encountered is Euclidean distance (Richards and Jia, 2006). Twenty-five spectral clusters for each image in each year were formed to separate the image information into a more readable form with a 0.97 convergence threshold. The ISODATA signatures were evaluated according to homogeneity, normality, and count. Then, a supervised algorithm was applied to the ISODATA signatures file. The parallelepiped decision rules as the non-parametric rule and maximum likelihood as the parametric rule were chosen. A supervised algorithm was applied only on the unclassified mask images; which resulted from the supervised

procedure in the previous step for each year. This limited the analysis to only the unclassified pixels produced from the supervised procedure.

2.4.2.3 Composite procedure

Both the threshold image (from supervised classification) and the image that resulted from unsupervised classification, for each date, were recoded to the same number of classes. This was done by identifying a real number for each class in all images (1 for urban or built-up land, 2 for agricultural land, 3 for water, and 4 for barren land). Each class was represented by a group of spectral signatures in the threshold image, or a group of clusters in ISODATA's result image. Thus, each group under the same class used the same number. Between each two dates, 1984–1990, 1990–1999, 1999–2004, and 2004–2008; the spatial modular was used to add each pair of images together, using the simple addition function. Some of classified pixels in one image, which were assigned numbers from 1 to 4, might be unclassified pixels in another image (assigned 0 value). The final thematic classified image for each date contains four land cover classes: urban or built-up land, agricultural land, water, and barren land.

2.4.3 NDVI

The NDVI is the most widely used index in the processing of satellite data (Myneni *et al.*, 1995; Tucker, 1979). It is defined by Rouse *et al.*, (1974) as; NDVI= (NIR-R) / (NIR+R). Where, NIR is near infrared (NIR) band and R is red (R) band. The NDVI values range from -1 to +1. Krishnaswamy *et al.* (2009) stated that, values 0 represent water and non-vegetated areas, while values >0 represent vegetation. The NDVI was calculated for each image at each date using band 3 (R) and band 4 (NIR) in each image. Five NDVI continuous images, for all dates, resulted from this step with float data type (continuous real numbers). Each image at each date was recoded to only two values: 0 and 1. Zero for the non-vegetated land and one for vegetated land.

2.4.4 Accuracy assessment

Accuracy was determined empirically, by independently selecting two-hundred random samples of pixels from each resulting map, from each technique at each date, and checking their labels against classes determined from reference data. The results were expressed in tabular form known as the error matrix. The error matrix has been previously presented by Congalton (1991). Two different measures can be derived from the values in an error matrix: user's and producer's accuracy (Campbell, 2002; Story and Congalton, 1986). The user's accuracy is the number of correctly identified sets in one class divided by the total number of sets recognized in that class. Inaccuracies insets are referred to as errors of commission. The producer's accuracy is calculated by dividing the number of correct pixels in one class by the total number of pixels derived from reference data and includes the error of omission. A Kappa coefficient (K^) is commonly used as a measure of map accuracy (Congalton and Green, 1999; Hudson and Ramm, 1987). It has become a widely used measure for classification accuracy and was recommended as a standard by Rosenfield and FitzpatrickLins (1986). Typically, the specified requirements take the form of a minimum level of overall accuracy (Foody, 2002). Thomlinson *et al.* (1999) set an overall accuracy target of 85% with no class less than 70% accuracy. The USGS proposed an accuracy level of 85% as the minimum requirement for land use/cover mapping with Landsat data (Anderson *et al.*, 1976).

2.4.5 Monitoring land cover changes

The thematic images resulting from the hybrid classification procedure and the continuous images resulting from the NDVI analysis were used to monitor and detect the changes in land cover classes in the *Bustan 3* area during different time series from 1984 to 2008. Between each two evaluated dates, a change map was produced by performing a logical intersection/cross-tabulation function. For the thematic images, changes in the four land cover

classes were detected. Conversely, for the NDVI images, only changes between non-vegetated and vegetated lands were identified. Presently, most image processing systems are integrated or compatible with GIS systems. Various classifications of remotely sensed data are commonly inputted to GIS systems (Coppin *et al.,* 2004, Michalak, 1993). Consequently, all images were imported to ArcGIS 9.2 software (ESRI, 2001) for display and presentation of the final results.

2.5 RESULTS AND DISCUSSION

2.5.1 Hybrid classification results

For each date, four land cover classes were examined in the *Bustan 3* area, Egypt: urban or built-up land, agricultural land, water, and barren land. Figure 2.3 and Table 2.2 explain the thematic classified images from the hybrid classification technique and the area coverage for each land cover class by square kilometer and percentage across several dates, respectively.

The results show that in 1984 and 1990 the barren land dominated *Bustan 3* area with coverage of 100% (341.27 km^2) and 99.65% (340.08 km^2), respectively. After the reclamation work of the 1990s, other land cover types could also be identified. In 1999, water and urban land occupied 3% (10.71 km^2) and 0.26% (0.9 km^2), respectively. Barren land covered 84.21% (287.40 km^2) and agricultural land occupied 12.38% (42.26 km^2). As a result of agricultural development in this area, a dramatic increase in agricultural land was observed in 2004 and 2008. During 2004, agricultural land covered 68.65% (234.30 km^2) of the *Bustan 3* area and barren land coverage was only 26.16% (89.29 km^2). The urban land and water grew to 1.32% (4.50 km^2) and 3.86% (13.19 km^2), respectively. In 2008, similar trends were observed. The agricultural land increased to cover 78.80% (268.92 km^2) of the area while barren land decreased to 15.97% (54.51 km^2). Water coverage was almost the same as in 2004 whereas the urban areas increased slightly to 1.37% (4.67 km^2).

Table 2.2. The area coverage by square kilometer and percentage for each land cover class on different dates according to thematic classified images in *Bustan 3* area, Egypt.

Year	Unit	Land cover classes				Total
		Urban or built-up land	Agricultural land	Water	Barren land	
1984	km^2	0.00	0.00	0.00	341.27	341.27
	%	0.00	0.00	0.00	100.00	100.00
1990	km^2	0.00	1.19	0.00	340.08	341.27
	%	0.00	0.35	0.00	99.65	100.00
1999	km^2	0.90	42.26	10.71	287.40	341.27
	%	0.26	12.38	3.14	84.21	100.00
2004	km^2	4.50	234.30	13.19	89.29	341.27
	%	1.32	68.65	3.86	26.16	100.00
2008	km^2	4.67	268.92	13.18	54.51	341.27
	%	1.37	78.80	3.86	15.97	100.00

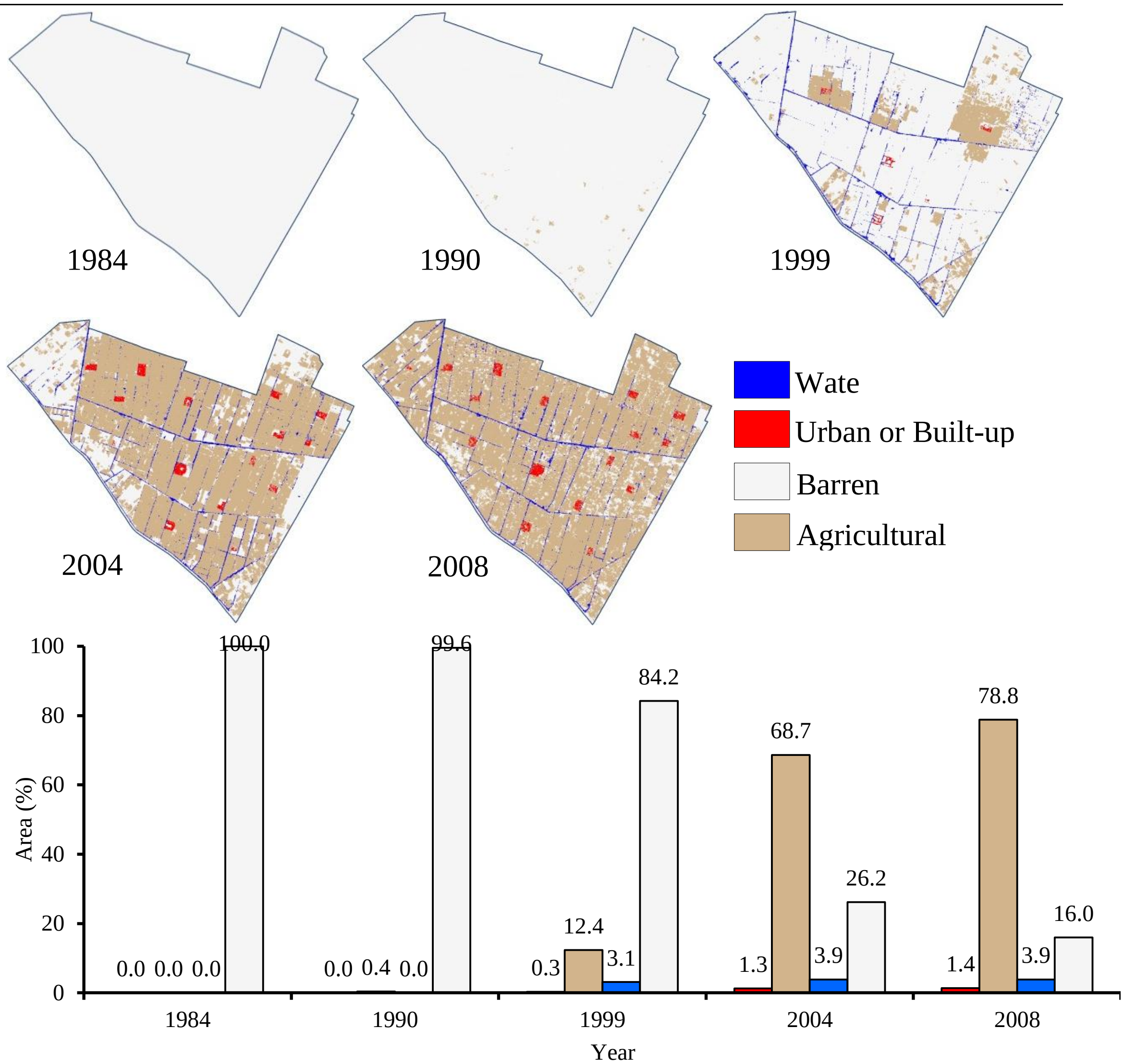

Figure 2.3. Thematic images represent the spatial distribution of different land cover classes on different dates in the *Bustan 3* area, Egypt. The chart explains the area by percentage of each land cover class on different dates.

2.5.2 NDVI results

The NDVI values were divided into two main classes: non-vegetated and vegetated lands. The NDVI negative values and zero represented non-vegetated land (urban land, water, and barren land), while positive values represented vegetated land (agricultural land). Figure 2.4 and Table 2.3 explain the NDVI continuous images and the area coverage for both classes by square kilometer and percentage on different dates, respectively.

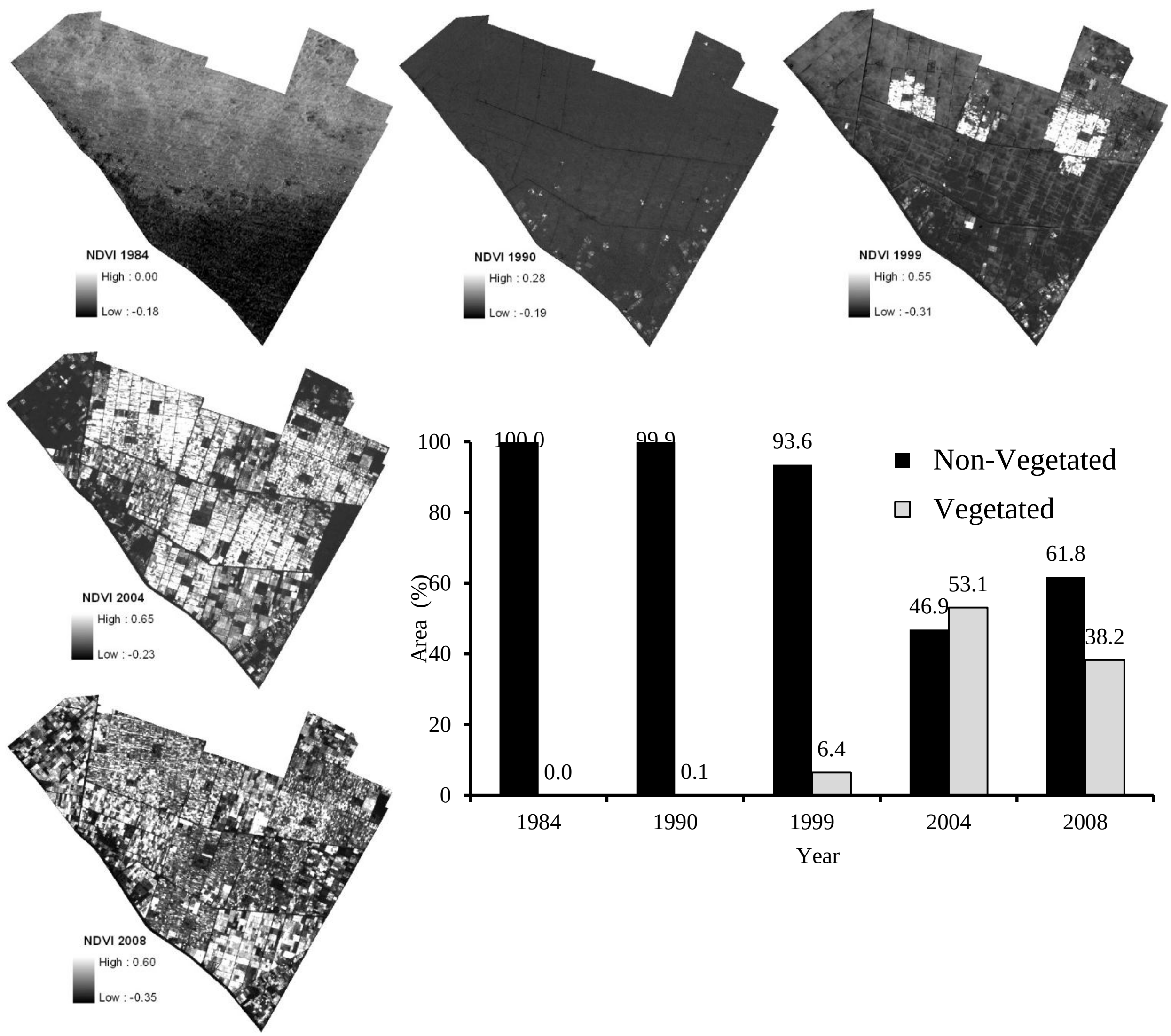

Figure 2.4. Continuous images represent the spatial distribution of NDVI values on different dates in the *Bustan 3* area, Egypt. The chart explains the area by percentage of non-vegetated and vegetated lands on different dates.

Table 2.3. The area coverage by square kilometer and percentage for each NDVI classes on different dates according to continuous classified images in *Bustan 3* area, Egypt.

Year	Unit	NDVI classes		Total
		Non-vegetated	Vegetated	
1984	km^2	341.27	0.00	341.27
	%	100.00	0.00	100.00
1990	km^2	341.00	0.27	341.27
	%	99.92	0.08	100.00
1999	km^2	319.31	21.96	341.27
	%	93.56	6.44	100.00
2004	km^2	159.97	181.30	341.27
	%	46.87	53.13	100.00
2008	km^2	210.76	130.51	341.27
	%	61.76	38.24	100.00

The results showed that in 1984 and 1990, the *Bustan 3* area were 100% and 99.92% (341.00 km^2) non-vegetated land, respectively. By 1999, the non-vegetated land coverage decreased to 93.56% (319.31 km^2) while the vegetated land grew to 6.44% (21.96 km^2). By 2004, non-vegetated land decreased to 46.88% (159.98 km^2) while the remaining area, 53.12% (181.30 km^2) was vegetated. Some erroneous patterns were discovered in the 2008 results, specifically, an increase in non-vegetated land and a decrease in vegetated land. However, ground truth points proved that this pattern was false. In reality, vegetated land increased and non-vegetated land decreased. These results are consistent with the literature (e.g. Huete and Jackson, 1987; Maselli, 2004; Pax Lenney *et al.*, 1996; Wallace *et al.*, 1993). All of these researchers proved that NDVI values for barren fields are indistinguishable from vegetated fields whenever the vegetation density is low or the fields are temporarily fallow. This scenario perfectly illustrates the erroneous pattern discovered in the 2008 the *Bustan 3* area data. Even though the land was vegetated, the NDVI analysis classified the land as non-vegetated.

2.5.3 Classification accuracy assessment results

To validate the accuracy of the classification procedures as well as the user who achieved the classification, accuracy assessment analysis was completed. Tables 2.4 and 2.5 identify the

error matrices for both the images developed from the hybrid classification technique and those which resulted from NDVI analysis on different dates. In Table 2.4, the overall classification accuracy and K^ statistics were 100% and 1, respectively, in 1984. This is true because, during this period, only barren land existed. A similar state was observed in 1990. By 1999, four land cover classes were distinguished and the overall accuracy and K^ statistics became 96.5% and 0.9, respectively. In 2004, the impressive change among the four classes made the differentiation between barren land and agricultural land more difficult. Accordingly, the overall classification accuracy and K^ statistic decreased to 94.5%, and 0.89, respectively. Similar circumstances were found in 2008, resulting in an overall classification accuracy and K^ statistics of 95% and 0.88, respectively. Table 2.5 shows the overall accuracies for NDVI images were 100-, 99.5-, and 94% for 1984, 1999, and 2004, respectively. The overall K^ statistics were 1 in 1984, 0.96 in 1999, and 0.88 in 2004. Conversely, in 2008, the results failed to represent reality. The overall accuracy and K^ statistics were 77.5% and 0.57, respectively.

2.5.4 Land cover changes results

According to the previous results from both the hybrid classification procedure and NDVI analysis, monitoring the changes in land cover between each two dates was performed. Pairs of images from two different dates were used to produce land cover change images and the cross-tabulation matrix between the dates.

2.5.4.1 Land cover changes using hybrid classification approach outputs

Figure 2.5 shows the thematic land cover change images using the outputs of the hybrid classification technique at two different dates. Table 2.6 shows the cross-tabulation matrix for the areas changed from one land cover class to another by percentage.

Table 2.4. Error matrix for thematic raster classified images for different land cover on different dates in *Bustan 3* area, Egypt.

Classified Data	*Reference Data*				*Classified Total*	*Users Accuracy*	*K^*
1984	Barren						
Barren Land	200				200	100%	1
Reference Total	200				200		
	Overall Classification Accuracy =					100%	
	Overall K^ Statistics =					1	
1990	Agricultural	Barren					
Agricultural Land	5	0			5	100%	1
Barren Land	0	195			195	100%	1
Reference Total	5	195			200		
Producers Accuracy	100%	100%					
	Overall Classification Accuracy =					100%	
	Overall K^ Statistics =					1	
1999	Urban	Agricultural	Water	Barren			
Urban Land	5	0	0	0	5	100%	1
Agricultural Land	0	27	0	0	27	100%	1
Water	0	0	10	0	10	100%	1
Barren Land	0	5	2	151	158	95.57%	0.8
Reference Total	5	32	12	151	200		
Producers Accuracy	100%	84.38%	83.33%	100%			
	Overall Classification Accuracy =					96.50%	
	Overall K^ Statistics =					0.91	
2004	Urban	Agricultural	Water	Barren			
Urban Land	5	0	0	0	5	100%	1
Agricultural Land	0	123	2	8	133	92.48%	0.8
Water	0	0	10	0	10	100%	1
Barren Land	0	1	0	51	52	98.08%	0.8
Reference Total	5	124	12	59	200		
Producers Accuracy	100%	99.19%	83.33%	86.44%			
	Overall Classification Accuracy =					94.5%	
	Overall K^ Statistics =					0.89	
2008	Urban	Agricultural	Water	Barren			
Urban Land	5	0	0	0	5	100%	1
Agricultural Land	0	146	4	2	152	96.05%	0.9
Water	0	0	10	0	10	100%	1
Barren Land	0	3	1	29	33	87.88%	0.9
Reference Total	5	149	15	31	200		
Producers Accuracy	100%	97.99%	66.67%	93.55%			
	Overall Classification Accuracy =					95%	
	Overall Kappa Statistics =					0.88	

Table 2.5. Error matrix for continuous raster images for NDVI on different dates in *Bustan 3* area, Egypt.

Classified Data	*Reference Data*		*Classified Total*	*Users Accuracy*	*K^*
1984	Non-Vegetated				
Non-Vegetated	200		200	100%	1
Reference Total	200		200		
Producers Accuracy	100%				
	Overall Classification Accuracy =			100%	
	Overall Kappa Statistics =			1	
1990	Non-Vegetated				
Non-Vegetated	200		200	100%	1
Reference Total	200		200		
Producers Accuracy	100%				
	Overall Classification Accuracy =			100%	
	Overall Kappa Statistics =			1	
1999	Non-Vegetated	Vegetated			
Non-Vegetated	186	1	187	99.47%	0.92
Vegetated	0	13	13	100%	1
Reference Total	186	14	200		
Producers Accuracy	100%	92.86%			
	Overall Classification Accuracy =			99.5%	
	Overall Kappa Statistics =			0.96	
2004	Non-Vegetated	Vegetated			
Non-Vegetated	83	11	94	88.3%	0.80
Vegetated	1	105	106	99.06%	0.98
Reference Total	84	116	200		
Producers Accuracy	98.8%	90.5%			
	Overall Classification Accuracy =			94%	
	Overall Kappa Statistics =			0.88	
2008	Non-Vegetated	Vegetated			
Non-Vegetated	79	45	124	63.7%	0.40
Vegetated	0	76	76	100%	1
Reference Total	79	121	200		
Producers Accuracy	100%	62.8%			
	Overall Classification Accuracy =			77.5%	
	Overall Kappa Statistics =			0.57	

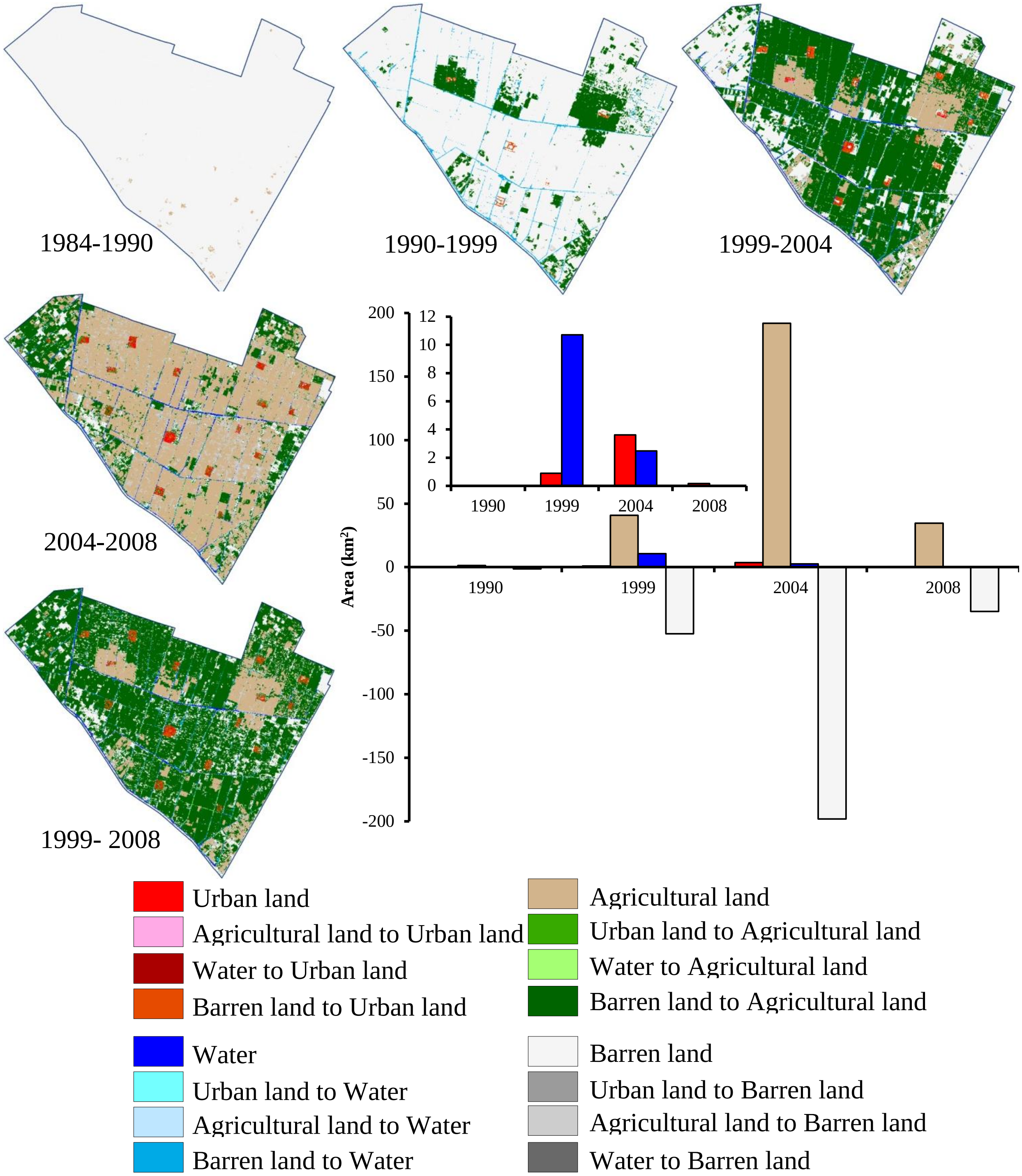

Figure 2.5. Thematic change images designate the land cover change detection between each two dates in *Bustan 3* area, Egypt. The chart clarifies the area for each land cover class by square kilometer which increased or decreased on different dates, the small one elucidates the areas that increased for both water and urban land on different dates.

Table 2.6. Cross-tabulation matrices for pairs of thematic raster classified images for different land cover classes on two different dates. The values symbolize the area by percentage.

	Land Cover classes	Urban or Built-up Land	Agricultural Land	Water	Barren Land
		1990			
1984	Urban or Built-up Land	---*	---	---	---
	Agricultural Land	---	---	---	---
	Water	---	---	---	---
	Barren Land	---	0.35	---	99.65
		1999			
1990	Urban or Built-up Land	---	---	---	---
	Agricultural Land	0.00	0.10	0.00	0.25
	Water	---	---	---	---
	Barren Land	0.25	12.28	3.14	83.97
		2004			
1999	Urban or Built-up Land	0.23	0.01	0.00	0.01
	Agricultural Land	0.01	11.29	0.02	1.05
	Water	0.06	0.95	1.62	0.51
	Barren Land	1.01	56.40	2.22	24.58
		2008			
2004	Urban or Built-up Land	0.90	0.39	0.01	0.02
	Agricultural Land	0.08	68.30	0.20	0.07
	Water	0.00	0.97	2.45	0.44
	Barren Land	0.38	9.13	1.20	15.45
		2008			
1999	Urban or Built-up Land	0.22	0.04	0.00	0.00
	Agricultural Land	0.02	11.87	0.08	0.40
	Water	0.01	1.37	1.29	0.46
	Barren Land	1.10	65.52	2.48	15.11

---* indicate that this class was not existing in that date

The results show that the land cover change rate was very small between 1984 and 1990. Barren land occupied almost the entire area (99.65%) with only very tiny spots of vegetation represented (0.35%). Between 1990 and 1999, the reclamation accelerated and the construction of new agrarian communities began. Consequently, new land cover classes were observed. About 0.25% and 3.14% of barren land was transformed to urban land and water bodies, respectively. Also, 12.28% of barren land was changed to agricultural land. This indicates that around 16% of the area changed from one land cover to another, while about 84% of *Bustan 3* area remained unchanged. Between 1999 and 2004, the whole infrastructure of *Bustan 3* area

was completed, therefore, impressive rates of change were observed. Around 56.4% of the barren land in 1999 was developed to agricultural land by 2004. Moreover, 2.22% was transformed to water bodies and 1% was changed to urban land. Due to the remarkable change which occurred during this period, areas of no-change represented only 37.7%, and the changed area represented 62.3%. From 2004 to 2008, changes in land cover also took place, but at a slower rate of change than 1999–2004. The percentage of barren land in 2004 which was changed to agricultural land, water, and urban land in 2008 was 9.1%, 1.2%, and 0.39%, respectively. The unchanged area represented 87.1% and 13% of the area was changed (Table 2.4). In order to monitor the change in land cover during a longer period (last nine years) after the reclamation process, another change matrix was calculated between 1999 and 2008. A dramatic change rate was realized, the most significant of which was 65.52% of barren land being improved to agricultural land. Additionally, 2.5% and 1% of barren lands were converted to water and urban land, respectively. The overall change area during those nine years was 71.5%, while 28.5% remained unchanged. Some illogical results were observed in Table 2.6 such as urban land being altered to water or barren land. These may be due to inaccurate analysis during the editing of signature classes. Ephemeral streams or irrigation/drainage canals are frequently dry, especially in the summer. This could lead to water in one year appearing as barren land in subsequent years.

2.5.4.2 Land cover changes using NDVI analysis outputs

Figure 2.6 and Table 2.7 show the continuous land cover change images using the outputs of the NDVI analysis at two different dates and the cross-tabulation matrix for the areas which changed from one land cover class to another by percentage, respectively. The results show that the change in vegetation status was minor between 1984 and 1990, as 99.9% of the *Bustan 3* area

was non-vegetated land during 1990. With development between 1990 and 1999, 6.43% of non-vegetated land was transformed to vegetated land; however, 93.5% of the area unchanged.

Between 1999 and 2004, 47.92% of non-vegetated lands were converted to vegetated lands, while 45.65% of the area remained non-vegetated as a result of the reclamation processes. From 2004 to 2008, the change rate was also remarkable, but also erroneous. Only 13.77% of non-vegetated land in 2004 was transformed to vegetated lands by 2008. Conversely, 28.65% of vegetated lands in 2004 were transformed to non-vegetated land by 2008. Thus, NDVI failed to accurately discriminate fallow and newly cultivated fields from barren lands during that period. Due to the erroneous patterns which were observed in 2008, monitoring of the changes in land cover during the last nine years was omitted.

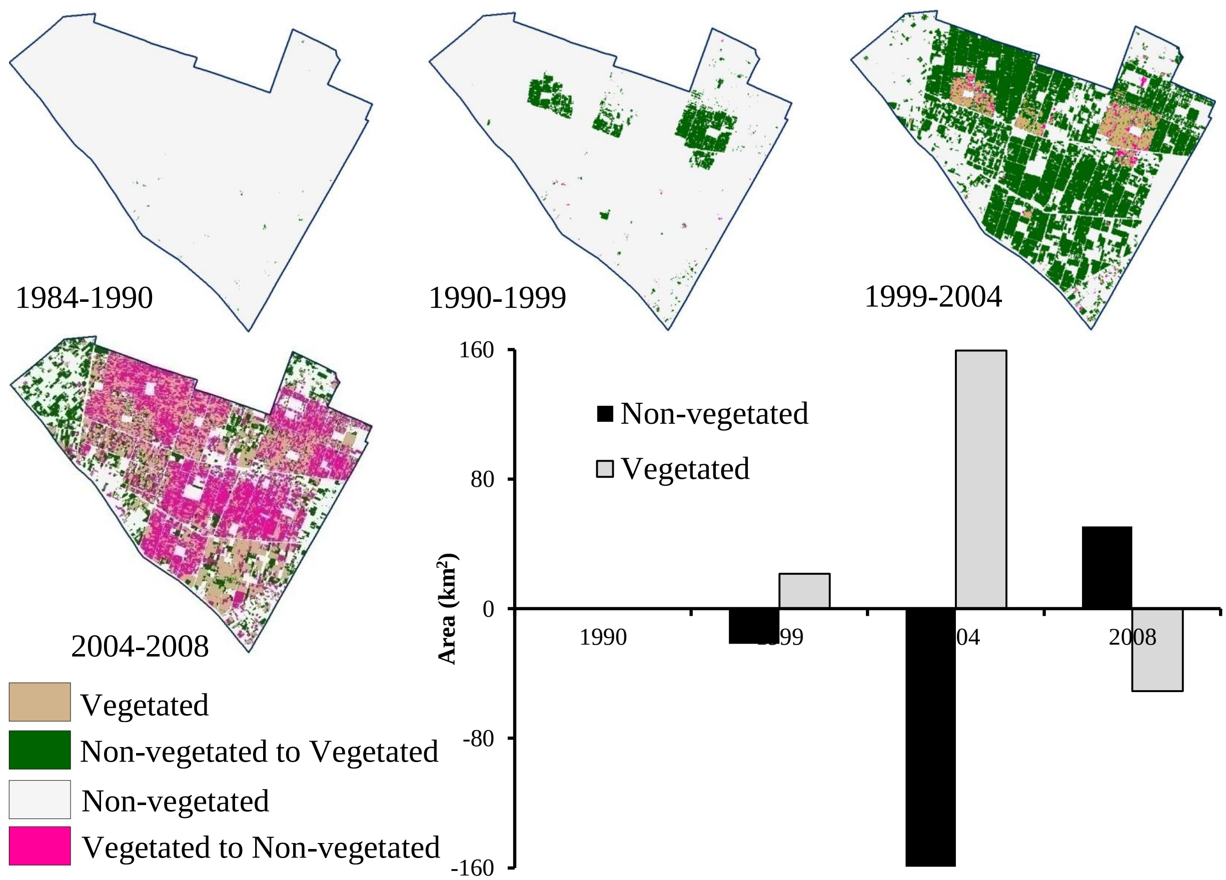

Figure 2.6. Thematic change images designate the NDVI change detection between each two dates in *Bustan 3* area, Egypt. The chart explains the area for each NDVI class by square kilometer which increased or decreased on different dates.

Table 2.7. Cross-tabulation matrices for pairs of continuous raster images for NDVI on two different dates. The values symbolize the area by percentage.

	NDVI classes	Non-vegetated	Vegetated
		1990	
1984	Non-vegetated	99.92	0.08
	Vegetated	---*	---
		1999	
1990	Non-vegetated	93.49	6.43
	Vegetated	0.07	0.01
		2004	
1999	Non-vegetated	45.65	47.92
	Vegetated	1.23	5.21
		2008	
2004	Non-vegetated	33.11	13.77
	Vegetated	28.65	24.48
		2008	
1999	Non-vegetated	59.24	34.32
	Vegetated	2.52	3.92

---* indicate that this class was not existing in that date

2.6 CONCLUSIONS

During the last 24 years (1984–2008), an impressive land cover change was observed in the *Bustan 3* area, one of the newly reclaimed areas in the western desert of Egypt. In this research, the hybrid classification approach and NDVI analysis were used to monitor the land cover changes during this period. According to the research results, the combination between supervised and unsupervised classification systems in the hybrid classification technique offered more reliable and accurate classified images that were used to monitor the change in land cover in this area. By contrast, the NDVI analysis failed to provide acceptable data during 2008. In areas like the *Bustan 3* area of Egypt, the mixed classes with medium resolution Landsat imagery require knowledge about the actual ground cover types to achieve satisfactory results.

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CHAPTER 3. MULTI-TEMPORAL ASSESSMENT OF LAND SENSITIVITY TO DESERTIFICATION IN A FRAGILE AGRO-ECOSYSTEM: ENVIRONMENTAL INDICATORS[2]

3.1 INTRODUCTION

The assessment of land degradation is a prerequisite procedure for achieving sustainable land use. Land degradation refers to reduction or loss of the biological or economic productivity and complexity of rain-fed cropland, irrigated cropland, or range, pasture, forest, and woodlands resulting from land uses or from a process or combination of processes arising from human activities and habitation patterns (United Nations, 1994). Based on this definition, land degradation is considered a serious problem in arid and semi-arid regions due to their fragility, and it is used to describe environmental phenomena affecting dry-lands (Salvati and Zitti, 2009; Salvati *et al.,* 2011). The fragility of dry-lands relates to many ecological features such as: limited water resources, rainfall variability, thin plant cover, and little development of surface deposits with low organic matter content (Kassas, 1995). Gao and Liu (2010) stated that severe degradation is blamed for the disappearance of approximately 5 to 10 million ha of agricultural land annually. Additionally, there has been a significant increase in soil degradation processes, and there is evidence that those processes will further increase if no action is taken (Montanarella, 2007). Desertification describes land degradation of dry-lands (Adamo and Crews-Meyer, 2006) and it affects approximately one-third of the Earth's surface area, mostly in developing countries (UNCCD, 2002). To differentiate between land degradation and desertification, UNCCD (1999) defined desertification as a process of land degradation in arid, semiarid, and dry sub-humid areas resulting from human activities and climate variation.

[2] Reprinted from Ecological Indicators, Vol. 15 (1), Bakr, N., D.C. Weindorf, M.H. Bahnassy, and M.M. El-Badawi, Multi-temporal assessment of land sensitivity to desertification in a fragile agro-ecosystem: Environmental indicators, 271-280, 2012, with permission from Elsevier.

In agronomic production, degradation can be a reversible and controlled process, while desertification is a permanent and practically irretrievable process with an almost total loss of biological potential (Basso *et al.*, 2000). In dry-lands, three agricultural land-use systems typically exist; irrigated cropland, rain-fed cropland, and rangeland grazing (Glenn *et al.*, 1998).

Deterioration in irrigated lands is often related to the rise of the water table (water-logging) which often entails salinization and other forms of chemical damage of the soil (Kassas, 1995). Combating desertification requires prevention and/or reduction of land degradation, rehabilitation of degraded land, and reclamation of desertified land (United Nations, 1994). Pagiola (1999) stated that agriculture expansion into new areas may mask the effects of land degradation. However, continued expansion will bring new marginal lands into use. Thus, to achieve sustainable agricultural production, consideration of land degradation and its associated risks is required and can be achieved using proper methods according to the locally dominant degradation-related processes (Contador *et al.*, 2009).

Field visits and remote sensing are suggested methods for studying land degradation. Compared to field evaluation, remote sensing data is cost-effective, time-efficient, and valuable in mapping land degradation risks (Gao and Liu, 2008; Li *et al.*, 2007). Multi-temporal remote sensing data is ideal for monitoring long-term trends of land degradation and assessing land degradation severity, which requires spatial comparison of multiple land cover maps at different times to determine spatial changes (Collado *et al.*, 2002; Geymen and Baz, 2008). If combined with GIS, remote sensing can be used to identify areas of land degradation and link them to physiographic settings (Van Lynden and Mantel, 2001). As environmental sensitivity is the response of the environment to a change in one or more external factors, degradation occurs when this response is deleterious to the environment. An Environmentally Sensitive Area (*ESA*) is a spatially delimited entity in which environmental and socio-economic factors are not

sustainable for that particular environment (Basso *et al.*, 2000). The effects of environmental changes on land degradation have been studied by many researchers (e.g. Salvati and Zitti, 2008; Santini *et al.*, 2010; Lautenbach *et al.*, 2011).

The Mediterranean Desertification and Land Use (*MEDALUS*) approach (Kosmas *et al.*, 1999) focuses on recognizing ESAs through multi-factor approaches. For defining the environmental sensitivity area index (*ESAI*); soil, climate, vegetation, and management qualities are considered. This approach is simple, robust, widely applicable, and adaptable to new information (Kosmas *et al.*, 1999; Kosmas *et al.*, 2003; and Brandt *et al.*, 2003). Three *ESAs* to desertification types can be distinguished by *MEDALUS* approach: a) *critical ESAs* for areas already highly degraded, b) *fragile ESAs* for areas in which any change in the delicate natural and human activity balance can lead to desertification, and c) *potential ESAs* for areas threatened under significant climate change or if a particular combination of land use practices are implemented. Areas with deep to very deep, nearly flat, well drained, coarse-textured or finer soils, and under semi-arid or wetter conditions are considered non-threatened by desertification. The *MEDALUS* approach has been broadly used in Europe as a successful tool for detecting the most vulnerable areas to degradation (e.g. Basso *et al.*, 2000; Salvati and Zitti, 2009; Contador *et al.*, 2009; Santini *et al.*, 2010). Although Egypt has different ecosystems, several studies have used the standard *MEDALUS* approach either on the whole Egyptian territory (Gad and Lotfy, 2006) or in specific parts of Egyptian lands (Ali and El Baroudy, 2008; Gad and Shalaby, 2010).

Egyptian agriculture lands can be divided into *Oldlands* and *Newlands*. *Oldlands* are found in the Nile Valley as well as the Nile Delta and include the lands that have been intensively cultivated for long periods of time. *Newlands* include lands that have been reclaimed relatively recently (post-1950) or are in the process of being reclaimed (UNDP, 2003). Land reclamation in the Egyptian context means converting desert areas into agricultural land by

extending water canals into the desert, enhancing soil fertility, and providing infrastructure for new village construction (Adriansen, 2009). In the present study, the *Bustan 3* area was chosen for assessing the *ESAs* to desertification. This area represents one of the newly reclaimed areas in the western Nile Delta, Egypt; having been changed from 100% bare soil in 1984 to ~79% agricultural land in 2008 (Bakr *et al.*, 2010). Consequently, there is a strong need for studying the impact of this change on the desertification process in such a fragile agro-ecosystem. As few studies have used the adjusted *MEDALUS* approach under local conditions (Sepehr *et al.,* 2007; Rasmy, *et al.*, 2010) similar to *Bustan 3*, the objectives of this research include: a) identifying the most environmental sensitive areas to desertification in the *Bustan 3* area, b) assessing the *ESAIs* of 1984 and 2008 to determine the effects of land reclamation processes, c) adjusting the *MEDALUS* factors for 2008 to obtain more reliable data at the local level, and d) monitoring the *ESAI* change between 1984 and 2008 over the studied area.

3.2 METHODS

3.2.1 Study site

The study was conducted in the *Bustan 3* area of the west Nile Delta reclamation zone, Egypt (Figure 3.1). The geographical location is in UTM zone 36 between longitude: 30° 8' to 30° 27' E and latitude: 30° 26' to 30° 39' N. The *Bustan 3* area occupies 341.27 km^2 (34,128 hectares). In the 1990s, the *Bustan 3* area was targeted for reclamation processes (IFAD, 1992), and as a result, between 1984 and 2008 *Bustan 3* landscapes have been transformed dramatically from 100% barren land (desert), to 79% agricultural land (Bakr *et al.*, 2010).

Climatic parameters were collected from the *Tahrir* meteorological station (latitude: 30° 39' N, longitude: 30° 42' E, elevation: 16 m) and the *Wadi El-Natroon* meteorological station (latitude: 30° 40' N, longitude: 30° 35' E, elevation: 1 m). Climate of the *Bustan 3* area is characterized by an arid to semi-arid Mediterranean climate with very low precipitation. The 35

mm of annual rainfall occurs during the winter months (October to March). Temperatures are high during summer months and relatively low in the winter. The hottest and the coldest months are August (35°C) and January (8°C), respectively. Relative humidity averages 73.5%. The average wind speed and sunshine are 264.8 km d^{-1} and 9.4 h, respectively (FAO, 1993). Soils of the *Bustan 3* area are classified as Typic Torripsamments (Soil Survey Staff, 2010).

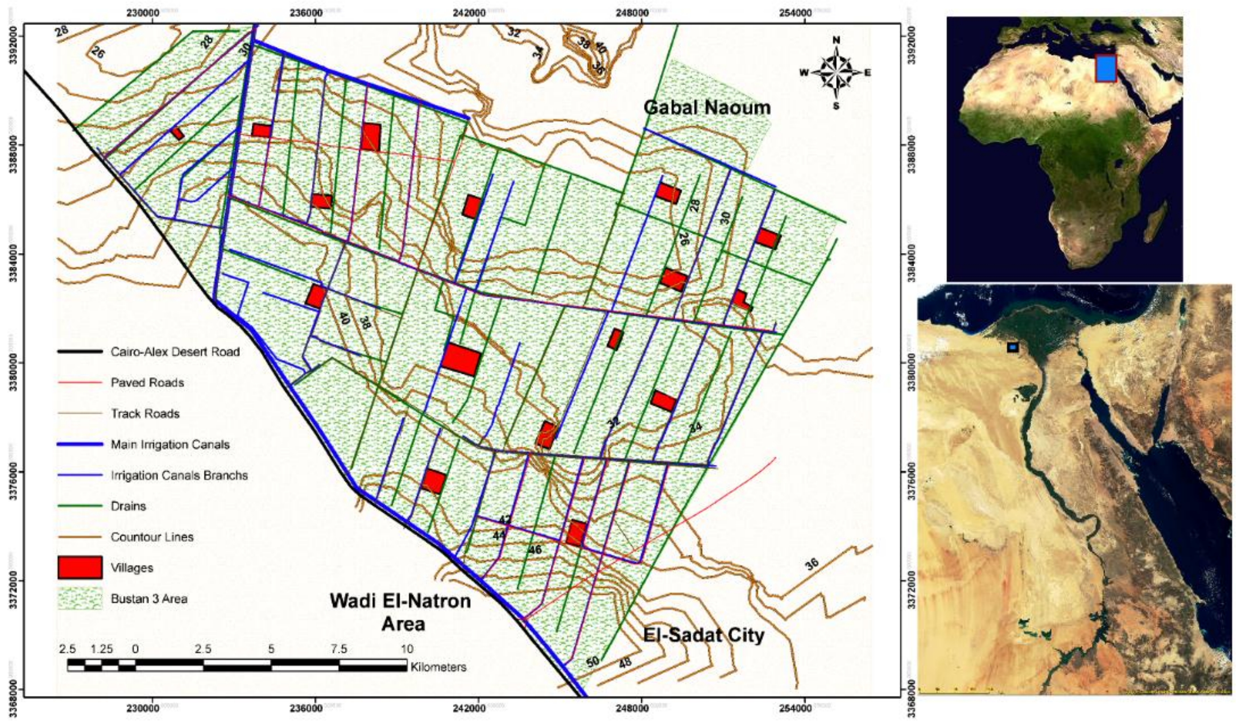

Figure 3.1. Infrastructure of the *Bustan 3* area, Egypt. (Source: Bakr *et al*., 2010)

The main landforms of the *Bustan 3* area are desert with relatively low altitude and sandy plains. The relatively low altitude landform is characterized by an undulating land form of coarse sand. Sandy plain landforms are sandy, nearly level sediments of the deltaic stage of river terraces (Sadek, 1993). The elevation ranges from 24 to 51 m based upon the digital elevation model (DEM), and the slope gradient varies from 0 to 3.5% according to slope analysis (Bakr *et al.,* 2009). Bakr *et al.* (2009) found that 70% of the *Bustan 3* area has good capability for agriculture and the percentage could be increased to 96% when best management practices are applied.

3.2.2 Data-set

Data analyses were based mainly upon land cover maps of the *Bustan 3* area from 1984 and 2008 (Figure 3.2). Those two maps were produced via satellite imagery (Figure 3.3); Landsat-5 (Multispectral Scanner, MSS), acquired on July 9, 1984 and Landsat-7 (Enhanced thematic mapper, ETM^+), acquired on July 3, 2008. Barren land was the only recognizable land cover type in the 1984 land cover map. However, the land cover classes represented in the 2008 map were; barren land, agricultural land, urban area, and water, as a result of agricultural activities (Bakr *et al.*, 2010). For producing the infrastructure map of the study area, three topographic maps sheets at a scale of 1:50,000 were digitized from paper maps to digital form (Figure 3.1), and used to rectify the satellite images (Bakr *et al.*, 2010).

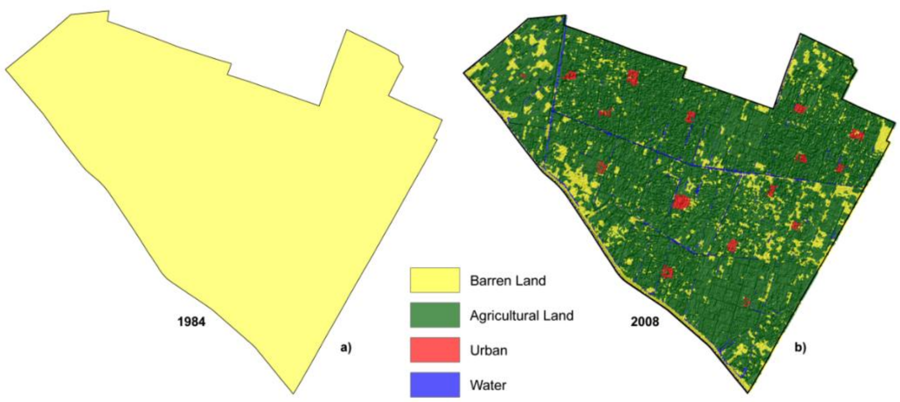

Figure 3.2. Land cover in the *Bustan 3* area, Egypt; a) land cover classes of 1984, b) land cover classes of 2008. (Source: Bakr *et al.*, 2010)

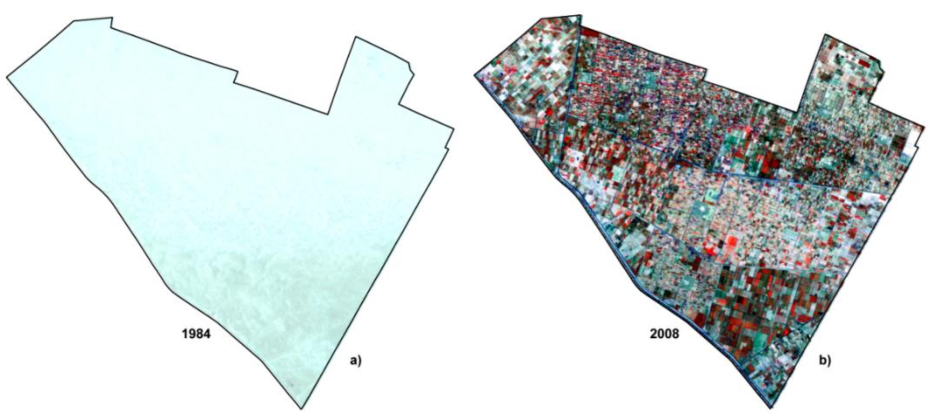

Figure 3.3 Landsat satellite images of the *Bustan 3* area, Egypt; (a) MSS 1984, bands 4-3-2 were assigned as RGB, (b) ETM^+ 2008, bands 4-5-3 were assigned as RGB. (Source: Bakr *et al., 2010*)

Quality indicators were assessed via 48 soil samples collected in July 2007. Each sample was geo-referenced using the global positioning system. The soil samples were air-dried, ground, and passed through a 2-mm sieve. Large fragments (>2 mm) ware expressed as stoniness percentage, while <2 mm soil was stored for further laboratory analysis. The soil pH and soil salinity (dS m^{-1}) were determined using a 1:1 soil/water extract, soil texture determined via dry sieving, and soil organic matter (SOM) content was carried out using the Walkley-Black method (Soil Survey Staff, 2004). In addition, seventeen water samples were collected from different irrigation sources (irrigation canals and groundwater). The water samples were appropriately handled and preserved. The pH and water salinity (dS m^{-1}) were measured promptly and then acidified to pH 2 for cation and anion analyses (Clesceri *et al.*, 1998). The analysis results were used to document water quality for agricultural purposes.

3.2.3. Environmental indicators

In the standard *MEDALUS* approach (Kosmas *et al.*, 1999), the score was allocated to each parameter in each quality index. The scores ranged from "1= best" (for the least sensitive areas to desertification) to "2= worst" (for the most sensitive areas to desertification). Table 3.1 provides a summary of the standard *MEDALUS* approach indicators, integrated parameters, scoring, and the data source for each parameter. The soil, climate, and vegetation quality indicators are related to the physical environment. However, the management quality indicator closely relates to human-induced stress on the environment (Kosmas *et al.*, 1999). Consequently, the standard *MEDALUS* approach was applied in 1984, before reclamation processes were initiated in the *Bustan 3* area, and 2008, when the study area was dramatically transformed into agricultural land (Bakr *et al.*, 2010).

Table 3.1. Quality indicators parameters, description, score, and data sources used for application of the standard *MEDELUS* approach in the *Bustan 3* area, Egypt.

Parameter	Class	Description	Score	Data Sources
		SQI		
Texture	1	L, SCL, SL, LS, CL	1	Lab analysis
	2	SC, SiL SiCL	1.2	
	3	Si, C, SiC	1.6	
	4	S	2	
Parent Materials	1	Shale, schist, basic, ultra basic, Conglomerates, unconsolidated	1	Soil map, field observations
	2	Limestone, marble, granite, Rhyolite, Ignibrite, gneiss, siltstone, sandstone	1.7	
	3	Marl, Pyroclastics	2	
Rock Fragment (%)	1	> 60	1	Lab analysis
	2	20 - 60	1.3	
	3	<20	2	
Slope gradient (%)	1	<6	1	Slope map
	2	6 – 18	1.2	
	3	18 – 35	1.5	
	4	>35	2	
Soil Depth (cm)	1	>75	1	Soil map, field observations
	2	30 – 75	1.2	
	3	15 – 30	1.5	
	4	<15	2	
Drainage status	1	Well drained	1	
	2	Imperfectly drained	1.2	
	3	Poorly drained	2	
		CQI		
Rainfall (mm)	1	>650	1	FAOClim-NET
	2	280 – 650	1.5	
	3	<280	2	
Aridity (P/PET)	1	Humid: >0.65	1	UNEP (1992)
	2	Dry Sub-Humid: 0.50–0.65	1.2	
	3	Semi-Arid: 0.20–0.5	1.5	
	4	Arid: 0.05–2.0	1.7	
	5	Hyper-Arid <0.05	2	
Aspect	1	NW – NE	1	Aspect map
	2	SW – SE	2	

Table 3.1. (Cont.)

Parameter	Class	Description	Score	Data Sources
		VQI		
Fire Risk	1	Bare land, perennial agricultural crops, annual agricultural crops	1	Land cover map
	2	Annual agricultural crops , mixed Mediterranean macchia/evergreen forest	1.3	
	3	Mediterranean macchia	1.6	
	4	Pine forest	2	
Erosion protection	1	Mixed Mediterranean macchia/evergreen forests	1	
	2	Mediterranean macchia, permanent grasslands, evergreen perennial crops	1.3	
	3	Deciduous forests	1.6	
	4	Deciduous perennial agricultural crops	1.8	
	5	Annual agricultural crops, annual grasslands, vines, bare land	2	
Drought resistance	1	Mixed Mediterranean macchia/evergreen forests, Mediterranean macchia	1	
	2	Conifers, deciduous, olives	1.2	
	3	Perennial agricultural trees	1.4	
	4	Perennial grasslands	1.7	
	5	Annual agricultural crops, annual grasslands, bare land	2	
Plant cover	1	>40	1	
	2	10-40	1.8	
	3	<10	2	
		MQI		
Land use intensity (cropland)	1	Low land use intensity	1	Land cover map, field observations
	2	Medium land use intensity	1.5	
	3	High land use intensity	2	
Policy	1	High degree of implementation of environmental protection policies	1	Land cover map, field observations
	2	Moderate degree of implementation of environmental protection policies	1.5	
	3	Low degree of implementation of environmental protection policies	2	

The *MEDALUS* approach is highly flexible and allows updates according to local conditions and the availability of information (Contador *et al.*, 2009). Accordingly, new parameters were introduced to the standard *MEDALUS* approach, as a result of the reclamation processes in the *Bustan 3* area, including SOM, electrical conductivity (EC), and soil pH for soil

quality. Also, the irrigation water quality indicator was inserted with associated parameters of EC_w, chloride (Cl), and sodium adsorption ratio (SAR). These new parameters were suggested by Sepehr *et al.* (2007) and the data range was selected according to the irrigation water quality guidelines (Ayers and Westcot, 1985). Table 3.2 shows the new parameters range, scoring, and the data source.

Table 3.2. Parameters added to the standard *MEDELUS* approach for the *ESAI* adjustment in the *Bustan 3* area, Egypt.

Parameter	Class	Description	Score	Data Sources
		SQI		
OM (%)	1	>3	1	Lab analysis
	2	2-3	1.2	
	3	1-2	1.5	
	4	0.5-1	1.7	
	5	<0.5	2	
EC (dS/m)	1	<1.2	1	
	2	1.2-2.5	1.2	
	3	2.5-4.5	1.5	
	4	4.5-9	1.7	
	5	>9	2	
pH (1:2.5)	1	<5.5	2	
	2	5.5-6.5	1	
	3	6.5-7.5	1.5	
	4	7.5-8.4	1.7	
	5	>8.4	2	
		IWQI		
EC (dS/m)	1	<0.7	1	
	2	0.7-3	1.5	
	3	>3	2	
Cl (meq/l)	1	<4	1	
	2	4-10	1.5	
	3	>10	2	
SAR	1	0-3	1	
	2	3-6	1.2	
	3	6-12	1.5	
	4	12-20	1.7	
	5	20-60	2	

3.2.4 Environmental Sensitivity Area Index

The general formula that has been utilized for each aforementioned indicator is:

$$\text{Index}_i = (\text{parameter}_1\text{* parameter}_2\text{* parameter}_3\text{*}\ldots\ldots\ldots\ldots)^{1/n} \quad (3.1)$$

where, *i* represents the different quality indices, and *n* represents the number of parameters.

The thematic indicators for the standard *MEDALUS* approach (Table 3.1) include; soil quality indicator (SQI), climate quality indicator (CQI), vegetation quality indicator (VQI), and management quality indicator (MQI). The indicators were calculated as the geometric mean of the different weights of each individual parameter as:

$$\text{SQI = (soil texture*parent material*rock fragment*slope*soil depth*drainage)}^{1/6} \quad (3.2)$$

$$\text{CQI = (rainfall*aridity*aspect)}^{1/3} \quad (3.3)$$

$$\text{VQI = (fire risk*erosion protection*drought resistance*plant cover)}^{1/4} \quad (3.4)$$

$$\text{MQI = (land use intensity*policy enforcement)}^{1/3} \quad (3.5)$$

In the adjusted *MEDALUS* approach (Table 3.2), the same equations for CQI, VQI, and MQI were applied. However, the SQI equation was adjusted and a new equation for irrigation water quality indicator (IWQI) was used as follows:

$$\text{SQI = (Texture*parent material*rock fragment*slope* depth*drainage*SOM*EC*pH)}^{1/9} \quad (3.6)$$

$$\text{IWQI} = (\text{EC}_w\text{*SAR*Cl})^{1/3} \quad (3.7)$$

Then, the ESAI for the standard *MEDALUS* approach was calculated by:

$$\text{ESAI = (SQI*CQI*VQI*MQI)}^{1/4} \quad (3.8)$$

While the adjusted *MEDALUS* approach equation was:

$$\text{ESAI = (SQI*CQI*VQI*MQI*IWQI)}^{1/5} \quad (3.9)$$

Based on the calculations, four types of *ESAs* were assigned in the *MEDALUS* approach; a) critical areas (*ESAI* > 1.38), b) fragile areas (1.38 > *ESAI* > 1.23), c) potential areas (1.23 >

ESAI > 1.17), and d) non-affected areas (*ESAI* < 1.17). The score range for each quality indicator and final *ESAI* score are given in Table 3.3.

Table 3.3. Final indicators and *ESAI* classes, description and ranges for the *Bustan 3* area, Egypt.

Indices	Quality Classes	Description	Range
SQI	1	High quality	<1.13
	2	Moderate quality	1.13 – 1.45
	3	Low quality	>1.45
CQI	1	High quality	<1.15
	2	Moderate quality	1.15 – 1.81
	3	Low quality	>1.81
VQI	1	High quality	1-1.13
	2	Moderate quality	1.13-1.41
	3	Low quality	>1.41
MQI	1	High quality	1– 1.25
	2	Moderate quality	1.25-1.5
	3	Low quality	>1.5
IWQI	1	High quality	<1
	2	Moderate quality	1-1.41
	3	Low quality	>1.41
ESAI	Critical	C3	>1.53
		C2	1.41-1.53
		C1	1.37-1.41
	Fragile	F3	1.32-1.37
		F2	1.26-1.32
		F1	1.22-1.26
	Potential	P	1.17-1.22
	Non affected	N	<1.17

3.2.5 Map generation

The land cover maps for 1984 and 2008 were previously produced by Bakr *et al.* (2010) using a hybrid classification technique with a high accuracy via ERDAS IMAGINE 9.3 (Leica Geosystems, 2008) software (Figure 3.2). Depending on the different parameters that produced each indicator, spline interpolation in ArcGIS 9.3 (ESRI, 2008) was used, as it offers good results with gently varying surfaces (Robinson and Metternicht, 2006), then the area for each quality indicator was calculated. To achieve the final ESAI maps, the ERDAS IMAGINE Spatial Modeler (Leica Geosystems, 2010) was used to build a model by overlaying different quality maps and using the corresponding algorithms. For the two specified time series, 1984 and 2008,

the standard *MEDALUS* approach was applied. There was no need to apply the adjusted *MEDALUS* approach in 1984 since all land cover was barren desert at that time (Figure 3.2). However, the adjusted *MEDALUS* approach was used for 2008 as the reclamation processes were highly established and cropland was flourishing.

3.3 RESULTS AND DISCUSSION

3.3.1 Standard *MEDALUS* approach

Table 3.4 shows the quality indicators' area coverage of the study area according to the standard *MEDALUS* approach for 1984 and 2008. In 1984, the SQI results indicate that 83% of the study area had a moderate soil quality. As the soil quality related parameters are inherent soil characteristics that are seldom affected with the reclamation processes, the SQI values are the same for 1984 and 2008 (Table 3.4). Figure 3.4a shows that the low soil quality class for 1984 as well as 2008 is distributed along the south-western and western borders of the *Bustan 3* area, while the rest of the study area has a moderate soil quality. The same interpretation for SQI results can be extended to analyze the CQI results. According to Table 3.4, in 1984 around 95% of the study area had a moderate CQI. In 2008, the annual precipitation was 40 mm and the maximum and minimum temperatures were 37°C and 7°C in June and January, respectively (FAOClim-NET, 2010). There was no observed difference between the climatic parameters of 2008 and 1984. Thus, the CQI distribution across the study area was the same in the two time series (Figure 3.4c). Since the annual precipitation and aridity index are constant across the area, the only (slight) change in the CQI value was due to aspect (Kosmas et al., 1999).

Since the *Bustan 3* area was completely bare soil in 1984 (Bakr *et al.*, 2010), it was expected that 100% of the area had low vegetation quality (Table 3.4), as the final VQI results are highly affected by the plant cover parameter.

Table 3.4 Quality indicators areal coverage from the standard *MEDALUS* approach parameters for the *Bustan 3* area, Egypt in 1984 and 2008.

Class	Quality	SQI		CQI		VQI		MQI	
		km^2	%	km^2	%	km^2	%	km^2	%
1984									
1	High	1.36	0.40	0.53	0.16	------	------	------	------
2	Moderate	281.83	82.58	324.94	95.22	------	------	------	------
3	Low	58.08	17.02	15.79	4.63	341.27	100.00	341.27	100.00
2008									
1	High	1.36	0.40	0.53	0.16	65.87	19.30	0.64	0.19
2	Moderate	281.83	82.58	324.94	95.22	164.39	48.17	21.09	6.18
3	Low	58.08	17.02	15.79	4.63	111.00	32.53	319.53	93.63

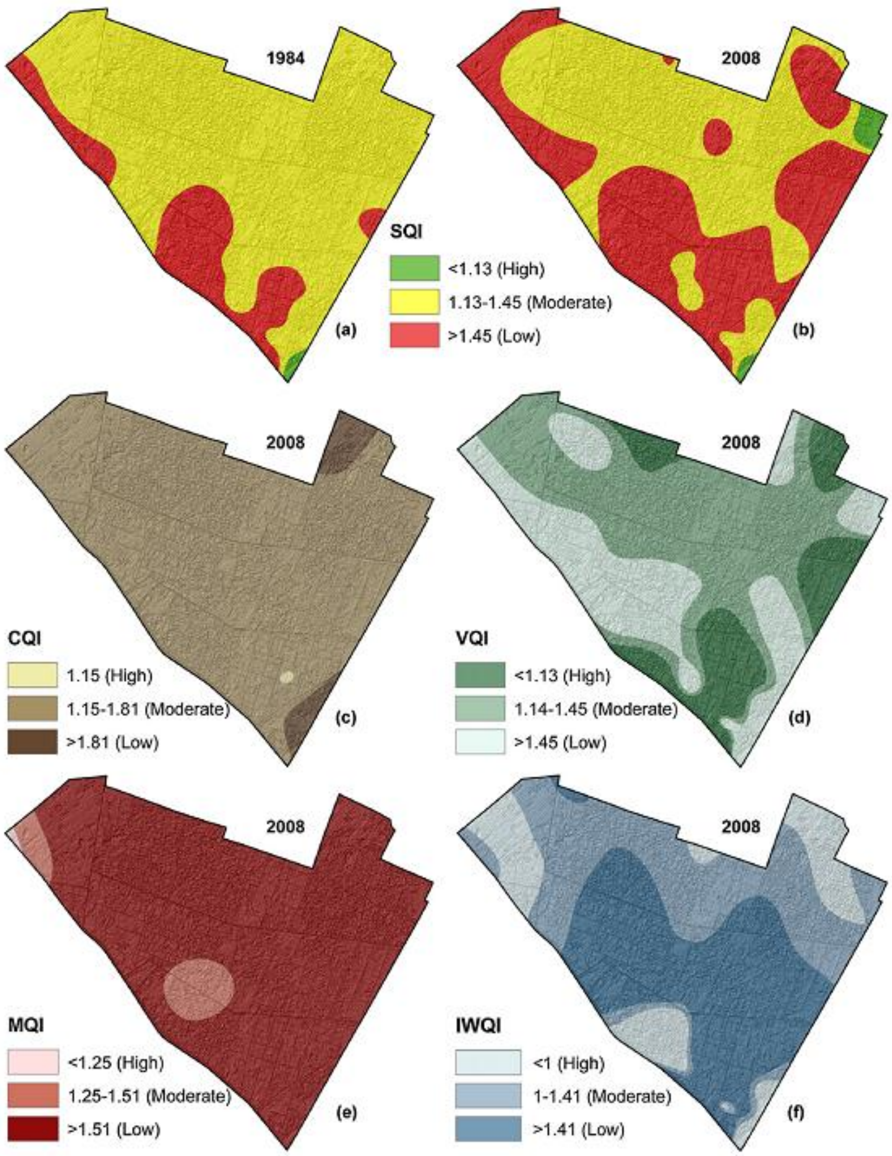

Figure 3.4. Soil quality indicators for the *Bustan 3* area, Egypt; a) SQI using standard *MEDALUS* approach in 1984 and 2008, b) SQI using adjusted *MEDALUS* approach in 2008, c) CQI 1984 and 2008, d) VQI in 2008, e) MQI in 2008, f) IWQI in 2008.

In 2008, the VQI results were totally changed due to the reclamation processes and agricultural activities. Consequently, different VQI classes were easily recognized, with VQI results in 2008 indicating that 20%, 48%, and 33% of the area were assigned as high, moderate, and low quality, respectively (Table 3.4). Figure 3.4d shows that the lowest VQI occurred in the southeastern and southwestern parts of the *Bustan 3* area. Similar to the VQI case, the results of MQI in 1984 indicate that 100% of the area was categorized as the lowest management quality (Table 3.4) since the *Bustan 3* area was a virgin desert where no management strategies were applied. By 2008, intensive agriculture existed with low fertility of natural resources, and actions such as highly mechanized cultivation and extensive use of fertilizers and pesticides that enhance the productivity were considered. Those subjects are related to the land use intensity parameter in the MQI (Kosmas *et al.*, 1999). The results of MQI for 2008 indicate that 94% of the *Bustan 3* area still suffered from poor or low management quality related to fragile ecosystems (Table 3.4). Fig. 4e shows the majority of low management quality across the whole area. The results of VQI and MQI reflect their critical role in influencing the ESAI. This finding is consistent with previous work of Rasmy *et al.* (2010).

3.3.2 Adjusted *MEDALUS* approach

Table 3.5 presents the quality indicators' area coverage of the *Bustan 3* area according to the adjusted *MEDALUS* approach for 2008. The SQI results show that only 54% of the area has a moderate soil quality compared with 83% when the standard *MEDALUS* approach was used (Tables 3.4 and 3.5). This decrease by 30% in the moderate soil quality class occurred after adding the SOM, EC, and pH parameters to the SQI. These results may be related to a high pH over all of the area, relatively high EC in some parts of the study area, and low organic matter content that can greatly affect physiochemical and biological indicators of soil quality (Reeves, 1997). Figure 3.4b shows that low soil quality areas occupy mainly the western, south-western,

and -eastern parts of the study area. As no adjustment was made to the CQI, VQI, and MQI, the distributions of those quality indicators were the same as the results from the standard *MEDALUS* approach for 2008 across the study area (Figure 3.4c, d, and e).

Table 3.5. Quality indicators areal coverage from the adjusted *MEDALUS* approach parameters for the *Bustan 3* area, Egypt in 2008.

Class	Quality	SQI		CQI		VQI		MQI		IWQI	
		km^2	%	km^2	%	km^2	%	km^2	%	km^2	%
1	High	5.4	1.6	0.5	0.2	65.9	19.3	0.6	0.2	60.4	17.7
2	Moderate	183	53.6	324.9	95.2	164.4	48.2	21.1	6.2	127.3	37.3
3	Low	152.8	44.8	15.8	4.6	111	32.5	319.5	93.6	153.5	45

The IWQI was included in the adjusted *MEDALUS* approach in order to assess the quality of water that was used for irrigation in the *Bustan 3* area. Table 3.5 shows that 45% of the study area was irrigated with low quality water and only 18% of the area used high quality water for irrigation. Due to the scarcity of rainfall and irregular presence of water in irrigation canals, farmers depend upon groundwater for irrigation. In most cases, groundwater has higher salinity and lower water quality than water in irrigation canals due to the low groundwater discharge and the high load of nutrients and fertilizers (Petheram *et al.*, 2008). Figure 3.4f shows that the southeastern and southwestern parts of the area use the lowest irrigation water quality in the *Bustan 3* area. Those aforementioned results demonstrate how the proposed methodology, using the adjusted *MEDALUS* approach, can greatly affect the final results of the quality indicators, for both the SQI and IWQI, and subsequently the estimation of *ESAI* to provide more reliable results depending upon the local conditions.

3.3.3 Environmental sensitivity area index (*ESAI*)

The *ESAIs* were calculated based on the overlaying technique of the different quality indicators which allows for the identification of links between those indicators and their spatial patterns. In agreement with Contador *et al.* (2009), the distribution of *ESAI* over the study area is closely related with the relationship between the parameters that were used to build the index.

Table 3.6 shows the areal coverage of the *ESAI* for the *Bustan 3* area, by square kilometer and percentage, using the standard and adjusted *MEDALUS* approaches for 1984 and 2008 over the *Bustan 3* area.

Table 3.6. Areal coverage for *ESAI* in the *Bustan 3* area, Egypt using the standard *MEDALUS* approach in 1984 and 2008 and the adjusted approach in 2008.

Class	Sub-class	ESAI 84 St.[1]		ESAI 08 St.[2]		ESAI 08 adj.[3]	
		km^2	%	km^2	%	km^2	%
Critical	C3	73.75	21.61	62.96	18.45	123.96	36.32
	C2	267.26	78.32	148.59	43.54	135.96	39.84
	C1	0.21	0.06	54.79	16.05	44.22	12.96
Fragile	F3	0.05	0.01	23.72	6.95	24.23	7.10
	F2	------	------	16.73	4.90	3.87	1.13
	F1	------	------	6.86	2.01	1.38	0.40
Potential	P	------	------	5.74	1.68	1.66	0.49
Non affected	N	------	------	21.88	6.41	5.99	1.75
Total		341.27	100.00	341.27	100.00	341.27	100.00

[1] ESAI that resulted from the application of standard MEDALUS approach in 1984
[2] ESAI that resulted from the application of standard MEDALUS approach in 2008
[3] ESAI that resulted from the application of adjusted MEDALUS approach in 2008

In 1984, the results reveal that almost 100% of the study area fell into the *critical* class (Table 3.6). These results were anticipated since the study area was barren land without any plant coverage at this time. Figure 3.5c shows that the whole area was classed as *critical* (C) with no appearance of the other classes. For 2008, two different *ESAIs* were estimated; the *ESAI* based on the standard and adjusted *MEDALUS* approaches. When the standard *MEDALUS* approach was applied, the results show that 78% and 14% of the study area were classed as *critical* and *fragile ESAs*, respectively (Table 3.6). Figure 3.5a displays the spatial distribution of *ESAI* over the *Bustan 3* area when the standard *MEDALUS* approach was applied. By contrast, using the adjusted *MEDALUS* approach, the *critical* class area coverage increased to occupy 89% and the *fragile* sensitivity class decreased to 9%. Figure 3.5b displays the spatial distribution of *ESAI* over the *Bustan 3* area when the adjusted *MEDALUS* approach was applied. The distribution of

ESAIs from Figure 3.5a or 3.5b perfectly matches with the quality indicator results (Figure 3.4) since the highest sensitivity areas to desertification are the lowest quality areas.

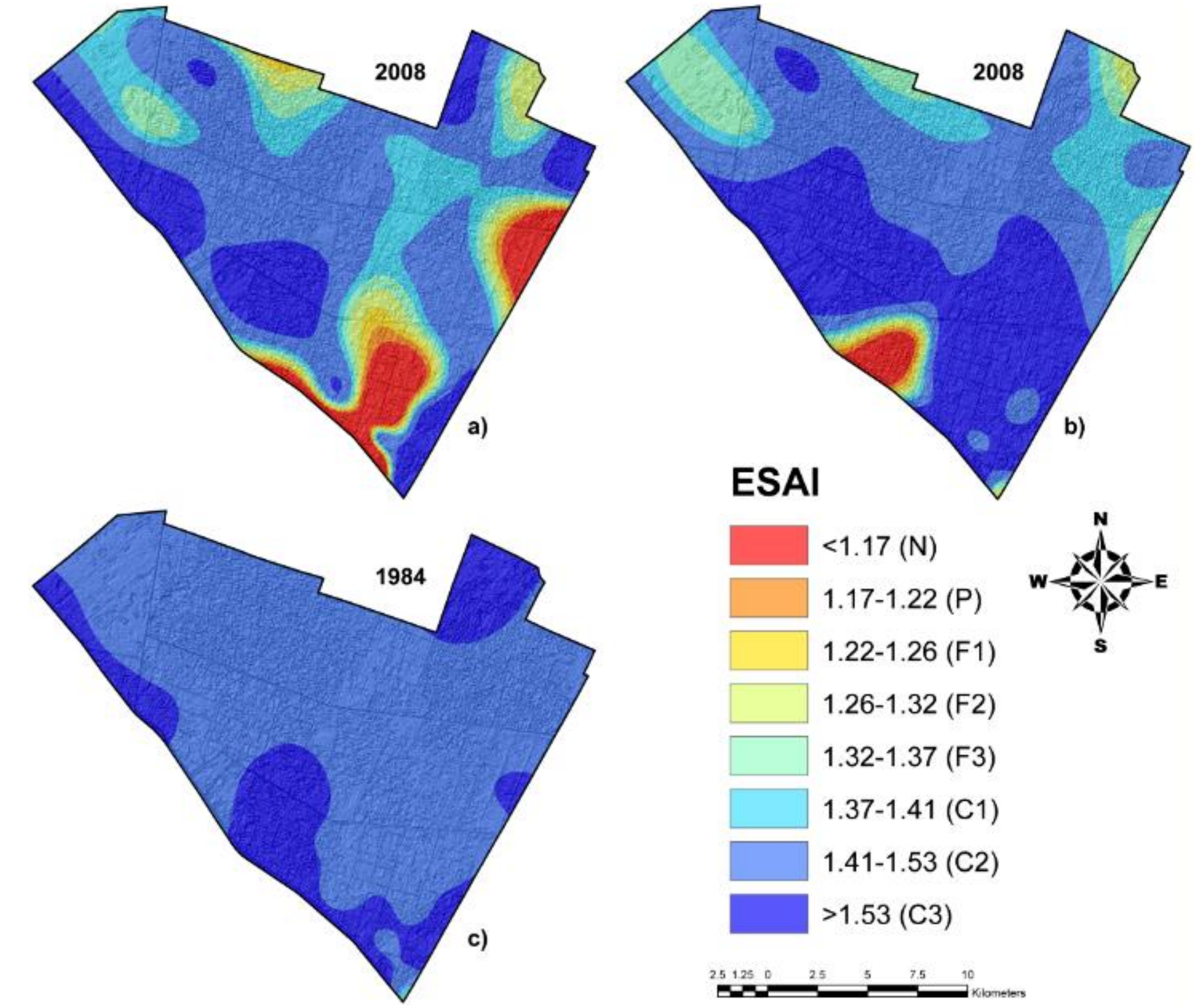

Figure 3.5. ESAI for the Bustan 3 area, Egypt; a) ESAI using the standard MEDALUS approach in 2008, b) ESAI using The adjusted MEDALUS approach in 2008, c) ESAI using the standard MEDALUS approach in 1984.

In general, the ESAI maps the *Bustan 3* area indicate that vegetation cover was the most important indicator affecting the final results when comparing the standard *MEDALUS* approach application for 1984 and 2008. This conclusion is consistent with the results of Sepehr *et al.* (2007). When comparing the standard and adjusted *MEDALUS* approaches in 2008, the results indicate that the IWQI and the parameters that were added to the SQI play an important role in increasing the sensitivity to desertification, especially in the western, southeastern, and

southwestern parts of the *Bustan 3* area. Thus, using modern irrigation systems, improving water management practices, and enhancing marginal land management will greatly combat the desertification process as reported by Abahussain *et al.* (2002).

3.4 CONCLUSIONS AND RECOMMENDATIONS

Identifying the appropriate parameters as well as choosing the suitable spatial and temporal scale, are essential for correctly identifying the ecosystem and assessing its sensitivity to desertification. In this case study, the standard *MEDALUS* approach was used in two time periods (1984 and 2008) to evaluate the impacts of the reclamation processes that take place in the *Bustan 3* area, Egypt. The adjusted *MEDALUS* approach for 2008 was applied by adding new parameters to the SQI and extending the quality indicators to include an IWQI. The results clearly elucidate the role that humans play in accelerating, slowing, or eliminating desertification processes. In a fragile, vulnerable agro-ecosystem such as the *Bustan 3* area, high sensitivity to desertification exists. Thus, decision-makers should give more attention to the most sensitive areas to desertification. Results of this study show that plant cover, management, and irrigation water quality dramatically impact desertification processes. Access to suitable irrigation water may remain problematic. However, the management of such areas can be improved much more easily. Finally, the monitoring of desertification processes over long periods of time provides valuable information and is highly recommended for proper land use planning as well as sustainable development.

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CHAPTER 4. LAND CAPABILITY EVALUATION IN NEWLY RECLAIMED AREAS: A CASE STUDY IN BUSTAN 3 AREA, EGYPT[3]

4.1 INTRODUCTION

Agriculture is a key sector of the Egyptian economy (IFAD, 2005). Egyptian agriculture can be divided geographically into *Upper* and *Lower* Egypt, where *Upper* Egypt comprises the Nile Valley from Giza to the south and *Lower* Egypt comprises the Nile Delta from Cairo to the north. These lands can be further divided into *Oldlands* and *Newlands*. *Oldlands* are found in the Nile Valley as well as the Nile Delta and include the lands that have been intensively cultivated for long periods of time. *Newlands* include lands that have been reclaimed relatively recently (post-1950) or are in the process of being reclaimed now. *Newlands* are less fertile, but with time and good management of water and cropping patterns, their productivity can improve (UNDP, 2003). Land evaluation and identifying limitations of soil utilization are essential elements for sustainable land use planning (Robert *et al.*, 1993). Land evaluation concerns the assessment of land performance when used for specified purposes and creates an interface between soil survey and land use planning (FAO, 1976). The function of land use planning is to guide land use decisions in such a way that the resources of the environment are put to the most beneficial use for humanity, while at the same time conserving those resources for the future (FAO, 1978). The term *land capability* is viewed as the inherent capacity of land to perform at a given level for a general use, or as a classification of land primarily in relation to degradation hazards (FAO, 1976). Land capability classification (*LCC*) is a system of grouping soils on the basis of their capability to produce common cultivated crops and pastureland, without deteriorating, over a long period of time (Soil Survey Staff, 1993). Geographic information systems (GIS) have greatly improved spatial data handling (Burrough and McDonnell, 1998), broadened spatial data

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[3] This reprint originally appeared as, Bakr, N., M.H. Bahnassy, M.M. El-Badawi, G.W. Ageeb, and D.C.Weindorf. 2009. Land capability evaluation in newly reclaimed areas: a case study in Bustan 3 area, Egypt. Soil Survey Horizon 51:90-95. Reprinted by permission of "Soil Horizons".

analysis (Bailey and Gatrell, 1995), and enabled spatial modeling of terrain attributes through digital elevation models (Moore *et al.*, 1991). The advent of GIS has brought about a whole set of new tools and enabled the use of methods that were not available at the time when the framework of land evaluation was developed (FAO, 1976, 2007).

The Cervatana model (De La Rosa, 2000) forecasts the general land use capability for possible agricultural uses. It represents one proprietary module of *MicroLEIS* software, which is an integrated system for agro-ecological land evaluation (FAO, 2007). This model has previously been applied in different areas in Egypt by Bakr (2003) and Gaber *et al*. (2003). Ali *et al*. (2007) reported that the capability classes of some old cultivated land and new reclaimed soils located west of the Nile Delta varied. The classes ranged from very high capability (*Class I*) for the old cultivated land to low capability (*Class III*) in the newly reclaimed areas, which were characterized by shallow soil depth, coarse texture, poor drainage, and salt accumulation.

The objectives of this study were to: (i) identify the land capability classes, (ii) categorize the limitation factors for sustainable land use planning in the *Bustan 3* area representing one of the new reclaimed areas in Egypt, and (iii) create scenarios to investigate the effect of improving the limitations on the general land use capability.

4.2 MATERIALS AND METHODS

4.2.1 Study area

The *Bustan 3* area (Figure 4.1) is one of the *Newlands* in the Western Nile Delta, Egypt and is in the process of being reclaimed. It is located between latitudes 3368500 to 3392000 N and longitudes 226000 and 255500 E, occupies around 341.27 km^2 (34,128 ha). The reclamation processes were taken place in this area during 1990s and nowadays the area is under agricultural activities (Figure 4.2).

The study area is characterized by a Mediterranean climate and can be considered semiarid. The *Bustan 3* soils are classified as Typic Torripsamments (Soil Survey Staff, 2006). Sadek (1993) reported that this area contains desert geo-morphic units such as sand dunes and sandy plains. The geological deposits represent the Pliocene, Holocene, and Pleistocene eras.

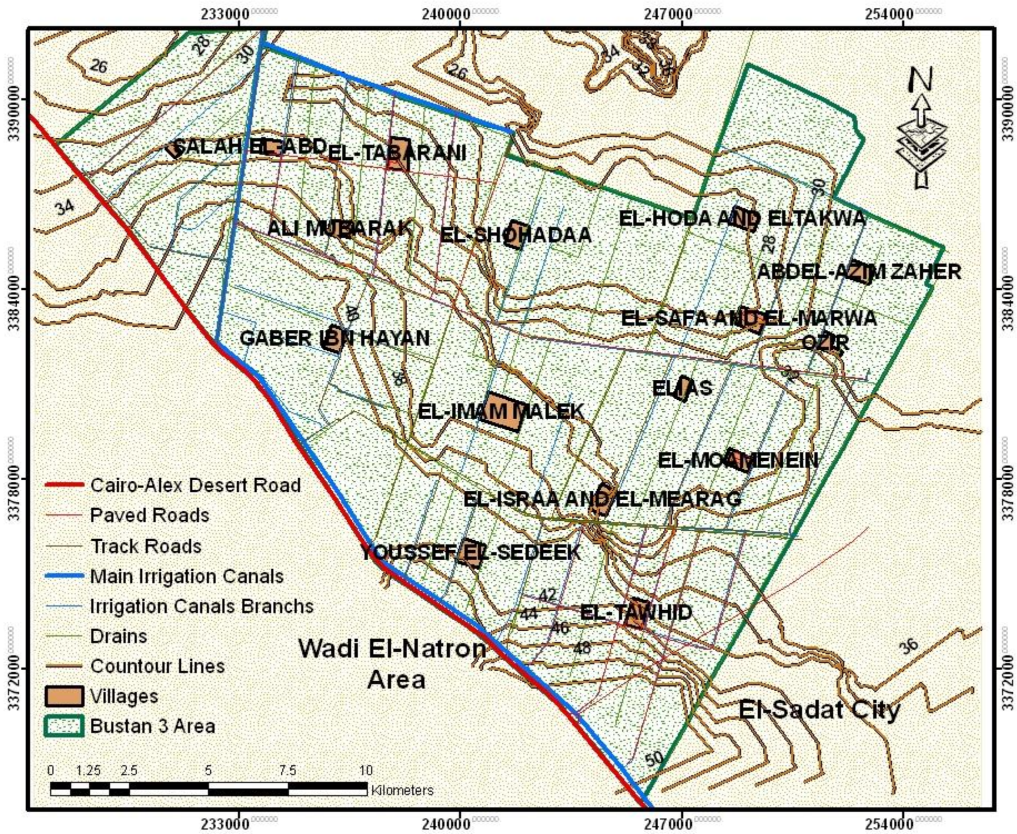

Figure 4.1. The *Bustan 3* area, Egypt.


Figure 4.2. The original *Bustan 3* land before start the agricultural production (left), and the one field in the *Bustan 3* area under agricultural activities (right).

4.2.2 Data sets

Several data sets were used in this research. First, advanced spaceborne thermal emission and reflection radiometer (ASTER) imagery acquired in 2007 (pass 177 and raw 39). Bands 1, 2, and 3 with 15 m X 15 m spatial resolution were used as a source of satellite imagery data (Figure 4.3). Satellite imagery was georeferenced (Jensen, 2004; Swann *et al.*, 1988) to convert the coordinate system to Universal Transverse Mercator (UTM) with a datum of "D_Egypt_1907". Second, the *Bustan 3* area topographic maps (1:50,000) were digitized. From digital topographic maps, the contour lines were extracted and edge-matched using ArcGIS 9.2 software (ESRI, 2001). A digital elevation model (DEM) was produced from the interpolation of vector contour lines using the spatial analyst tool in ArcMap 9.2. Finally, the High Dam soil survey map sheets (FAO, 1966) were digitized and transferred to ArcMap 9.2. Different physiographic units in the study area were extracted from the digital soil survey.

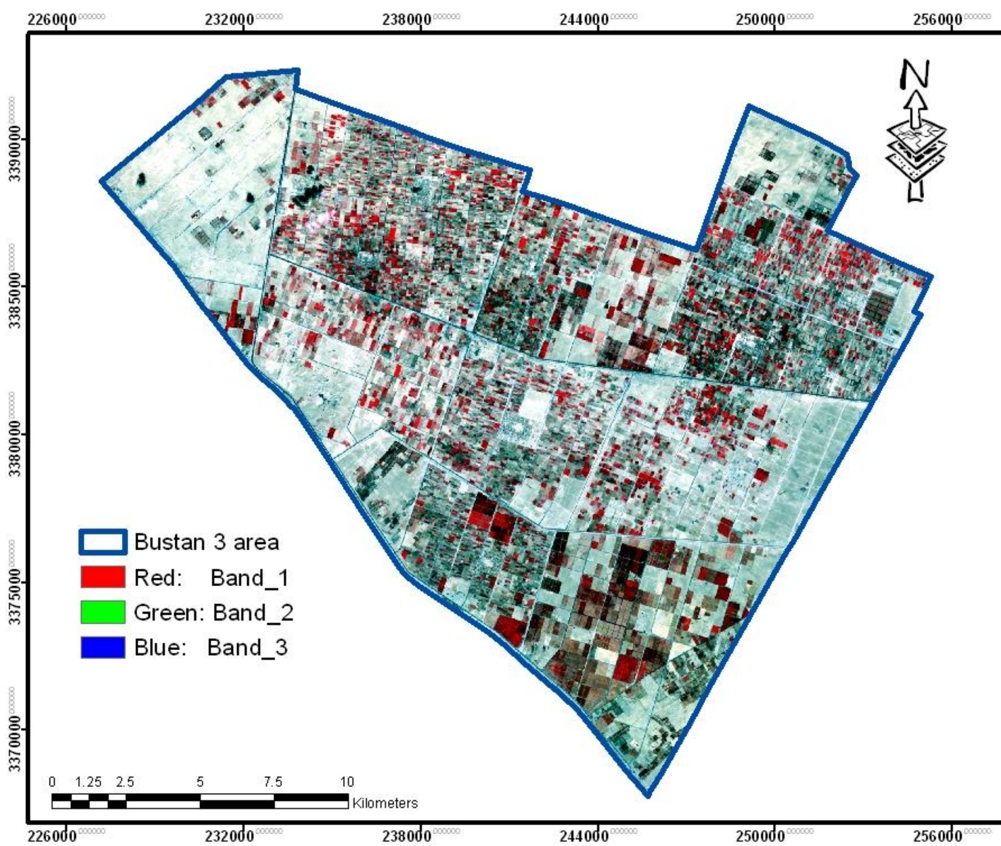

Figure 4.3. ASTER image for the *Bustan 3* area, Egypt.

4.2.3 Soil sampling and attributes data

Twenty-five profiles representing the different geomorphologic units were excavated (Figure 4.4) and morphologically described according to FAO (2006). The soil samples were air-dried, ground, and passed through a 2-mm sieve. The relative percentage large fragments (>2 mm) on the soil surface was computed and expressed as stoniness percentage. The fragment <2 mm was stored for further laboratory analysis. The soil salinity (dS m^{-1}) was determined using a 1:1 soil/water extract, and soil texture determined via dry sieving according to Page (1982), Klute (1986), and the Soil Survey Staff (2004). Using ArcMap 9.2, all attribute data were converted to georeferenced attribute tables.

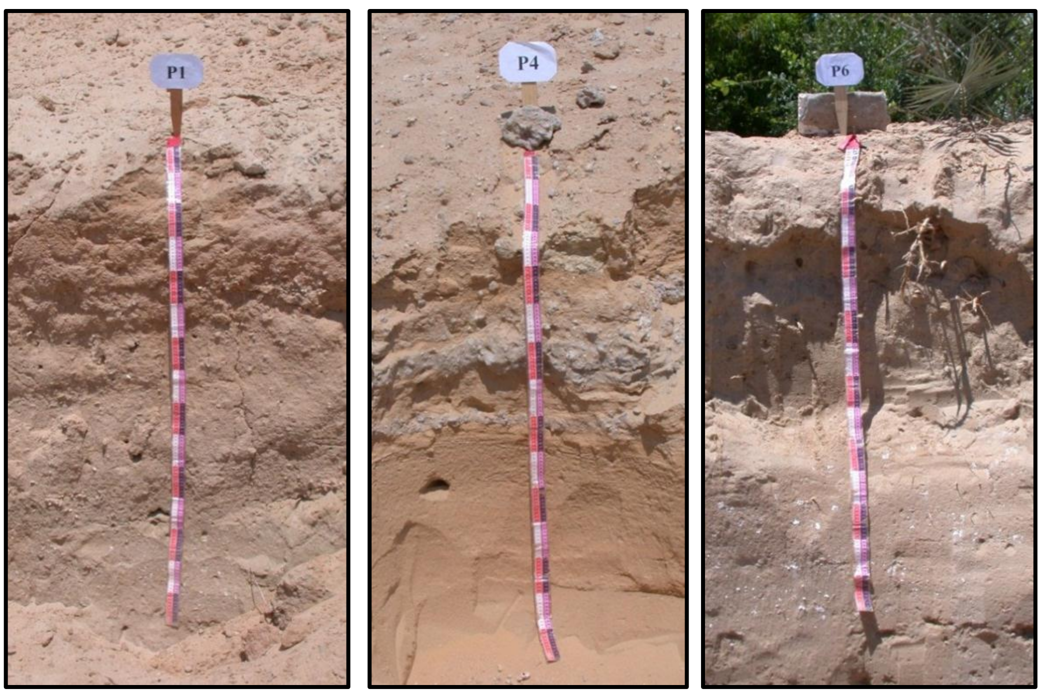

Figure 4.4. Examples of soil profiles collected from the *Bustan 3* area, Egypt.

Using ArcMap 9.2, all attribute data were converted to georeferenced attribute tables. The spatial distribution of all attributes was produced using inverse distance weighting (IDW) interpolation under ArcMap 9.2 to expand the point observations to incessant units (Burrough and McDonnell, 1998). The spatial analysis tool in ArcMap 9.2 was used to reclassify the interpolated maps, define the units, and calculate the areas of each unit.

4.2.4 Cervatana model description

The Cervatana model uses soil (l), site (t), erosion hazard (r), and bioclimatic deficit (b) factors to predict the general land capability class (De La Rosa, 2000). The prediction of general land use capability is the result of a qualitative evaluation process of the following biophysical factors: relief, soil, climate, and current use or vegetation. The land units are grouped in four classes. The first three (S1, S2, and S3) include land considered capable of supporting continuous, intensive agricultural production, while the land class (N) is for nonagricultural uses. Appropriate subclasses are established depending on the limiting factors. In each case, the most limiting criterion is given priority. The procedure of maximum limitation is used, with matrices of degree, to relate the characteristics directly with the classes of land use capability. Table 4.1 shows the parameters that were established for each limitation factor and classes of land use capability.

Some input data for the model result from soil sample analysis such as texture, stoniness percentage, and soil salinity. Other factors are based on field observations like drainage status and vegetative cover. Finally, some input data, such as slope gradient, are extracted directly from spatial analysis in ArcMap 9.2. Mathematical calculations are used to compute variables such as erodibility, erosivity, water deficit, and frost risk. Output land capability classes generated from the Cervatana model were represented as string data, where S1 is land with excellent use capability, S2 is land with good use capability, S3 is land with moderate use capability, and N is marginal or nonproductive land.

4.2.5 Scenario planning

On the basis of the model output data, three different scenarios were evaluated to predict the enhancement of land capability classes due to improvements in land capability limitation factors. A) Soil salinity scenario, in which the soil limitation factor can be enhanced by reducing

soil salinity from highly and very highly saline soil to moderately saline and from moderately saline to nonsaline. These results are possible when good quality water for leaching is available. B) Vegetation cover scenario, in which the erosion hazard limitation factor can be reduced if the soil is covered by vegetation. The vegetative cover was improved from none or low vegetation to moderately vegetated and from moderately vegetated to highly vegetated. C) Combined scenario, in which both of the aforementioned scenarios were implemented simultaneously by holding all other variables constant.

Table 4.1. Parameters established for the each limitation factor and classes of land use capability for Cervatana model.

		Site (*t*)		
Slope (%)	Gentle (<7)	Moderate (7-15)	Strong (15-30)	Steep (>30)
Class	S1	S2*t*	S3*t*	N*t*
		Soil (*l*)		
Useful depth (cm)	High (>75)	Moderate (50-75)	Shallow (25-50)	Superficial (<25)
Texture	Balanced	Slight/Heavy	---	---
Stoniness (%)	Slight (<15)	Moderate (15-40)	High (>40)	---
Drainage	Good	Moderate	Deficient/Excessive	---
Salinity (dS/m)	Nil or Slight (<4)	Moderate (4-8)	High (8-12)	Very high (>12)
Class	S1	S2*l*	S3*l*	N*l*
		Erosion hazard (*r*)		
Erodibility	Slight	Moderate	High >30	---
Slope (%)	<15	15-30	---	---
Vegetation	High	Moderate	Nil	---
Erosivity	Slight (<150)	Moderate (150-200)	Strong (200-300)	Very strong(>300)
Class	S1	S2*r*	S3*r*	N*r*
		Bio-climatic Deficit (*b*)		
Water deficiency	Low (h1)	Moderate (h2)	High (h3)	Very high (h4)
Frost risk	Slight (f1,f2)	Moderate (f3)	High (f4)	---
Class	S1	S2*b*	S3*b*	N*b*

4.3 RESULTS AND DISCUSSION

The different geomorphological units in the *Bustan 3* area, Egypt and the locations of soil profiles are shown in Figure 4.5. Around 65% of the study area is in the “windblown sand” physiographic unit. The southern part of the study area belongs mainly to deltic stage and river terrace units. The DEM analysis indicated that the elevation of the study area ranged from 24 to 51 m, with more than 50% of area with an elevation between 30 and 40 m (Figure 4.6a). Based on the DEM, slope analysis showed that the area is gently sloping or almost flat. The slope gradient is very low, from 0 to 3.5% (Figure 4.6b).

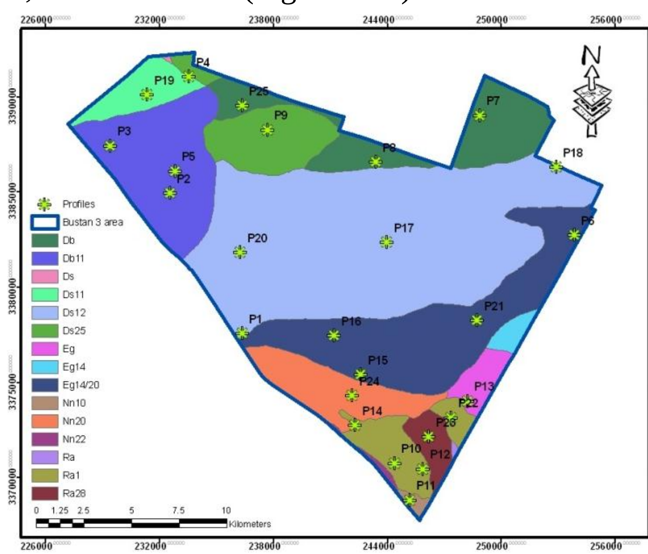

Figure 4.5. Geomorphic units in *Bustan 3* area, Egypt.

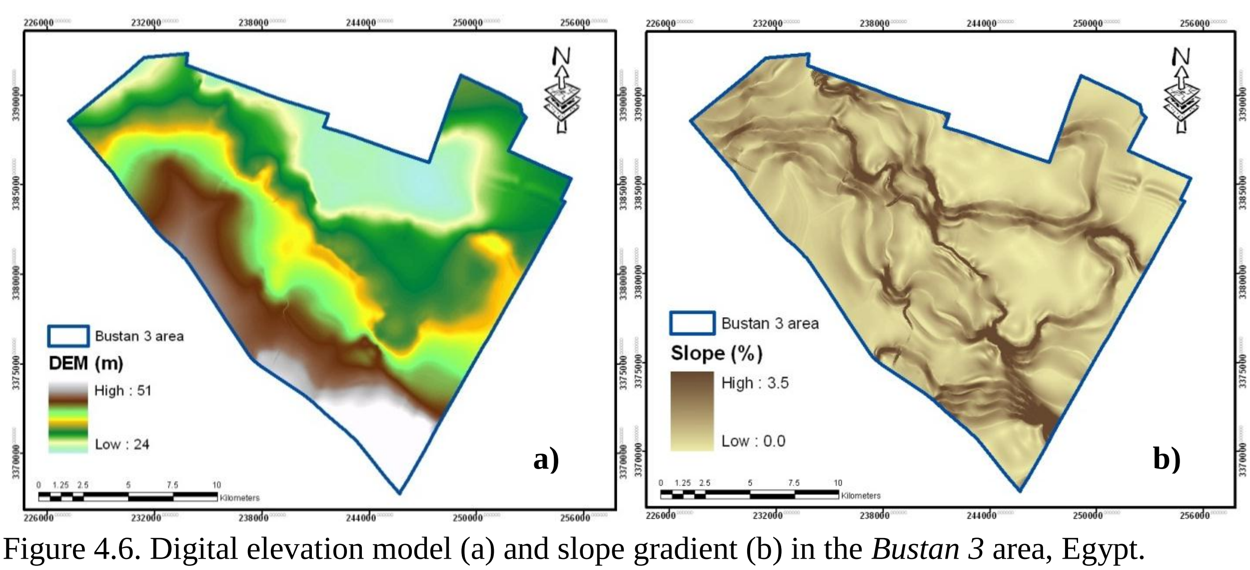

Figure 4.6. Digital elevation model (a) and slope gradient (b) in the *Bustan 3* area, Egypt.

Soil depth for almost all of the area is optimum (≥150cm) except for two profiles (P3 and P16), where hardpans exist at 55 and 15 cm, respectively. Particle size analysis showed that the texture of soil across the whole area is sand. The drainage status for most of the area is excellent, with the exception of shallow hardpan affected areas and excessive drainage in coarse stony textures (Table 4.2). Soil salinity ranged from 0.2 to 16.66 dS m^{-1} (Table 4.2). Approximately 80% of the study area has none to slight salinity. The southeastern part of the *Bustan 3* area has a higher salinity compared with other parts in the study area. Only profile P13 (EC = 16.66 dS m^{-1}) was very strongly saline (Figure 4.7a). Coarse fragment percentage (stoniness) varied from 0 to about 52% (Table 4.2). Around 74% of the area has null to slight stoniness, and about 25% is slightly to moderately stony. The stony area is located in the southern part of the *Bustan 3* area (Figure 4.7b). Visual interpretation of satellite imagery and fieldwork observations of ground cover showed that more than 50% of the *Bustan 3* area has low vegetative cover or is not cultivated. Erodibility, which is related to wind erosion hazard, is judged according to three soil characteristics: depth, texture, and stoniness. Based on existing data, the soil erodibility of the study area ranged from low to moderate. Erosivity, which is related to rainfall, is calculated from the Fournier index (FAO, 1977) and humidity index. Erosivity in Egypt is always low due to the semiarid climate and low annual precipitation. Each profile has its own capability class with different limitation factors.

To manipulate the string data in a GIS environment, the classes were rated mathematically from 0 to 1 as: 0.25 for class N, 0.50 for class S3, 0.75 for class S2, and 1.0 for class S1. When present, each limitation factor was rated equally (0.0625) and subtracted from each class. Consequently, the output rate was 0.0 to 0.25 for N, 0.25 to 0.50 for S3, 0.50 to 0.75 for S2, and 0.75 to 1.0 for S1. For example, consider an output of class S2lr. The maximum value of S2 (0.75) was reduced by the two limiting factors (l and r, each with a value of 0.0625)

to 0.63. Land capability classes varied from good (S2) to marginal (N). Table 4.3 shows the land capability classes and limitation factors for each soil profile.

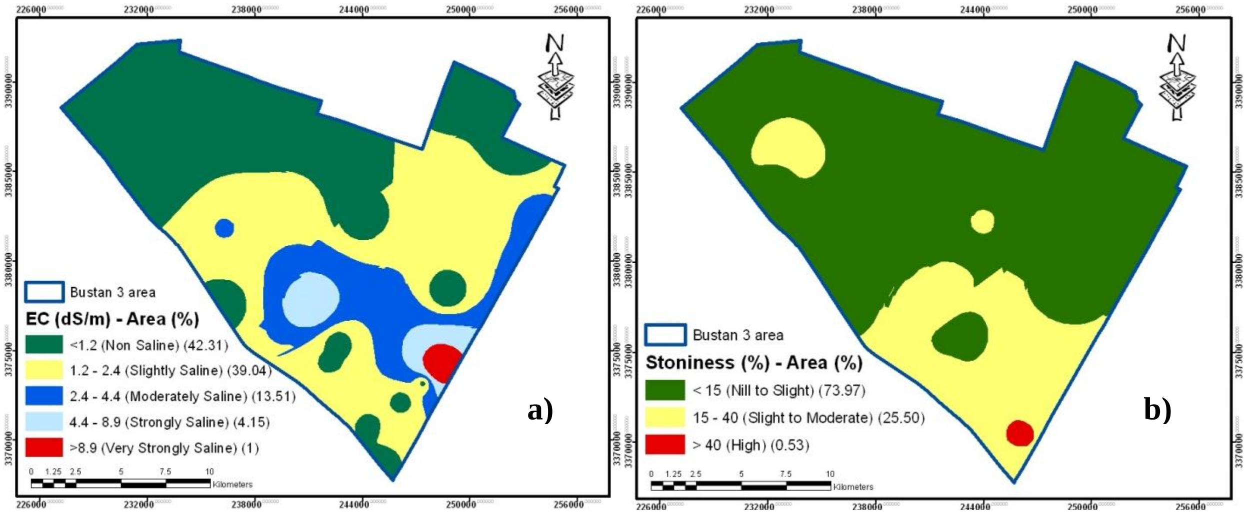

Figure 4.7. Spatial distribution of, a) electrical conductivity and b) stoniness (%) in the *Bustan 3* area, Egypt.

Table 4.2. Soil characteristics used as input in Cervatana model.

Soil Sample	Depth (cm)	Soil Texture	Stoniness (%)	EC (dS/cm)	Drainage
P1	150	Coarse Sand	13.02	0.53	Moderate
P2	150	Coarse Sand	5.54	0.56	Well
P3	55	Coarse Sand	0.58	0.35	Moderate
P4	150	Coarse Sand	7.95	0.38	Well
P5	150	Coarse Sand	39.56	0.79	Moderate
P6	150	Coarse Sand	13.04	2.66	Well
P7	150	Coarse Sand	9.77	0.92	Well
P8	150	Coarse Sand	3.62	0.85	Well
P9	150	Coarse Sand	13.79	0.65	Well
P10	150	Coarse Sand	26.61	0.86	Well
P11	150	Coarse Sand	17.23	0.68	Well
P12	150	Coarse Sand	52.47	1.55	Excessive
P13	150	Coarse Sand	21.27	16.66	Well
P14	150	Coarse Sand	30.43	1.72	Well
P15	150	Coarse Sand	0.00	0.50	Well
P16	15	Coarse Sand	20.27	7.71	Well
P17	150	Coarse Sand	15.50	0.53	Well
P18	150	Coarse Sand	3.28	0.34	Well
P19	150	Coarse Sand	0.00	0.44	Well
P20	150	Coarse Sand	0.00	2.49	Well
P21	150	Coarse Sand	6.43	0.28	Well
P22	150	Coarse Sand	15.74	1.01	Well
P23	150	Coarse Sand	37.04	0.57	Moderate
P24	150	Coarse Sand	19.65	0.87	Well
P25	150	Coarse Sand	0.00	0.20	Well

Table 4.3. Land capability classes for each profile in the *Bustan 3* area, Egypt.

Soil Sample	Capability class	Rating
P1	S3r	0.44
P2	S2lr	0.63
P3	S3r	0.44
P4	S3r	0.44
P5	S3r	0.44
P6	S2lr	0.63
P7	S3r	0.44
P8	S2lr	0.63
P9	S2l	0.69
P10	S2lr	0.63
P11	S3r	0.44
P12	S3l	0.44
P13	Nl	0.19
P14	S3r	0.44
P15	S2lr	0.63
P16	Nl	0.19
P17	S2l	0.69
P18	S2lr	0.63
P19	S2lr	0.63
P20	S3r	0.44
P21	S2lr	0.63
P22	S2l	0.69
P23	S2l	0.69
P24	S3r	0.44
P25	S2lr	0.63

The spatial distribution analysis for the model's output data indicated that approximately 70% of the study area has a good capability for agriculture use (S2), while the rest (around 30%) has moderate capability (S3). The model's results show that erosion risk (r) and soil (l) factors are the dominant limiting factors in this area (Figure 4.8a). Based on the output of the land capability evaluation, three different scenarios were evaluated by changing one factor and keeping the other factors constant. The results of the scenario analysis follow:

- Scenario 1 output data indicated that reducing salinity improved the land use capability. The good capability class for agriculture use (S2) increased to 73% instead of 70% in the original data (Figure 4.8b). This increase is due to the adjustment of profile P13. In the original data, it had a capability class of Nl. After scenario 1 was applied the capability class improved to S3r.

As a result, interpolations of the new value of P13 increased the acreage area of S2 to 73%. Although, the change in land capability class area was not extremely high as a value, the ecological effect of the change is substantial since the reduction in soil salinity will lead to more suitable land for agronomic production. Leaching to decrease the soil salinity would be accomplished carefully, with precision controlled irrigation technologies such as drip and sprinkler irrigation. This would allow for leaching of the salts below the rooting zone of most crops, but not so deep as to increase groundwater salinity.

- Scenario 2 concerned vegetation cover as an important factor because of its effect on the erosion hazard. Changing the land cover status from noncultivated to cultivated, the wind erosion hazard was reduced, and therefore the land capability increased. While this seems counterintuitive initially, recall that the study area is dominantly dune sand with little to no vegetative cover and is subject to severe wind erosion. Establishing any kind of vegetative cover, even in form of cultivated row crops, reduces wind erosion hazard and increases land use capability by stabilizing the surface. Figure 4.8c shows that the good capability class for agriculture use (S2) represented about 93% of the study area after cultivation instead of 70% before cultivation. For example, soil profiles P3, P4, P5, P7, P11, P14, P20, and P24 have capability class S3r in the original output data model. After implementation of this scenario, they become S2lr. This change dramatically increased the acreage of (S2) class to 93%.

- Scenario 3 evaluated the change in the two variables simultaneously (soil salinity and vegetation cover). The vast change occurred due to improvements in the vegetative cover status (as in the second scenario) combined with minor changes in the first scenario. Results (Figure 4.8d) show that after implementation, the good capability class for agriculture use (S2) occupied about 96% of the study area instead of 70% in the original data.

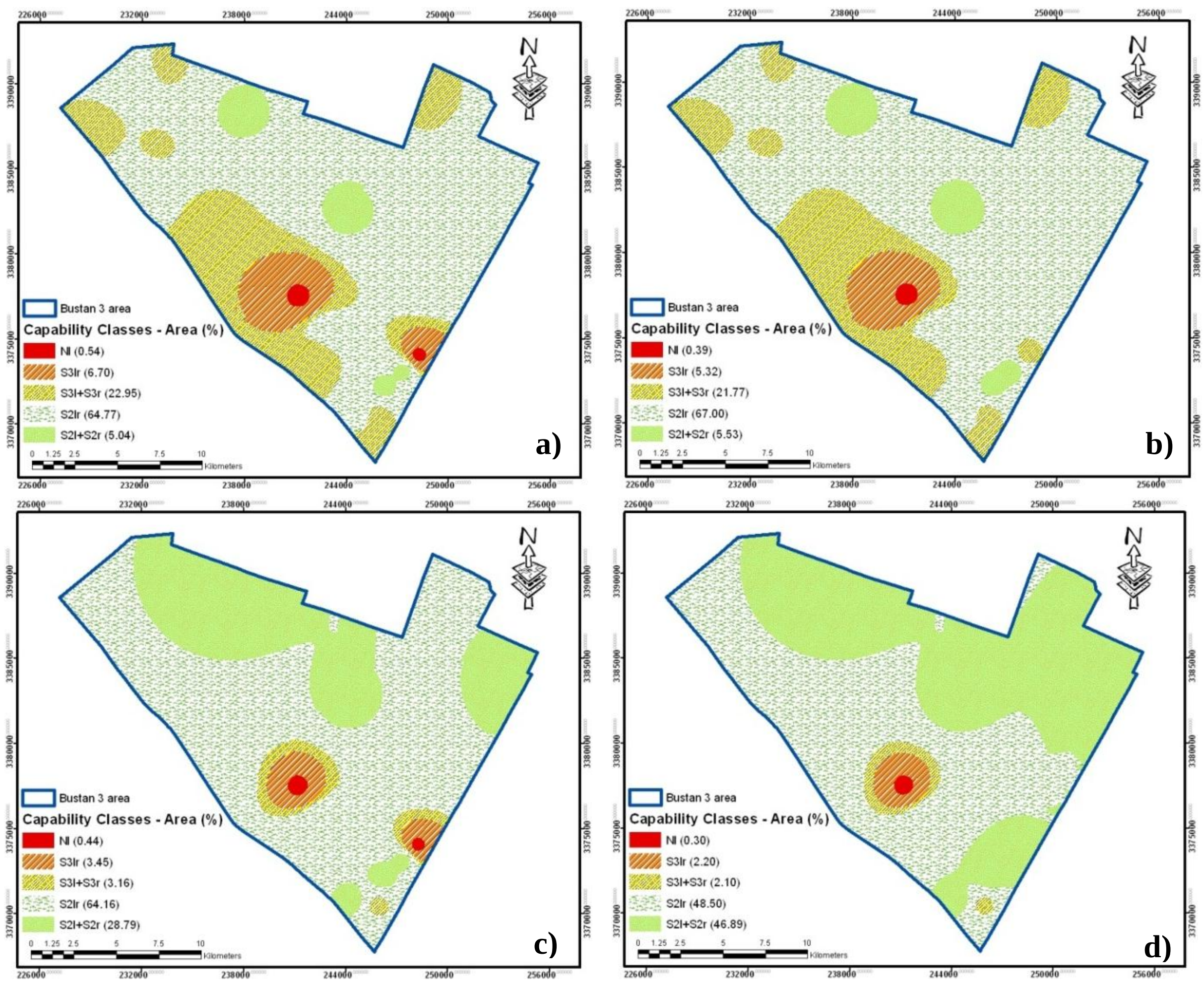

Figure 4.8. Spatial distribution of capability classes for the *Bustan 3* area, Egypt, using; a) current model, b) Scenario 1, c) Scenario 2, and d) Scenario 3.

4.4 CONCLUSIONS

Computerized land evaluation systems and land use models can be advantageous in estimating and predicting the land performance if sufficient data exist. This research indicated that the use of GIS has significantly improved spatial data handling and analysis and has enabled spatial modeling of terrain attributes through digital elevation models and interpolation of terrain attributes tables. This study concluded that the *Bustan 3* area, Egypt, has a good capability for agricultural production. Therefore, the coupling between modeling and GIS serves to improve land use planning and consequently enhance the decision-making process, especially in newly reclaimed areas.

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CHAPTER 5. MONITORING THE CHANGES IN ROADSIDE SOIL PROPERTIES CORRESPONDING TO COMPOST/MULCH APPLICATION

5.1 INTRODUCTION

Soil erosion occurrence and intensity can be strongly influenced by soil properties. No single property causes soil erodability; almost any soil property may influence erosion response. However, soil type, climatic conditions, and soil management often have a direct effect on soil erosion. In practice, soil aggregation, consistency and shear strength usually influence soil erodability. Collectively, all soil properties influence water movement, the distribution of erosive forces, and soil resistance (Bryan, 2000). Texture and clay mineralogy appear to be the most significant factors determining the sensitivity of a soil to inter-rill erosion induced by drying (Kuhn and Bryan, 2004). The most susceptible soils to erosion are soils with high silt, low clay, and low organic matter (Wischmeier and Mannering, 1969).

Soil moisture data are fundamentally important for a wide variety of agricultural, engineering, environmental science, and hydrological applications. Soil moisture content is key in hydrological processes occurring at or near the land surface and regulating the rate of infiltration, runoff, storage in the root zone, percolation to ground water, evapotranspiration, and water availability to plants (Sheikh *et al.*, 2009). Beyond soil moisture, soil temperature has an important role in soil physical, chemical, and biological properties which extends to its role on soil taxonomy, plant growth, and crop yield. Temperature influences the rate of organic matter decomposition which consequently affects soil structure and water movement in the soil (Tenge *et al.*, 1998). The relationship between soil temperature and soil moisture is related to the soil texture. Olmanson and Ochsner (2006) found that an increase in ambient temperature caused increased soil moisture regardless of moisture levels or soil types with the exception of sand at low water content. Letey (1985) introduced the concept of a nonlimiting water range in soils that

described the importance of soil moisture (θ) on soil physical properties affecting plant growth. Considering that concept, the decreasing water content increases the mechanical resistance of soils, soil temperature, and aeration.

The significant influence of rainfall and soil surface characteristics on soil erosion have been confirmed in different studies. Best management practices (BMPs) can be effectively used to control soil erosion through decreasing the erosive power of raindrops, runoff velocity, and increasing soil infiltration rate. Organic residue, compost, and mulch have been used as successful BMPs for soil and water conservation for many decades. The application of residue on the soil surface without tillage incorporation is commonplace; numerous studies have confirmed the effectiveness of "no-till" in conserving soil and water resources when crop residues were used for reducing runoff and soil loss (Ruiz-Colmenero *et al.,* 2011; Bhatt and Khera, 2004). Weindorf *et al*. (2006) indicated that the use of compost significantly reduced the coefficient of linear extensibility and increased the water content in Texas soils. When rain falls on recently tilled soil, the loose soil surface can produce a crust that changes the initial soil characteristics (Kuhn and Bryan, 2004). While tillage practices may provide temporary infiltration benefits, longer term changes in soil structure could lead to sealing and increased runoff and erosion (Ruiz-Colmenero *et al.,* 2011).

Considering the term *''greenhouse effect''*, mulches increase the soil temperature by trapping solar energy that passes through the mulch to beneath soil then heat the soil. Additionally, mulches greatly retard the loss of moisture from the soil which maintains higher and more uniform soil moisture (Ramakrishna *et al.,* 2006). The best sustainable management scenario for effectively reducing soil loss, decreasing soil temperature, and increasing moisture content can be achieved by using minimum tillage coupled with a mulch cover (Bhatt and Khera, 2004). Based on Olasantan (1999), mulch was shown to reduce nutrient losses by runoff, erosion,

and leaching. Additionally, mulch coverage decreased maximum soil temperature by 2-7°C and conserved additional moisture (50-120 g kg^{-1}), as compared to zero mulch. Findeling *et al*. (2003) found that the use of mulch can effectively reduce runoff, enhance flow rate, increase soil roughness, and, over the long term, mulch can conserve topsoil and increase water conductivity. A 5 year study conducted by Xin-Hu *et al*. (2011) indicated that grass cover and mulch were highly effective in conserving soil and water on steep slopes (25%). Thus, soil physical properties are highly affected by mulch cover. Based on a 3 year study by Jordán *et al*. (2010), increasing the rate of straw mulch on soils led to increased soil porosity, stability of aggregates, and organic matter content, and a decrease in bulk density. Additionally, runoff rate and soil loss were highly reduced as a result of mulch cover. In a study of soils that receive intense rain over a short time and on steep slopes, Ramakrishna *et al. (*2006) found the use of organic mulches was better economically and environmentally compared to polythene and chemical mulch. Dahiya *et al*. (2007) found that the use of mulch reduced soil water loss on average by 0.39 mm d^{-1} compared to a control plot.

In Louisiana, intensive precipitation is commonplace, with an annual average of 119-180 cm. Furthermore, the state is rife with loose alluvial sediment (Weindorf, 2008). The combination of the aforementioned factors leaves Louisiana topsoils highly susceptible to erosion. Finally, roadside soils usually feature considerable slopes and leave disturbed soils exposed to erosion hazards. The main goal of this study was to evaluate and monitor the effect of compost/mulch applications on physiochemical properties of roadside soils in Louisiana.

5.2 MATERIALS AND METHODS

5.2.1 Study site description

Soil samples were collected from four study sites in 2010-2012; one located at US Highway 61 in West Feliciana Parish and three located at IH-49 in Rapides Parish, Louisiana,

USA (Figure 5.1). Slopes at the sites ranged from 34% to 10%, and featured inherent differences in soil properties. Additionally, US Highway 61 was under construction activities during 2010/2011 while IH-49 was in an established area prone to rill and gully soil erosion in different areas along the roadside. The sites involved eight plots; two plots at each site (side-by-side), at a fixed size of 4.0 m X 4.0 m.

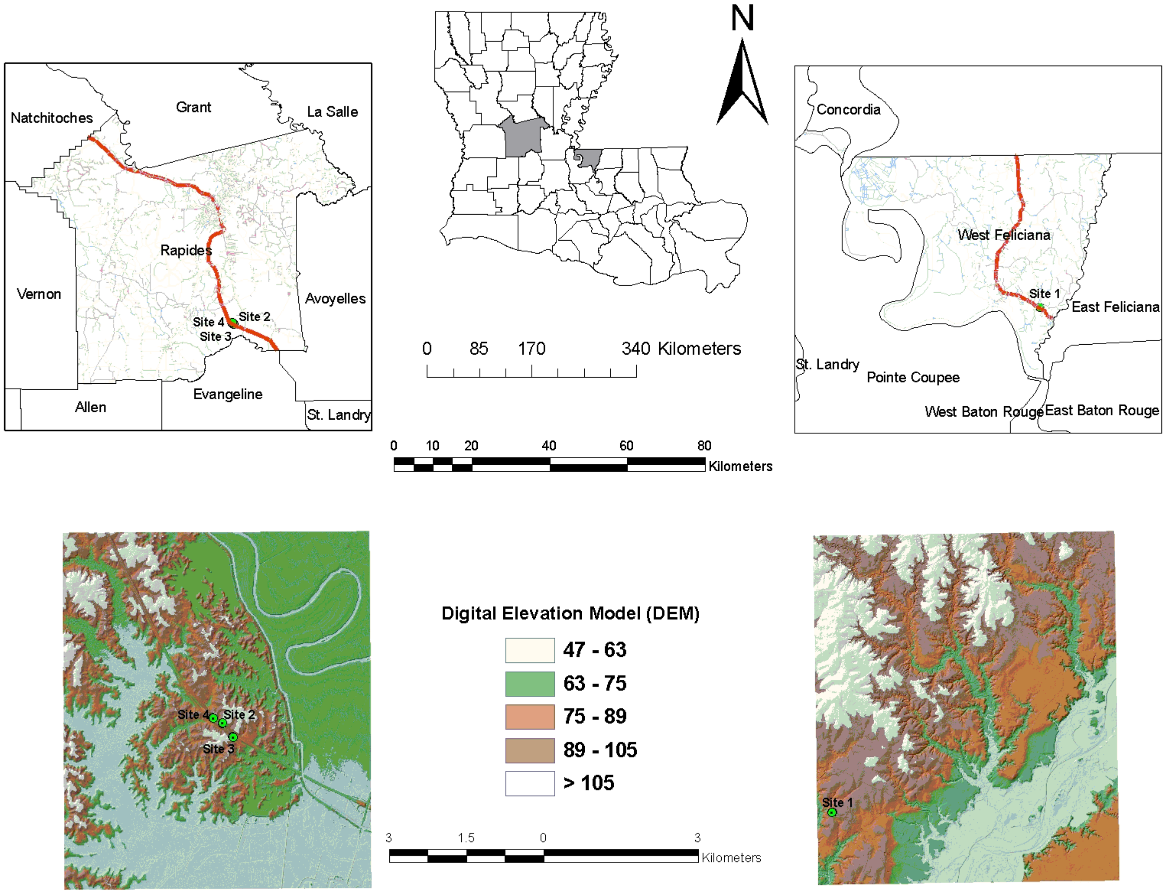

Figure 5.1. The four studied sites in West Feliciana and Rapides parishes, Louisiana, USA.

5.2.2 Management practices

For each study site, one plot was lightly tilled at the beginning of the experiment (February, 2010) and the other one kept untilled. The compost/mulch was used as a BMP to reduce the soil erosion from the highway right-of-ways. The compost/mulch materials used in this study involved wood chips (70% hardwood and 30% pine trees harvested locally) while the compost was a double-ground, screened, recycled wood fiber material, also harvested locally

(Bakr *et al.*, 2012). Three compost/mulch treatments were applied on target plots; 0 as control, 5cm, and 10cm compost/mulch coverage. The treatment scheme, plot identification and description, and slope percentage are presented in Table 5.1.

Table 5.1. Compost/mulch thickness, location, slope, tillage, and identification for each plot at each site in Louisiana, USA.

Sites	1		2		3		4	
Slope (%)	34		25		15		10	
Location	US Highway 61		IH-49		IH-49		IH-49	
Construction	Active		Established		Established		Established	
Plot	S1A	S1B	S2A	S2B	S3A	S3B	S4A	S4B
C/M*thickness (cm)	10	5	10	0	5	5	0	10
Tillage	no-tillage	light-tillage	no-tillage	light-tillage	no-tillage	light-tillage	no-tillage	light-tillage

* Compost/mulch

5.2.3 Soil sample collection

As a slope factor was considered in this study, surface soil samples (0-10cm) were collected at the beginning (2010) and the end of the experiment (2012), from up and down slope within each plot as well as the area in between the two plots at each site. The soil texture laboratory results were used to evaluate the spatial distribution of silt and clay fractions in each plot. The inverse distance weighting (IDW) interpolation technique in spatial analyst tools under ArcMap 10 (ESRI, 2011) was used to produce the spatial distribution maps. Two sets of soil samples were collected in 2010 and 2012. The first set was collected in February, 2010 during the plot preparation and equipment installation, while the second set was collected after project conclusion in May, 2012. Collected samples were dried and ground to pass a 2mm sieve prior to laboratory analysis. Parameters evaluated included soil texture by pipette method (Gee and Bauder, 1986), pH (1:1, soil:water extraction) via an Orion 2 Star pH meter (Thermo Scientific, Waltham, MA), and soil organic matter (SOM) via loss-on-ignition (LOI) (Nelson and Sommers, 1996). Additionally, soil samples were extracted by Mehlich-3 extraction solution (Mehlich,

1984) to determine nutrient and heavy metal concentrations and then were measured via inductively coupled plasma atomic emission spectrophotometer (ICP–AES).

5.2.4 Soil moisture/temperature data collection

Soil temperature and soil moisture data were continuously recorded in all plots from March 2010 to May 2012. Eight HOBO® Micro Station (H21-002) dataloggers (Onset Computer Corporation, Bourne, MA, USA), were installed to record soil temperature and moisture from each plot in-situ. Two 12-Bit Temperature Smart Sensors (S-TMP-M006) and two Soil Moisture Smart Sensors (S-SMx-M005) were attached to each datalogger. Each soil moisture/temperature pair was placed up and down slope within each plot. The soil moisture and temperature were recorded every 2 min and a 10 min average was read. Data was regularly downloaded using HOBOware® Pro Software Version 2.3.0 (On-set Computer Corp., Bourne, MA, USA) and then filtered to obtain weekly averages.

5.2.5 Statistical analysis

The main goal for the statistical analysis was to detect if there was any difference in the soil physiochemical properties due to the compost/mulch treatments. Boxplots were developed using the PLOT statement of the BOXPLOT procedure in SAS® 9.3 software (SAS, 2011). Boxplots display measurements with respect to their mean, median, quartiles (25% and 75%), minimum, and maximum observations for each group of data. A one way analysis of variance (ANOVA) was used, in which the t-test and the differences of least squares means were calculated using a significance level of $p<0.05$. The PROC MIXED procedure in SAS® 9.3 software (SAS, 2011) was used to perform the analysis. The Tukey-Kramer technique was used to compare the least squares means between different plots corresponding to each soil property under investigation.

5.3 RESULTS AND DISCUSSION

5.3.1 Soil moisture and temperature

Weekly average soil moisture and soil temperature for each plot at each site are presented in Figures 5.2 and 5.3, respectively. The soil moisture results indicated that the surface soils in the control plots (S2B and S4A) were drier and highly fluctuated compared to the plots that were treated by compost/mulch (Figure 5.2b and d). Additionally, the results showed that for the lightly tilled plot (S1B, S2B, and S3B) the soil moisture values were less than the adjacent untilled plots (Figure 5.2a, b, and c). The only exception to this was plot S4B, since it was treated with 10 cm compost/mulch. With this higher rate of compost/mulch, even with tillage practices, the surface soil for this plot had higher moisture compared to the adjacent control plot (S4A) which was untilled (Figure 5.2d). The results also explained that the application of 10 cm compost/mulch with no-tillage (S1A and S2A) caused an increase in soil moisture when compared to 5 cm compost/mulch application or control plots (Figure 5.2a and b). Furthermore, the 10 cm compost/mulch application kept soil moisture more stable compared to the tilled plots, 5 cm compost/mulch treated plots, or the control plots; all of which had higher fluctuation in soil moisture values (Figure 5.2).

Regardless the differences in compost/mulch treatments, slope, or tillage practices, all plots showed the same pattern of temperatures being lowest during winter months (December-February) and highest during summer months (June-August) (Figure 5.3). However, results showed that the control plots (S2b and S4A) had higher temperatures during the summer season and lower temperatures in the winter season compared to the adjacent compost/mulch plots at both sites 2 and 4 (Figure 5.3b and d). The control plots also had higher temperatures at the soil surface compared to the adjacent compost/mulch plots for each site. Moreover, soil temperatures fluctuated more in both control plots compared to compost/mulch treated plots. Compost/mulch

treated plots (5- or 10 cm) effectively moderated soil temperatures. Specifically, the temperatures of topsoil in the treated plots were moderately cooler during the summer season and warmer during the winter season compared to the control plots.

In order to quantify the difference in water content as well as soil temperature based on the compost/mulch treatments, boxplot and ANOVA statistical analyses utilized. Figure 5.4 shows the boxplots that represented the distribution of soil water content and soil temperature within each plot. The results indicated that the highest moisture value (0.36 cm^3 cm^{-3}) was associated with the thickest compost/mulch application (10 cm) with no-tillage (S1A and S2A). By reducing the compost/mulch coverage to 5 cm, soil moisture of 0.35 cm^3 cm^{-3} was obtained. The lowest soil moisture values were found in the control plots, S2B and S4A, with values of 0.28 and 0.26 cm^3 cm^{-3}, respectively (Figure 5.4a). Contrary to soil moisture results, the soil temperature values were the highest for the control plots with plots S2B and S4A achieving soil temperatures of 38° and 37°C, respectively. With 5 cm compost/mulch, maximum temperatures were reduced to around 33°C. No change in the maximum soil temperature was observed by applying 10 cm compost/mulch (Figure 5.4b). In ANOVA, the Tukey-Kramer technique was used by comparing the difference in least squares means in soil moisture and temperature between the plots with a significance level of 0.05. The results showed no significant differences in soil temperature data between all plots. However, there were significant differences between plots for the soil moisture data. Table 5.2 displays only the statistically significant p values based on soil moisture data.

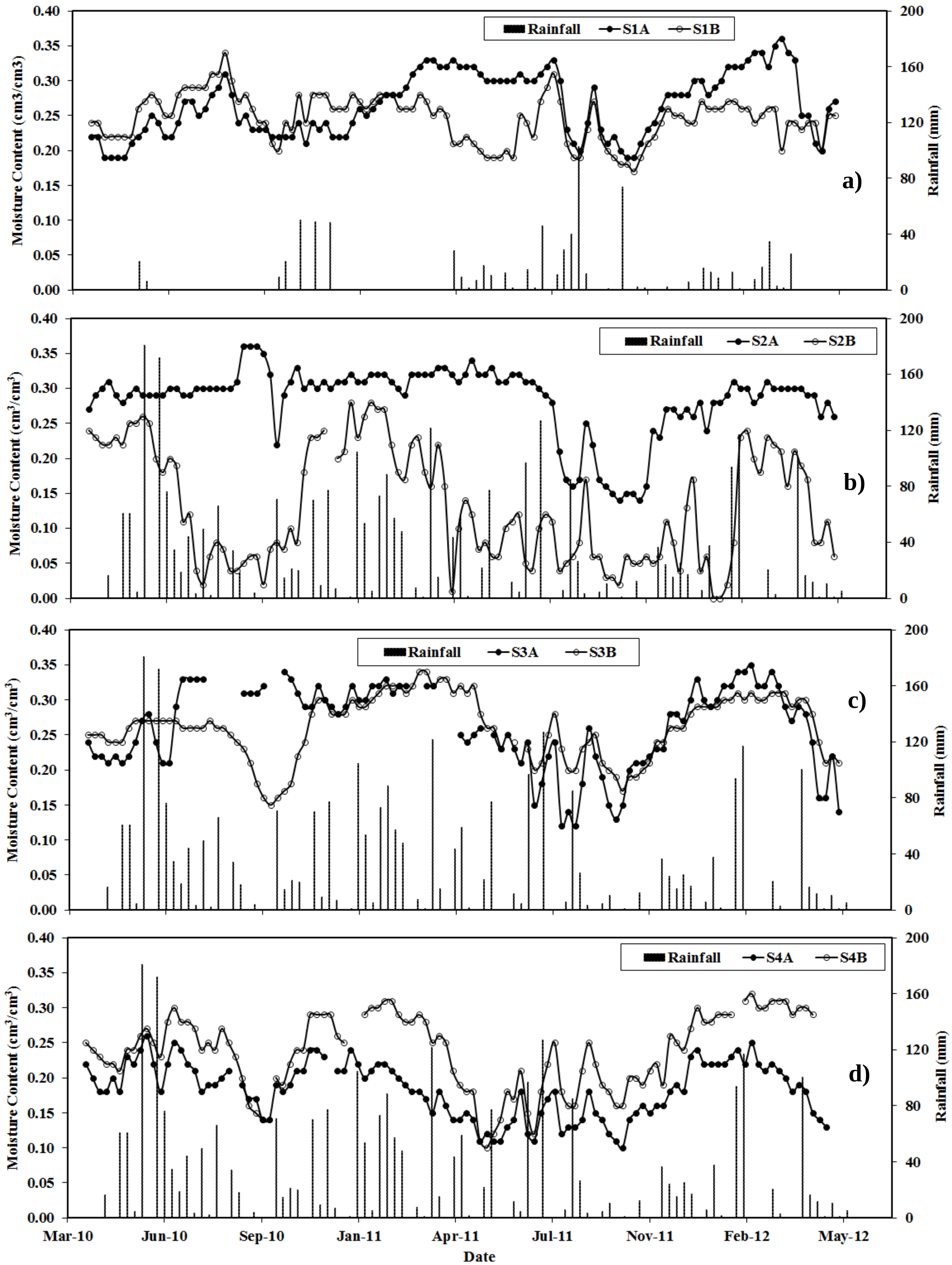

Figure 5.2. Weekly average of soil moisture and monthly summation of rainfall at: a) site 1, b) site 2, c) site 3, and d) site 4 in Louisiana, USA.

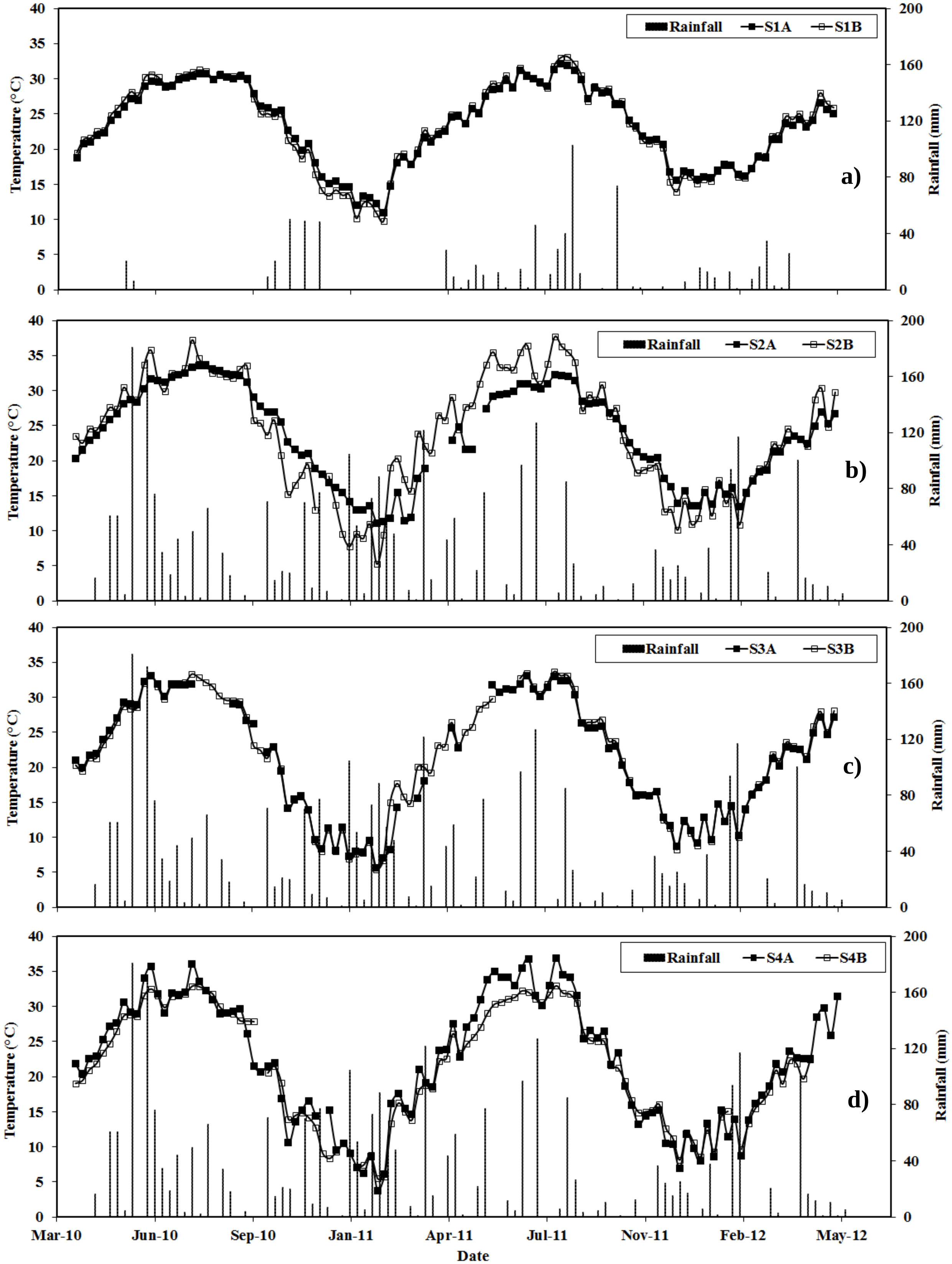

Figure 5.3. Weekly average of soil temperature and monthly summation of rainfall at: a) site 1, b) site 2, c) site 3, and d) site 4 in Louisiana, USA.

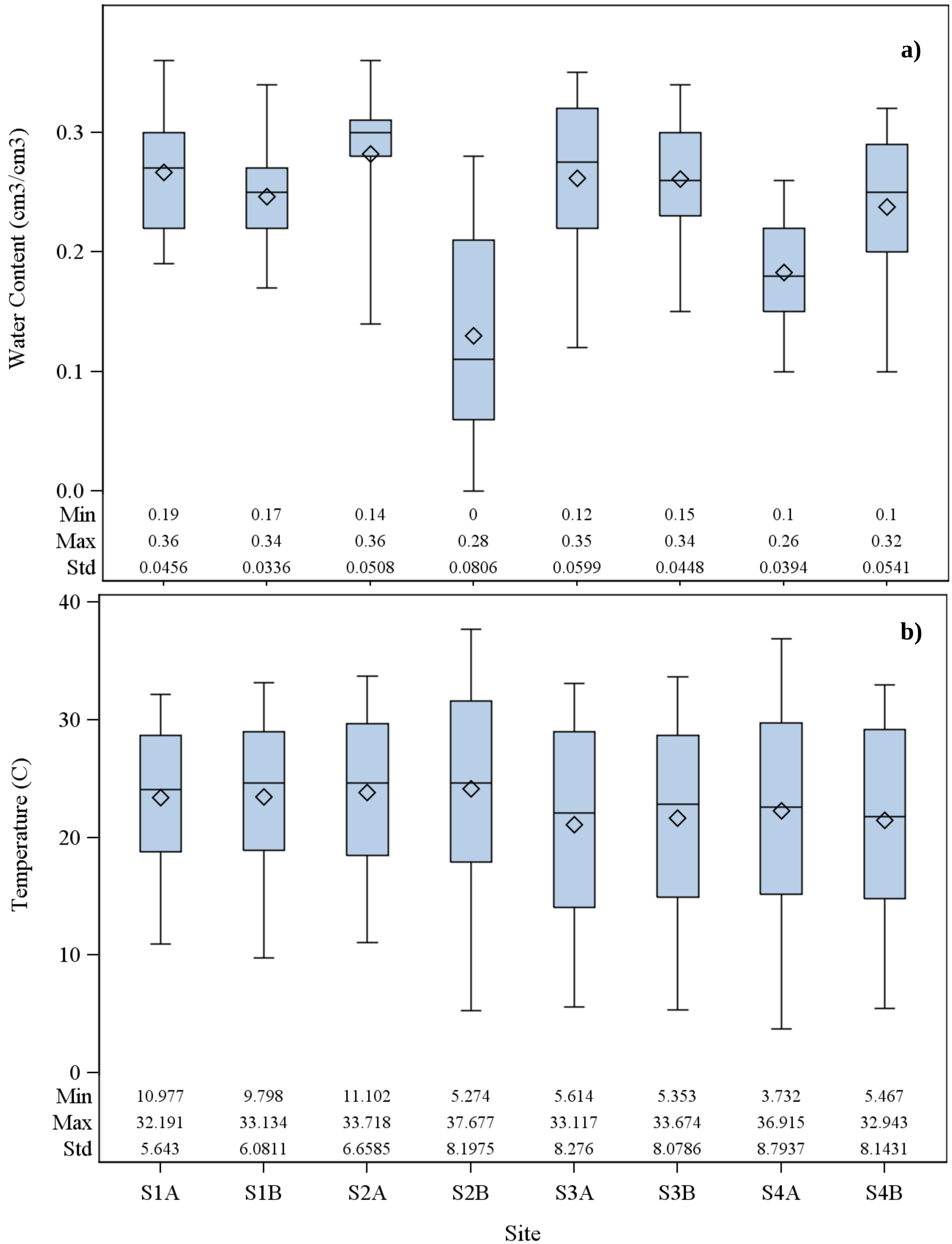

Figure 5.4. Boxplot of soil moisture (a) and temperature (b) with minimum, maximum, and standard deviation at each studied plot for compost/mulch amended roadside soils in Louisiana, USA.

Table 5.2. Significant p values (at significant level=0.05) based on soil moisture data for compost/mulch amended roadside soils in Louisiana, USA.

Sites		**P**	**Sites**		**P**
S1A	S2B	<0.0001	S2B	S3A	<0.0001
	S4A	<0.0001		S3B	<0.0001
	S4B	0.0019		S4A	<0.0001
S1B	S2A	<0.0001		S4B	<0.0001
	S2B	<0.0001	S3A	S4A	<0.0001
	S4A	<0.0001		S4B	0.0317
S2A	S2B	<0.0001	S3B	S4A	<0.0001
	S4A	<0.0001		S4B	0.0273
	S4B	<0.0001	S4A	S4B	<0.0001

The results revealed that the compost/mulch application, as a BMP, increased moisture retention within the soil surface. The organic mulch essentially reduces or prevents evaporation from the soil surface and alters the soil microclimate. As a result, soil moisture has been conserved mainly within the topsoil layer (Sarkar *et al*., 2007). Bristow and Campell (1986) had similar conclusions noting a 36% reduction in evaporation when the soil surface was covered by organic residue compared to bare soil. Also, the results of our study showed that the use of compost/mulch moderated soil temperature. Similar soil temperature moderating effects of mulch have been reported by Acharya *et al*. (1998) who stated that the presence of mulch increased minimum soil temperature and decreased maximum soil temperature compared with bare soil. This is due to high heat capacity and low thermal conductivities of the mulch materials compared to mineral soil (Cook *et al*., 2006). Dahiya *et al*. (2007) indicated that mulch cover had lower soil heat fluxes during daytime and higher soil heat fluxes during nighttime than control sites. Additionally, Duppong *et al*. (2004) explained that by adding mulch to the soil surface, daily average and daily maximum temperatures may be impacted due to the darker color of the mulch, potentially retaining more heat.

5.3.2 Soil texture

Table 5.3 shows the average sand, silt, and clay percentages of the soil samples that were collected from each plot as well as the area in between the two plots at each site, up and down slope, in 2010 and 2012. Figure 5.5 displays the average sand, silt, and clay percentages for up and down slope within each plot in 2010 and 2012. The results revealed that there were no observed changes in the textural classes within the experiment period and loamy textures were associated with all soil samples. The results also showed that site 1 had the highest clay percent of 37 % in 2010 and 36 % in 2012. Sites 2 and 3 had the highest silt percent of 61 % and 49 %, respectively in 2010. In 2012, the silt percent was 45 % and 55 % for sites 2 and 3, respectively. However, the highest sand percent was found at site 4 with values of 67 % and 69 % sand in 2010 and 2012, respectively (Table 5.3, Figure 5.5). As expected, the area in between the two plots at each site (with native vegetation and no treatments) had almost no change. In most cases, the clay percentages were higher in the up slope position than lower slopes within each plot.

As the silt and clay are highly correlated (Iwashita *et al*., 2012; Zhang *et al*., 2012), they were chosen to indicate the change in particle size distribution within the experiment period. The particle size laboratory results were imported as an attribute table to ArcMap 10 (ESRI, 2011) to produce the spatial distribution maps for the silt and clay fraction within each plot, using IDW interpolation technique. Figures 5.6 and 5.7 present the spatial distribution of silt and clay fractions for each plot, respectively.

As the studied plots were established on highly distributed roadside soil, not naturally developed soil, there was a relatively high variation in the distribution of particle size within the small plot size (4 m X 4 m). This variation could be observed mainly in the spatial distribution maps for the silt fraction for plots S1B in 2010 and S2B both years, since there are relatively large differences between minimum and maximum values within the same plot.

Table 5.3. Soil texture in 2010 and 2012 at each plot, up and down slope, Louisiana, USA.

Plot	Slope	2010				2012			
		Sand	Clay	Silt	Texture class	Sand	Clay	Silt	Texture class
		--------------(%)--------------				--------------(%)--------------			
S1A	Up	50.66	30.65	18.69	Sandy Clay Loam	34.40	33.34	32.26	Clay Loam
	Down	62.46	23.48	14.07	Sandy Clay Loam	36.91	33.22	29.87	Clay Loam
S1B	Up	32.97	32.30	34.73	Clay Loam	31.36	35.57	33.06	Clay Loam
	Down	58.76	20.60	20.64	Sandy Clay Loam	60.43	16.48	23.09	Sandy Loam
Between [a]	Up	33.52	36.82	29.66	Clay Loam	29.99	34.24	35.77	Clay Loam
	Down	67.44	21.45	11.10	Sandy Clay Loam	53.76	20.08	26.16	Sandy Clay Loam
S2A	Up	25.53	30.05	44.42	Clay Loam	27.02	28.05	44.93	Clay Loam
	Down	31.91	24.25	43.84	Loam	34.13	21.14	44.72	Loam
S2B	Up	34.57	31.04	34.39	Clay Loam	36.74	28.28	34.98	Clay Loam
	Down	17.44	21.53	61.03	Silt Loam	41.94	22.32	35.74	Loam
Between	Up	29.42	29.79	40.79	Clay Loam	30.25	31.27	38.49	Clay Loam
	Down	45.36	25.80	28.84	Loam	49.66	19.90	30.44	Loam
S3A	Up	33.63	21.12	45.25	Loam	17.19	28.02	54.78	Silt Clay Loam
	Down	39.70	21.72	38.58	Loam	38.56	20.63	40.80	Loam
S3B	Up	30.64	26.27	43.08	Loam	31.97	26.51	41.52	Loam
	Down	20.03	33.87	46.11	Clay Loam	24.04	32.94	43.02	Clay Loam
Between	Up	28.13	24.79	47.07	Clay Loam	27.71	31.53	40.77	Clay Loam
	Down	23.93	27.06	49.01	Clay Loam	29.52	29.13	41.36	Clay Loam
S4A	Up	58.72	19.00	22.28	Sandy Loam	56.34	12.52	31.14	Sandy Loam
	Down	58.76	13.59	27.66	Sandy Loam	62.80	9.17	28.03	Sandy Loam
S4B	Up	67.05	16.07	16.89	Sandy Loam	61.54	18.66	19.80	Sandy Loam
	Down	61.41	16.70	21.90	Sandy Loam	61.77	14.99	23.24	Sandy Loam
Between	Up	64.75	19.43	15.82	Sandy Loam	63.48	15.91	20.61	Sandy Loam
	Down	62.27	12.10	25.63	Sandy Loam	69.23	10.63	20.14	Sandy Loam

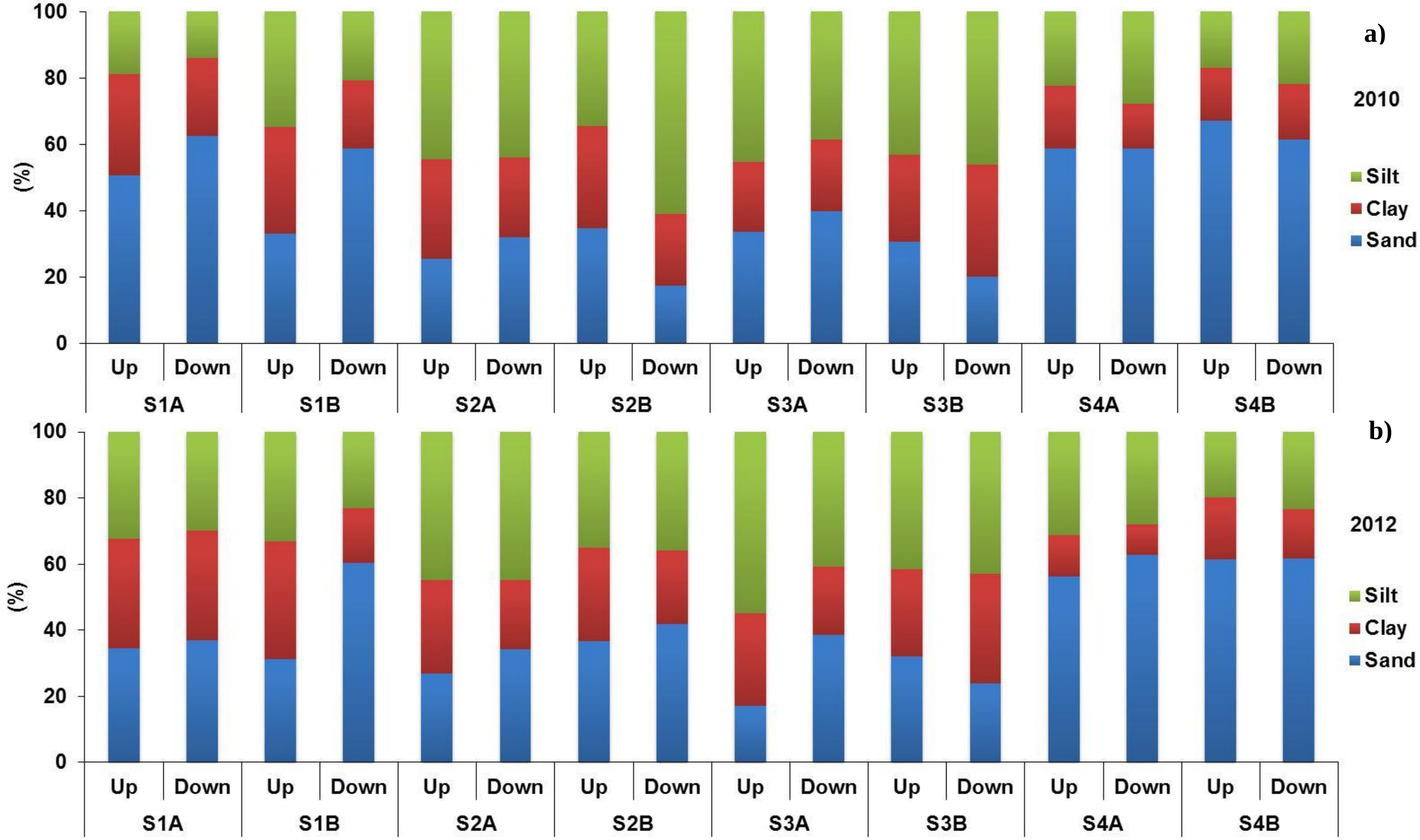

Figure 5.5. The average percentages of sand, silt, and clay in each plot up and down slope in: a) 2010 and b) 2012 in compost/mulch amended roadside soils in Louisiana, USA.

For plot S1A, the results showed that the spatial distribution patterns for silt and clay fractions were the same for 2010. Thus, the areas that had high silt content also had high clay content. In 2012, two years after amending the soil with 10 cm compost/mulch and no tillage, a homogenous distribution with vertical patterns was observed for both silt and clay fractions. The silt fraction decreased from left to right, while the clay fraction decreased from right to left within the same plot. For plot S1B, the same distribution patterns for silt and clay was observed, as high silt and clay percentages decreased down slope for both 2010 and 2012. Application of 5 cm compost/mulch altered the distribution of the silt fraction and conserved the distribution of the clay fraction for 2012, even with light tillage.

Although a heterogeneous distribution pattern was observed for the silt fraction of plot S2A in 2010, the variation between the minimum and maximum percentage was relatively low. In 2012, a thick layer of compost/mulch (10 cm) with no tillage produced more homogeneity for silt distribution. Almost no change was observed for the distribution of clay between 2010 and 2012 for plot S2A. Control plot S2B had the highest silt percent of all plots in 2010 with increased silt content downslope. As topsoil was not covered with compost/mulch (control), heterogeneity was produced for the silt fraction in plot S2B in 2012 due to erosion. The change in the clay fraction for S2B was relatively small; a small increase in clay was noticed downslope in 2012 compared to 2010.

At site 3, both plots were treaded with 5 cm compost/mulch. In plot S3A, the addition of 5 cm compost/mulch with no tillage improved the silt distribution pattern for 2012 samples compared to 2010. Little effect of clay distribution was observed in S3A. When the 5 cm compost/mulch was incorporated into the topsoil by light tillage in plot S3B, a different silt distribution pattern was produced in 2012 compared to 2010 as a result of tillage. Almost the same clay distribution patterns were noted with less clay downslope in 2012.

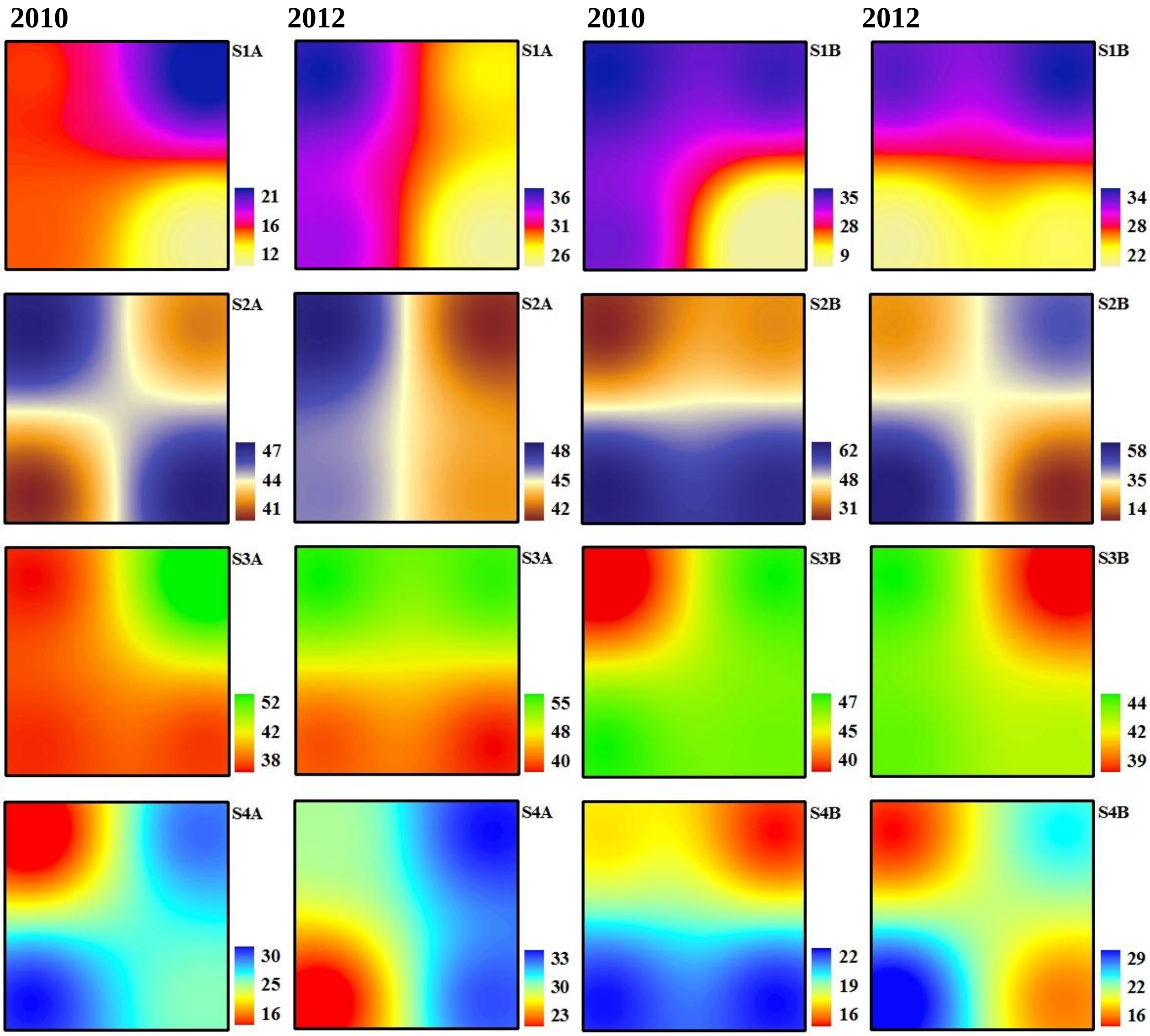

Figure 5.6. The spatial distribution of the silt fraction (%) for each plot in 2010 and 2012 for the four sites, Louisiana, USA. The legend displays maximum, mean, and minimum for each plot.

The second control plot, S4A, showed relatively high variation in silt distribution for both 2010 and 2012. In 2012, there was an increase in silt downslope on one side of the plot and a decrease on the other side compared to the silt distribution in the same plot in 2010. The clay distribution for plot S4A remained virtually constant through the experimental period. Plot S4B was treated with 10 cm compost/mulch which was incorporated into the soil surface by light tillage. Due to tillage practices, a nonhomogeneous distribution pattern could be observed in the

silt fraction in 2010 as well as 2012. The same discussion could be extended to interpret the change in clay distribution for plot S4B in 2012 compared to 2010.

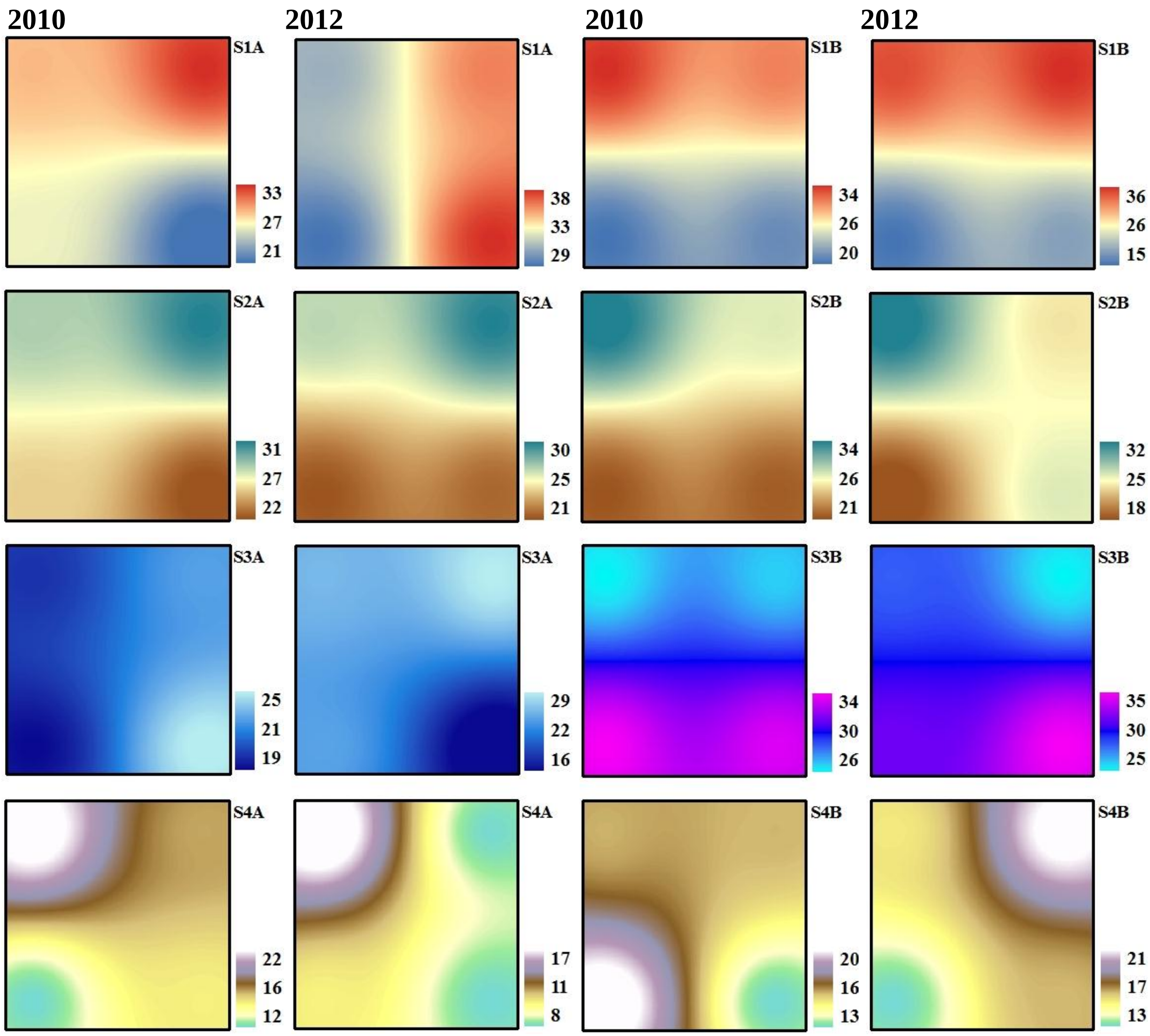

Figure 5.7. The spatial distribution of the clay fraction (%) for each plot in 2010 and 2012 for the four sites, Louisiana, USA. The legend displays maximum, mean, and minimum for each plot.

Based on the aforementioned results of soil texture, sites 2 and 3 are more exposed to erosion hazard since soils with high silt and low clay are known to be most susceptible to erosion (Wischmeier and Mannering, 1969). As site 4 had the highest sand content, conservation management is essential for increasing the available water capacity at this site. Cook *et al*. (2006) noted that low available water capacities for sandy or clay-rich topsoil textures make

water conservation essential for adequate water availability to plants. Additionally, soil texture results within our plots supported the relationship between silt content and site position on slopes elucidated by Iwashita *et al*. (2012), whereby silt content is linked to topographic position on sloped landscapes.

5.3.3 Chemical analyses results

Per Soil Survey Staff (2004), 18 chemical soil elements were extracted using Mehlich-3 extraction (Mehlich, 1984) and measured by ICP-AES. Those elements include Aluminum (Al), Arsenic (As), Barium (Ba), Calcium (Ca), Cadmium (Cd), Cobalt (Co), Chromium (Cr), Copper (Cu), Iron (Fe), Potassium (K), Magnesium (Mg), Manganese (Mn), Sodium (Na), Nickel (Ni), Phosphorus (P), Lead (Pb), Selenium (Se), and Zinc (Zn). Cd concentration was very low and virtually nondetectable via ICP, so this element was omitted from the results presented. Shacklette and Boerngen (1984) presented the average and range of 50 chemical elements in surface soil and other regolith samples through the entire U.S. Results from our samples are presented along with averages of Shacklette and Boerngen (1984) in Table 5.4.

The compost/mulch used to amend the soils was subjected to laboratory characterization to determine its potential effect on the soil surface and soil chemical properties in 2012 soil samples. The laboratory analysis results were presented by Bakr *et al*. (2012) and indicated that the compost/mulch used for this study was safe. Trace element and heavy metal contents were well below maximum limits that addressed by USEPA (EPA, 2012). The results showed that the concentrations of 18 elements were less than the average level stated by Shacklette and Boerngen (1984) (Table 5.4). Specifically, Al concentrations in control plots S2B and S4A decreased considerably in 2012 compared to the initial concentration in 2010 (around 200 $\mu g\ g^{-1}$ differences between the average value). However, with the application of 10 cm compost/mulch, the Al concentrations for the two sets of soil samples were close.

Table 5.4. Average and (standard deviation in parentheses) of; element concentrations, pH, and soil organic matter (%) for compost/mulch amended roadside soils in Louisiana, USA.

Plot		S1A		S1B		S2A		S2B		Average [a]
Year		2010	2012	2010	2012	2010	2012	2010	2012	
Al	µg g^{-1} (SD**)	579 (70)	635 (190)	494 (78)	547 (180)	1093 (142)	794 (254)	1001 (78)	796 (183)	72,000
As		4.5 (1.5)	0.5 (0.2)	3.4 (0.4)	0.5 (0.2)	7.2 (0.9)	0.8 (0.3)	6.8 (0.5)	0.9 (0.2)	7.2
Ba		27.7 (6.5)	32.7 (16)	21.6 (9.7)	28.9 (16.7)	47.3 (7.1)	43.5 (14.2)	45.2 (10.6)	46.4 (9.9)	580
Ca		722 (58)	1004 (162)	628 (225)	831 (317)	463 (60)	973 (308)	438 (22)	455 (30)	24,000
Co		0.28 (0.06)	0.20 (0.17)	0.18 (0.03)	0.10 (0.06)	1.35 (0.26)	0.84 (0.23)	1.08 (0.11)	0.96 (0.22)	9.1
Cr		0.91 (0.26)	0.29 (0.06)	0.74 (0.28)	0.26 (0.13)	1.32 (0.20)	0.42 (0.10)	1.33 (0.12)	0.29 (0.07)	54
Cu		0.43 (0.12)	0.78 (0.17)	0.39 (0.07)	0.77 (0.16)	0.74 (0.13)	1.28 (0.29)	0.55 (0.05)	0.68 (0.13)	25
Fe		47.4 (5.8)	53.7 (9.4)	36.4 (4.8)	34 (7.8)	167 (26)	222 (50.6)	168 (34)	108.8 (36)	26,000
K		18 (6.6)	60.5 (21)	9.7 (6.3)	47.7 (28.2)	37.8 (4.9)	88.8 (26.3)	14.9 (11)	38.1 (7.9)	15,000
Mg		265 (28)	253 (62)	252 (93)	258 (110)	431 (48)	338 (113)	339 (185)	405 (199)	9,000
Mn		3.7 (1.9)	3.8 (1.8)	0.77 (0.18)	1.6 (1.8)	55 (19.1)	41.8 (12)	36.1 (16)	23.2 (16)	550
Na		ND*	152 (26.2)	6.9 (3.4)	160 (39)	6.7 (5.6)	150 (58.7)	81.2 (37.9)	367 (201)	12,000
Ni		0.98 (0.22)	0.01 (0.03)	0.49 (0.04)	0.01 (0.02)	3.8 (0.9)	0.52 (0.17)	2.9 (0.6)	0.72 (0.25)	19
P		2.1 (0.73)	4.2 (0.65)	0.97 (0.34)	3.8 (0.63)	2.4 (0.96)	5.6 (1.2)	2 (0.53)	3.4 (1.4)	430
Pb		3.6 (0.37)	2.6 (0.45)	3.4 (0.58)	2.4 (0.59)	6.4 (0.41)	3.9 (1.1)	6.4 (0.52)	3.8 (0.99)	19
Se		0.88 (0.16)	0.64 (0.36)	0.89 (0.13)	0.51 (0.3)	1.8 (0.24)	1.2 (0.43)	1.7 (0.14)	1.2 (0.37)	0.39
Zn		ND	0.83 (0.34)	ND	0.87 (0.43)	0.29 (0.21)	1.1 (0.49)	0.05 (0.06)	0.4 (0.09)	60
pH		4.83 (0.16)	5.13 (0.2)	5.29 (0.18)	5.43 (0.22)	4.92 (0.1)	5.31 (0.33)	5.03 (0.16)	5.03 (0.26)	---
SOM[b]	(%)	1.33 (0.22)	1.84 (0.43)	0.95 (0.33)	1.69 (0.79)	2.44 (0.22)	2.87 (0.29)	2.01 (0.36)	2.25 (0.09)	---

Table 5.4. (Cont.)

Plot		S3A		S3B		S4A		S4B		Average [a]
Year		2010	2012	2010	2012	2010	2012	2010	2012	
Al	µg g^{-1} (SD**)	993 (105)	1061 (316)	915 (125)	925 (247)	554 (132)	340 (131)	499 (91)	464 (98)	72,000
As		6.9 (0.6)	1.2 (0.3)	6.2 (0.9)	1.0 (0.2)	3.9 (0.8)	0.4 (0.1)	3.4 (0.5)	0.6 (0.1)	7.2
Ba		45.3 (7.1)	61.9 (17.1)	82.7 (24)	78.6 (29.7)	38.2 (8.6)	33.1 (13.6)	43.2 (9)	39.6 (6.8)	580
Ca		325 (195)	777 (460)	1149 (472)	1317 (607)	287 (80)	293 (119)	312 (107)	754 (179)	24,000
Co		2.10 (0.75)	2.12 (1.36)	4.27 (1.71)	3.22 (1.47)	0.98 (0.17)	0.78 (0.32)	0.78 (0.32)	0.59 (0.12)	9.1
Cr		1.09 (0.09)	0.23 (0.06)	1.27 (0.10)	0.29 (0.13)	0.48 (0.09)	0.16 (0.03)	0.46 (0.11)	0.36 (0.08)	54
Cu		0.86 (0.72)	0.69 (0.24)	0.66 (0.12)	0.89 (0.36)	0.38 (0.06)	0.49 (0.11)	0.41 (0.09)	0.97 (0.22)	25
Fe		80.8 (11.9)	80.5 (35)	101.6 (18)	103 (43)	79.1 (8.7)	61.9 (12.5)	68.5 (10.4)	71.5 (44.5)	26,000
K		39.9 (28.7)	69.8 (16.1)	65 (10.7)	107 (27.8)	13.8 (5.5)	26.8 (14.4)	24 (2.3)	64.6 (23.2)	15,000
Mg		335 (41)	369 (104)	553 (193)	476 (180)	373 (97)	282 (135)	415 (88)	401 (97)	9,000
Mn		64.8 (23.6)	52.8 (56.3)	117 (45.4)	107 (57.8)	11 (4)	8.4 (2.5)	7.8 (4)	7.1 (2)	550
Na		100 (23)	234 (66.8)	211 (101)	233 (26.8)	98.1 (51.7)	210 (91.1)	119 (59.3)	205 (45.1)	12,000
Ni		4.5 (1.2)	0.81 (0.61)	7.7 (2)	1.5 (0.75)	1.2 (0.3)	0.28 (0.04)	1.3 (0.5)	0.45 (0.13)	19
P		1.4 (1.0)	3.5 (1.2)	1.8 (0.44)	4.3 (1.5)	0.79 (0.36)	2.4 (0.56)	1.3 (0.84)	4.5 (0.97)	430
Pb		5.8 (0.18)	4.7 (0.56)	5.8 (0.26)	4.4 (0.31)	3.2 (0.49)	1.8 (0.47)	3.2 (0.44)	2.3 (0.28)	19
Se		1.8 (0.1)	1.7 (0.37)	1.8 (0.17)	1.5 (0.34)	1.1 (0.21)	0.5 (0.17)	0.95 (0.16)	0.76 (0.12)	0.39
Zn		0.81 (1.3)	0.45 (0.19)	0.2 (0.14)	0.47 (0.22)	0.12 (0.1)	0.24 (0.19)	0.13 (0.15)	0.3 (0.16)	60
pH		5.38 (0.11)	5.45 (0.3)	5.51 (0.14)	5.77 (0.32)	5.98 (0.19)	6.68 (0.16)	5.87 (0.30)	6.3 (0.33)	---
SOM[b]	(%)	1.81 (0.5)	2.62 (0.39)	2.23 (0.23)	2.84 (0.4)	1.26 (0.34)	1.2 (0.5)	1.21 (0.2)	1.64 (0.21)	---

* Not Detected, ** standard deviation, [a] element average from Shacklette and Boerngen (1984), [b] Soil organic matter
Note: Two decimal numbers accuracy were used, because of space limitation one and zero decimals were used.

The results also indicated that Ca remained fairly constant in the control plots; but it doubled in the 10 cm compost/mulch treated plots in 2012 soil samples. The K concentration increased in 2012 regardless of the compost/mulch applications or thickness. Fe concentrations showed no change between the two years of compost/mulch application. For Mg, little change was observed due to the compost/mulch applications. However, Na concentrations increased substantially due to the compost/mulch treatment, as the concentration nearly doubled in 2012 compared to 2010. The P concentrations also increased in 2012, although they were generally low in the topsoils. The remaining chemical elements evaluated were mainly heavy metals and were found in low concentrations for both years.

The pH results indicated that the soils of sites 1, 2, and 3 were strongly acidic, while site 4 soils were moderately acidic (Soil Survey Division Staff, 1993). The SOM content increased with the application of compost/mulch in 2012 samples. Based on the soil texture of the studied plots, the SOM content was moderate for almost all sites. Site 4 had coarser soil texture (higher sand) and, as a result, with even lower SOM content it was considered a moderate content based on the soil texture recommendations of Baldock and Skjemstad (1999).

5.4 CONCLUSIONS

The current study confirmed the effectiveness of compost/mulch in improving soil water content and moderating surface soil temperature compared to bare soils (control plots). Soil pH did not significantly change due to compost/mulch application. However, as expected, SOM increased as a result of adding organic materials on the soil surface. The results of this study underscore the importance of understanding the linkages between inherent soil properties, rainfall, and soil erosion in order to improve land use planning and identify better sustainable management.

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CHAPTER 6. EVALUATION OF COMPOST/MULCH AS HIGHWAY EMBANKMENT EROSION CONTROL IN LOUISIANA AT THE PLOT-SCALE[4]

6.1 INTRODUCTION

Soil erosion is an environmental concern due to loss of nutrient-rich topsoil resulting in increased sedimentation, turbidity, and levels of pollutants in adjacent water bodies (Bhattarai *et al.*, 2011; Ebisemiju, 1990; Girmay *et al.*, 2009; Hopmans *et al.*, 1987; Pieri *et al.*, 2007). Roads affect both the biotic and the abiotic components of landscapes by changing the dynamics of populations of plants and animals, altering flows of materials in the landscape, introducing exotic elements, and changing levels of available resources, such as water, light and nutrients (Coffin, 2007). As cities grow, new highways are developed; consequently, the streams within highway paths are susceptible to impacts from construction activities (Berndtsson, 2010; Chen *et al.*, 2009). These activities could increase topsoil removal, destroy native vegetation, and result in severe surface runoff and water erosion (Xu, 2006). Although construction is not a source of water pollution, the sedimentation processes due to soil disturbance during construction activities are considered a major nonpoint source (NPS) of pollution (Houser and Pruess, 2009; Lane and Sheridan, 2002). Keller and Sherar (2003) pointed out that roads are to blame for approximately half of the erosion from logging operations, and most erosion occurs during the first rainy season after disturbance. Ziegler *et al.* (2001) showed that for linearly connected systems of roads, large volumes of overland flow may transfer to a stream network and lead to hydraulic erosion processes, causing stream sedimentation even during low rainfall events. Based on sediment loading rates throughout the U.S., erosion from various construction sites can raise as much as 500 fold when compared to undisturbed natural areas (US Environmental Protection Agency, USEPA, 2005). Forman (2000) reported that around 19% of the total land area has been affected

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by the public roads system. Road construction impacts on soil are significantly increased in humid areas, where high rainfall exacerbates soil erosion risk and amplifies this risk on steep hill slopes in such areas.

As erosion is a natural process it cannot be completely eliminated. However, best management practices (BMPs) can be used to control and manage sediment loading (USEPA, 2005). Along highways, numerous BMPs have been used to impound runoff and control soil erosion such as: vegetated buffers and mulches, porous pavement materials, retention or detention basins and ponds, silt fence, hydroseeding, and the placement of natural fiber mats (Han et al., 2005; Hogan and Walbridge, 2007; Houser and Pruess, 2009; Li *et al.,* 2006). However, the effectiveness of some implemented BMPs on water quality protection is still unclear (Easton *et al.,* 2008). Keller and Sherar (2003) indicated that bare soils should be covered with grass or mulch to control erosion processes. Nunes *et al.* (2011) concluded that soil coverage is essential for protecting the soil from erosion by intercepting rainfall, reducing the erosive power (kinetic energy) of raindrops, and decreasing the volume of water reaching the soil surface. So, the intensity of surface erosion is significantly affected by land use/cover (García-Ruiz, 2010; Kosmas *et al.,* 1997), however, the types of land use/cover should also be considered (Nunes *et al.,* 2011). The *Delaware Erosion and Sediment Control Handbook* (2003) stated that wood chip mulch is well suited for roadside right-of-way placement at the rate of around 14.8 tons ha^{-1} or 1.4 kg m^{-2}. Poesen and Lavee (1991) stated that there is a negative relation between runoff volume, sediment concentration in runoff, and mulch cover. In a study of 41 different plot lengths (0.1-30.5m), Smets *et al.* (2008) concluded that plot length is an important factor in determining the effectiveness of a mulch cover in reducing soil erosion. Monitoring soil loss using runoff plots is a suitable, cost-effective, and useful approach that helps decision makers to identify soil erosion risk and improve their management practices (Hartanto *et al.,* 2003). In the

United States, compost and mulch filter berms have been approved in some states as effective alternatives to silt fences for erosion control and storm water protection (Smith, 2002).

Louisiana has plentiful surface waters; including >106,690 km of rivers and streams, 436,264 ha of lakes and reservoirs, 2,246,394 ha of fresh and tidal wetlands, and 1,982,898 ha of estuaries (Louisiana Department of Environmental Quality LDEQ, 2006). Among all these bodies of water, > 35 different suspected causes of impairment[5] were reported in Louisiana surface waters. At least eight of these causes, including total suspended solids (TSS), turbidity, and biochemical oxygen demand (BOD), are related to NPSs of pollution caused by storm water runoff from different areas including construction sites (LDEQ, 2006). Weindorf (2008) noted that total annual precipitation in Louisiana ranges from 119 to 180 cm; substantial precipitation capable of causing severe water erosion. Considerable literature exists related to erosion control in humid environments. However, few studies evaluated the effect of compost/mulch thickness, slopes, and tillage practices in active and established construction areas, on sediment loss rates and runoff water quality from highway right-of-ways. In this study, our main goal was to examine the influence of compost/mulch application in the reduction of TSS and turbidity in runoff water from highway right-of-ways in Louisiana. The specific objectives were to: 1) evaluate storm water runoff rates on plots receiving compost/mulch, and 2) assess the effect of compost/mulch thickness, plot slope, tillage, and construction activities on water quality.

6.2 MATERIAL AND METHODS

6.2.1 Study site and plot design

Two locations on highway right-of-ways were monitored. First, US Highway 61 (~8 km from St. Francisville, LA) in West Feliciana Parish was undergoing active road construction and

[5] A waterbody with chronic or recurring violations of applicable numeric and/or narrative water quality criteria (USEPA, Clean Water Act, 303d).

had one site (site 1). Second, IH-49 (~20 km from Bunkie, LA) in Rapides Parish was prone to erosive undercutting in numerous areas. The latter location had three sites (site 2, 3, and 4). Eight plots were constructed (e.g. two plots side-by-side at each site), all at a fixed size of 4m X 4m and were surrounded by heavy gauge steel edging on all sides to prevent flow from outside the plots. Site 1 had the steepest slope at 34%. Site 2 had a 25% slope in an erosive "blowout" area. Site 3 had a 15% slope in an erosive "backcut" area. Site 4 had a 10% slope in an erosive "blowout" area. The sites were geo-referenced with global positioning system receivers and the data was imported into ArcGIS 9.3 (ESRI, 2008) to produce a location map (Figure 6.1).

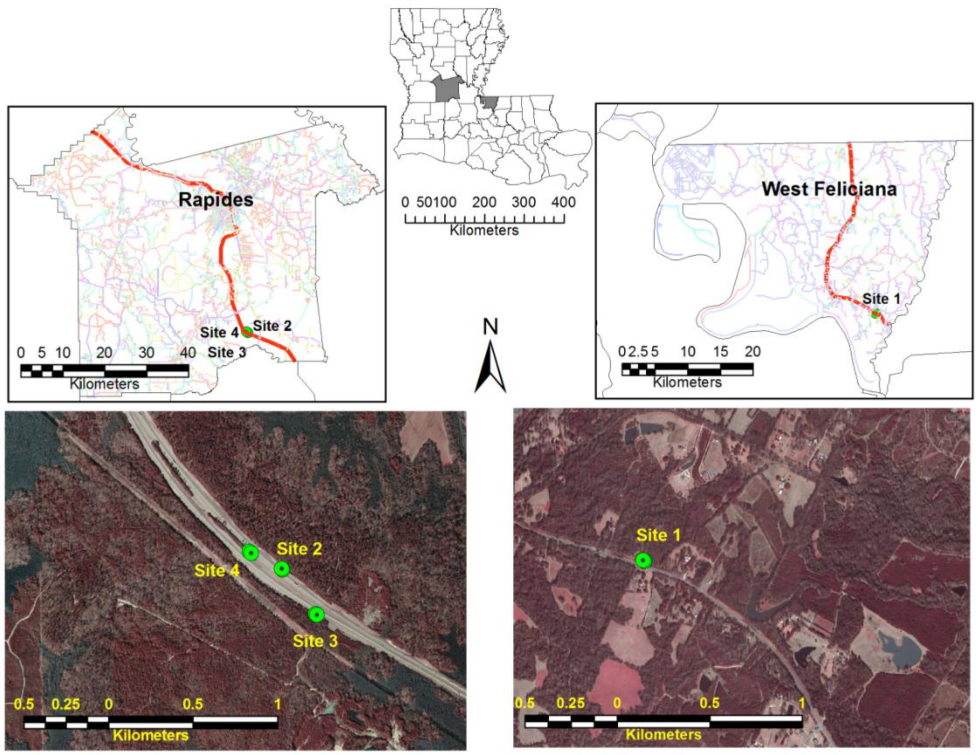

Figure 6.1. The geographic site locations along US Highway 61 and IH-49 highways in West Feliciana and Rapides Parishes, respectively, Louisiana, USA.

In March of 2010, field equipment was installed. Different thicknesses of compost/mulch were applied for each plot as a BMP for erosion control. At each site, one plot was lightly tilled and one remained non-tilled. Compost/mulch was provided by Bob's Tree Preservation (Church Point, LA). Wood chips for the compost/mulch blend were 70% hardwood and 30% pine trees

harvested locally while the compost was a double-ground, screened, recycled wood fiber material, also harvested locally (B. Thibodeaux, personal communication, 2010). *Test Methods for Examination of Composting and Compost* (TMECC, 2001) was used to analyze pH, electrical conductivity (EC), organic matter (OM), particle size, and carbon to nitrogen (C:N) ratio of the applied compost/mulch. Two control plots did not receive compost/mulch applications, since the bare soil is a good indicator of the soil's vulnerability to erosion risk (Nunes *et al.*, 2011). One of the control plots was tilled and one left non-tilled. In all other plots, 5 cm and 10 cm of compost/mulch were applied. The compost/mulch application rates were established per recommendations of Alexander (2002). He reported that with high annual rainfall, recommended application rates for vegetated and non-vegetated compost surface mulch were 2.5 to 5 cm and 5 to 10 cm, respectively. The compost/mulch applications within the plots, locations, slope, tillage practices, and identification for each plot per site are summarized in Table 6.1.

Table 6.1. Compost/mulch applications, locations, slope, tillage practices, and identification for each plot at each site in Louisiana.

Parish	Location	Construction	Site	Plot	Slope (%)	Compost (cm)	Tillage
W. Feliciana	US Highway 61	Active	1	A	34	10	no-tillage
				B		5	light-tillage
Rapides	IH-49	established	2	A	25	10	no-tillage
				B		0	light-tillage
			3	A	15	5	no-tillage
				B		5	light-tillage
			4	A	10	0	no-tillage
				B		10	light-tillage

6.2.2 Sampling

Runoff from storm water events was directed into H-flumes, 0.305 m depth, for sampling and quantification of flow in relation to received precipitation for each plot (Grant and Dawson, 1997). A refrigerated ISCO® Model 6712 (Teledyne ISCO, Lincoln, NE) auto-sampler was used in each plot and programmed for uniform 24-h composite samples with 5-min frequency time intervals. Level data were used to calculate water flow rate using Flowlink® 4.15 software

(ISCO, 2002). Two rainfall gauges were associated with ISCO samplers to record the rainfall in-situ at each chosen location. Water pH, EC, TSS, turbidity, and BOD were studied to determine runoff water quality. Analyses were conducted based on the *American Public Health Association* (APHA, 2005). Soil samples were collected from each site prior to plot instrumentation. Soil pH (1:1 soil to water), EC, cation exchange capacity (CEC, Ammonium Acetate pH 7), OM (LOI), and soil texture (pipette method) were analyzed per the Soil Survey Staff (2004).

6.2.3 Statistical analysis

Factor analysis was performed to illustrate the patterns of covariance between the comparative variables (construction activities, slopes, tillage practices, and compost/mulch applications). Following factor analysis, PROC UNIVARIATE in SAS® 9.2 software (SAS, 2008) was used to determine the general trend of water quality parameters. Also, a two samples *t*-test was generated in SAS to estimate the significant differences of each water quality parameter due to the effect of each variable. As sites 1 and 2 had 34% and 25% slopes, respectively, the Soil Survey Staff (1993) would assign the same slope class to both (steep), providing a basis for their statistical comparison. As such, plots S1A (active construction site) and S2A (established site) were statistically analyzed to determine the effect of construction activities on runoff water quality. Additionally, sites 3 and 4 had 15% and 10% slopes, respectively, and both sites were within the moderately steep slope class (Soil Survey Staff, 1993). Accordingly, plots S3A and S4A as well as S3B and S4B were compared to assess the effect of compost/mulch thickness on the runoff water quality. Finally, the tillage effect was evaluated using both plots of site 3 since one plot was non-tilled (S3A) and one was lightly tilled (S3B). Based on the aforementioned pairs, three null hypotheses (H_0) were suggested: there was no difference in the water quality parameters due to (1) the construction activities, (2) the compost/mulch thickness, and (3) the tillage practices.

6.3 RESULTS AND DISCUSSION

6.3.1 Compost/mulch characterization

The effectiveness of compost/mulch used for erosion control applications depends upon its characteristics. Generally, coarser compost/mulch texture applied at relatively high application rates is required for soils prone to erosion (Alexander, 2002). Compost/mulch laboratory analysis results from the current study were compared with different approved specifications (Alexander, 2002; Storey *et al.*, 1996; USCC, 2001; USEPA, 2011) previously used (Table 6.2).

Table 6.2. Compost/mulch characterizations compared to different compost specifications and the maximum acceptable heavy metal values for compost used for erosion control.

Analysis	Current study		USCC (2001)	Alexander (2002)	USEPA (2011)	Storey *et al.* (1996)
pH	6.41±0.25		5.5-8.0	N/A	5.5-8.6	5.5-8.5
EC (mmhos/cm)	0.30±0.04		Varied	< 5	< 10	< 10
Moisture Content (%)	9.68±6.55		Varied	30-60	30-60	N/A
Organic Matter (%)	85.01±2.32		N/A	25-100	30-65	> 60
Particle Size (%)						
Passing 16 mm	98.08±1.75		N/A	N/A	98	98
Passing 9.5 mm	86.07±6.35		Varied	N/A	N/A	70
CN ratio	139.41±29.22		N/A	N/A	N/A	N/A
Trace elements/ Heavy metals	Dissolved	Total	------------------------μg g^{-1}------------------------			
As (mg L^{-1})	ND	59.44±3.79	75	75	75	10
Cd (mg L^{-1})	ND	0.51±0.07	85	85	85	16
Cr (mg L^{-1})	ND	11.74±0.83	3000	3000	3000	180
Cu (mg L^{-1})	0.20±0.01	23.75±1.81	4300	4300	4300	1020
Ni (mg L^{-1})	0.66±0.01	38.66±0.64	420	420	420	160
Pb (mg L^{-1})	0.81±0.03	39.61±1.92	840	840	840	300
Zn (mg L^{-1})	0.14±0.01	38.20±4.96	7500	7500	7500	2190

The pH value was generally slightly acidic (the mean 6.4±0.25) with an average EC of 300±40 μS/cm. Particle size analysis for the compost/mulch showed that 98% ± 2% of the sample passed through a 16 mm sieve. Organic matter content for the compost/mulch was 85% ± 2%. The C:N ratio results showed that the compost/mulch was generally N poor, facilitating N immobilization at a mean C:N ratio of 139:1. Trace element and heavy metal contents in the

compost/mulch were well below maximum limits for safe use even when extracted via total digestion (HNO_3/HCL).

6.3.2 Soil characterization

The physiochemical soil properties for each site are given in Table 6.3. Soil pH (4.98-5.90) ranged from very strongly acidic to moderately acidic (Soil Survey Staff, 1993) at all sites. The CEC values were related to clay content. Clay content ranged from 21% to 25% and CEC values ranged from 9 to 11 cmol kg^{-1}; for sites 1, 2, and 3. At site 4, as the clay content was 8%, the CEC value decreased to 5 cmol kg^{-1}. The OM contents were lower at sites 1 and 4 (1%) compared with sites 2 and 3 (2%). Site 2 was covered by grass and site 3 was partially protected by trees, thus the organic materials were already being naturally introduced into the soil. Site 3 had the highest silt percentage (51%), which decreased to 44% for site 2. Sites 1 and 4 had the highest sand percentage (57%). Results also revealed that sites 2 and 3 had loam and silt loam soil textures with silt percentages of 44% and 51%, respectively, which directly affected runoff and soil erosion. Soils with high silt, low clay, and low organic matter, are known to be most susceptible to erosion (Wischmeier and Mannering, 1969). Sites 1 and 4 had sandy loam soil texture, both with 57 % sand. The C:N ratio of the soil was 7:1 with low C and N content.

Table 6.3. Select physiochemical soil properties for each study site along US Highway 61and IH-49 highway right-of-ways, Louisiana.

Site	pH (1:1) Soil:Water	C:N ratio[a]	CEC[b] cmol kg^{-1} soil	OM[c]	Sand	Silt	Clay
				----------------------------	---------------------------%---	-------------------------	-------------------
1	5.1±0.3	2.9±0.9	10.9±5.1	1.2±0.3	57.2±15.9	20.5±9	22.3±7.8
2	5.0±0.1	10.5±3.9	9.2±0.8	2.1±0.5	31.0±6.1	44.1±4.1	24.9±3.0
3	5.5±0.1	7.1±2.6	11.4±2.0	2.1±0.4	27.6±3.8	51.3±4.1	21.1±4.2
4	5.9±0.3	7.0±2.8	5.5±1.2	1.2±0.3	56.5±4.6	35.4±5.2	8.1±2.7

[a] carbon:nitrogen ratio, [b] cation exchange capacity, [c] organic matter

6.3.3 Rainfall/flow rate

In-situ rainfall data was recorded by ISCO® auto-samplers then downloaded and analyzed by Flowlink 4.15 software. Monthly rainfall (mm) as recorded in-situ for each location was compared with archived Louisiana climate data (Louisiana Office of State Climatology, 2012 and Louisiana Agriclimatic Information System, 2012), and are presented in Table 6.4.

Table 6.4. Monthly rainfall (mm) at the two locations compared to the closest metrological stations to the study areas.

Month	BRT[a]	US Highway 61	Dean Lee[b]	IH-49
Jun-10	46.2	10.4	137.7	416.1
Jul-10	29.2	NR*	121.4	114.4
Aug-10	168.7	NR	124.7	142.5
Sep-10	44.9	NR	28.9	22.8
Oct-10	31.2	29.2	42.9	106.1
Nov-10	42.9	147.9	169.9	357.6
Dec-10	96.0	NR	41.4	132.5
Jan-11	42.7	NR	93.5	228.9
Feb-11	18.5	NR	45.7	85.2
Mar-11	95.3	28.4	139.2	179.9
Apr-11	18.5	35.1	39.4	82.8
May-11	7.4	24.5	99.6	89.3
Jun-11	89.7	62.2	11.7	228.6
Jul-11	49.8	188.8	NR	119.9
Aug-11	40.6	6.0	124.7	14.8
Total	821.7	532.5	1220.7	2321.40

[a]East Central Region "Baton Rouge" (Louisiana Office of State Climatology, 2012), [b]Alexandria (Dean Lee R/S) (Louisiana Agriclimatic Information System, 2012), * Not recorded.

The archived precipitation values were different than the in situ recorded values since they were recorded at meteorological stations several kilometers away from the experiment locations. The maximum daily average runoff flow rate ($m^3\ d^{-1}$) for each month of the study was calibrated per Grant and Dawson (1997). The maximum daily average flow rate results (Table 6.5) were compatible with rainfall (mm) and influenced by the compost/mulch applications and tillage practices. During the winter season (December, January, and February); ISCO samplers could not record level data since the sensors were stopped with low temperatures. Results show

that a tilled control plot (S2B) had the highest maximum flow rate (12 m^3 d^{-1}) among all plots. A second control plot, non-tilled and on a low slope, also had higher maximum flow (6 m^3 d^{-1}) compared with all other plots. The tilled plot with 5 cm compost/mulch application (S3B) had higher maximum flow (5.6 m^3 d^{-1}) compared with other tilled or untilled treatment plots. Plots treated with 10 cm compost/mulch had the lowest flow rates.

Table 6.5. Maximum daily average flow rate per month for each studied plot in Louisiana.

Month	S1A	S1B	S2A	S2B	S3A	S3B	S4A	S4B
	--------------------------------			m^3 d^{-1}				--------------------------------
Jun-10	ND*	0.004	0.690	11.000	1.758	5.222	4.927	0.119
Jul-10	ND	ND	ND	4.371	1.489	5.609	5.295	0.120
Aug-10	ND	ND	ND	4.458	2.965	3.801	6.039	2.264
Sep-10	ND	0.063	ND	0.375	2.342	4.228	0.338	0.601
Oct-10	0.065	0.136	1.859	0.824	1.764	4.406	1.775	0.962
Nov-10	0.165	0.193	0.998	7.405	1.043	2.887	2.952	3.354
Dec-10	NR**	NR	NR	NR	NR	NR	NR	NR
Jan-11	NR	NR	NR	NR	NR	NR	NR	NR
Feb-11	NR	NR	NR	NR	NR	NR	NR	NR
Mar-11	0.654	0.146	0.253	5.177	1.486	0.117	0.676	0.033
Apr-11	0.611	0.056	0.067	7.476	ND	0.059	0.101	0.029
May-11	0.757	0.390	1.977	11.929	0.202	0.498	3.483	2.134
Jun-11	0.196	0.399	2.941	5.176	1.388	0.374	2.342	2.264
Jul-11	0.746	0.449	3.926	8.411	0.961	1.366	0.084	3.749
Aug-11	0.075	0.177	0.110	8.887	1.679	0.055	0.178	0.736
Mean	0.715	0.352	2.973	26.114	5.756	6.712	4.085	2.474
SD[a]	0.641	0.291	3.607	29.728	5.458	8.723	5.339	2.928

*Not detected (sensor malfunction), **Not recorded (low temperature), [a]standard deviation

Monthly total flow (m^3) for each plot and the monthly rainfall (mm) as recorded in-situ for each location, are given in Figure 6.2. Plot S2B produced the highest total flow over all plots with a maximum of 100 m^3, mean of 26 m^3, and standard deviation of 29 m^3 (Figure 6.2b). The 10 cm compost/mulch application in S2A sharply reduced the total flow by around 90% and led to excellent conservation of the soil surface. Such results are consistent with previous works of Bhattarai *et al.* (2011), García-Ruiz (2010), Keller and Sherar (2003), and Nunes *et al.* (2011).

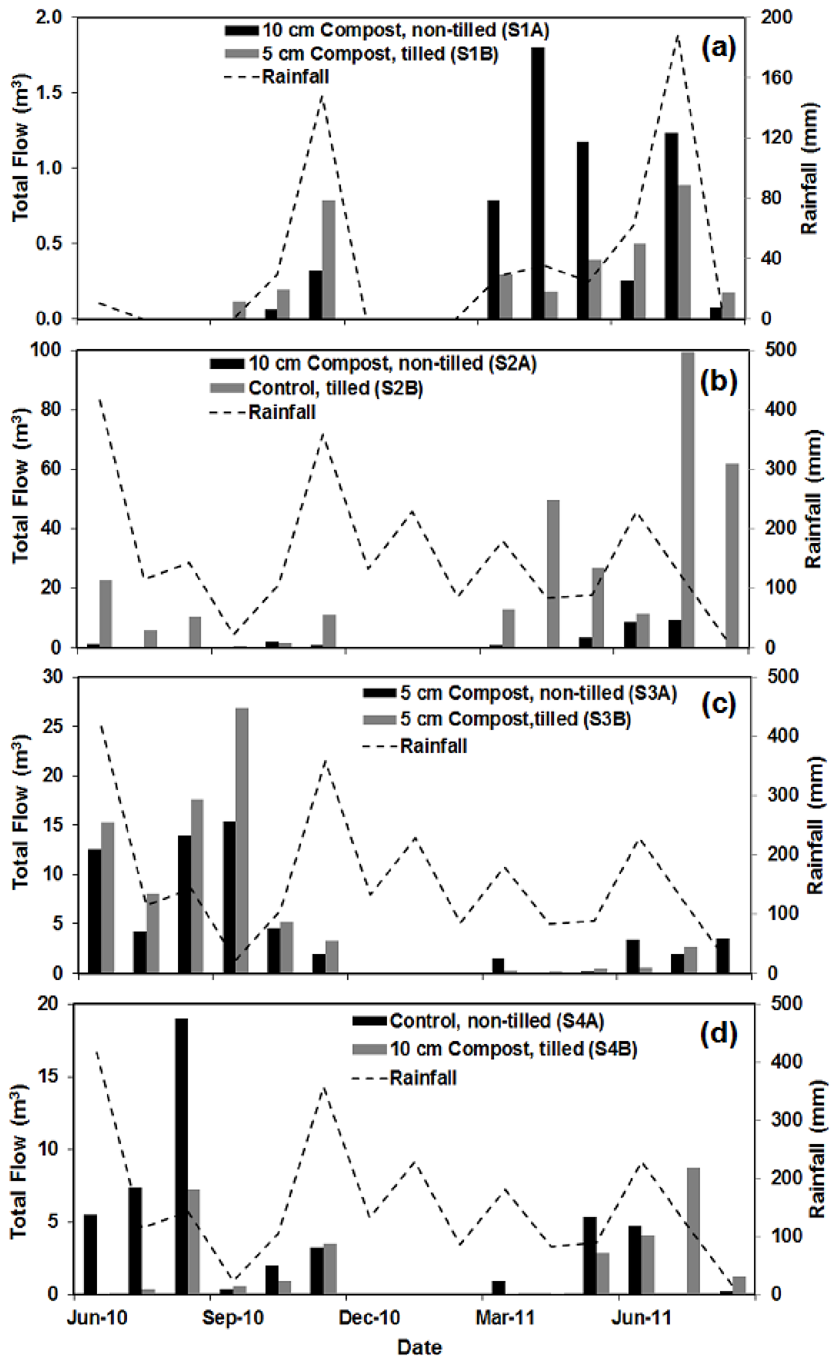

Figure 6.2. Monthly total flow; a) site 1, b) site 2, c) site 3, and d) site 4 in Louisiana, USA.

Plot S4A (control plot), exhibited higher total flow compared with S4B (10 cm compost/mulch treatment with tillage) (Figure 6.2d). Those results confirm the previous observation of Hartanto *et al.* (2003) as they reported that at the plot scale, the presence of organic materials is important for preventing soil detachment and providing surface roughness,

which reduces runoff and soil particle movement. The tillage effects can be observed in plots S3B vs. S3A, since both of them received the same amount of compost/mulch, 5 cm, where S3A was non-tilled and S3B was lightly tilled. As expected, the tillage incorporation increased the total flow for S3B (Figure 6.2c). Generally, the tilled plots tended to have a higher total flow compared with the non-tilled plots mostly for 2010 results (Figure 6.2). However during 2011, construction activities were near the experiment location and could have affected the results.

6.3.4. Statistical analysis

6.3.4.1 Factor analysis

The method employed was an interdependence technique which made no distinction between dependent and independent variables. The variables that were highly correlated, positively or negatively, were likely influenced by the same factors, while the relatively uncorrelated variables were likely influenced by other factors. Thus, highly correlated variables tended to have similar loading patterns, showing a clustering effect (Hatcher, 1994).

As the factor analysis was applied to the water runoff data with all variables included (TSS, Turbidity, BOD, pH, and EC), the results showed that 80% of the variance within the dataset can be explained by the first two factors, with 63.7% and 16.9% contributions from factors 1 and 2, respectively (Figure 6.3). The samples were grouped in terms of slope percentage, tillage practices, compost/mulch thickness, and construction activities in the coordinate system of factors 1 and 2 to examine the respective effects. The results showed that grouping of slopes, tillage, and construction effects were overwhelmingly mixed with each other in the coordinate system. However, the clustering effect of compost/mulch thickness groups was readily identifiable: the samples from control plots (no compost/mulch) had negative concentrated loadings on factor 1 and almost positive-only loadings on factor 2. Samples from the 5 cm compost/mulch treatment had positive concentrated loadings on factor 1 and almost

positive-only loadings on factor 2, while samples from the 10 cm compost/mulch treatment had positive-only loadings on factor 1 and predominantly negative concentrated loadings on factor 2. Comparing the geometric centers of each treatment, longer distances were also found among compost/mulch thickness than among the other treatments, which imply compost/mulch thickness introduced more variations than the others. This implied that the majority of sample variances were mainly introduced by the compost/mulch thickness, and the effects of slope, tillage, and construction activities on water quality were overshadowed by compost/mulch thickness. Thus, the dominant effect of compost/mulch application could be potentially used to overcome the adversity of steep slopes and other factors during highway construction.

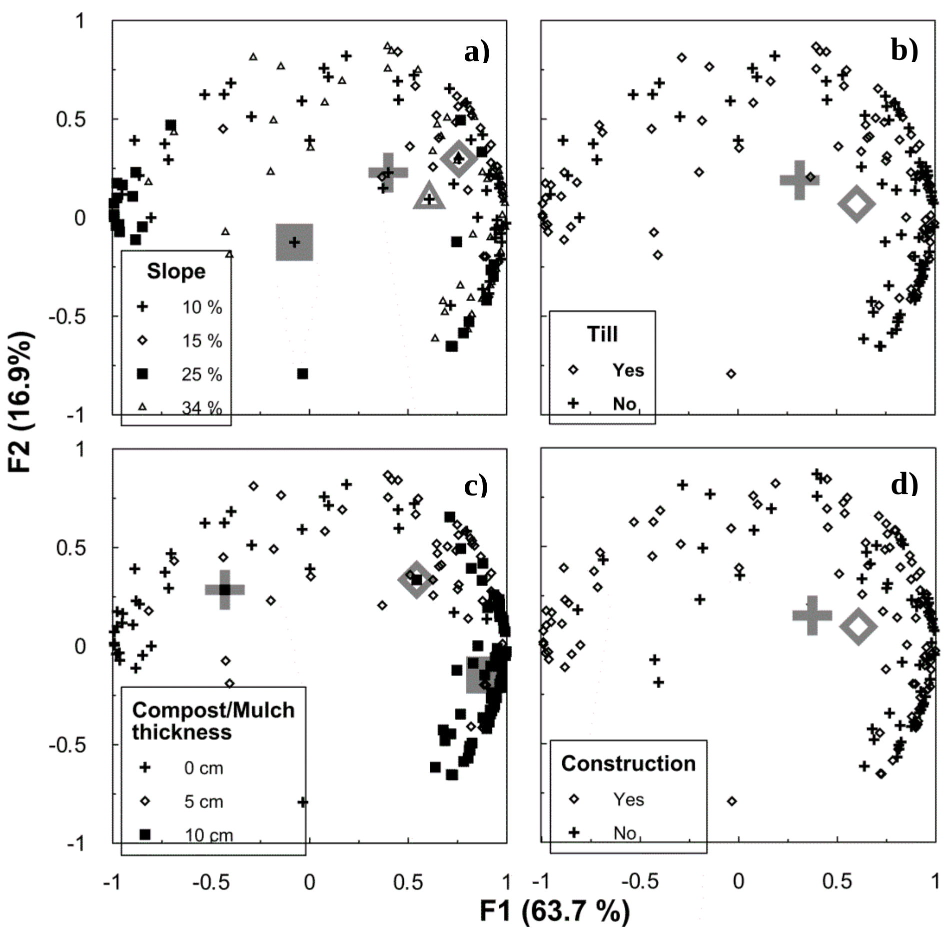

Figure 6.3. Factor analysis results according to the cluster groups of: a) slope percentage, b) tillage practices, c) compost/mulch thickness, and d) construction activities based on all runoff water samples data of all sites, in Louisiana. The gray larger symbols refer to the geometric centers of each group.

Factor analysis conclusively demonstrated that the majority of sample variances were mainly due to the thickness of applied compost/mulch. Following this analysis, plots were separated into matched pairs to examine the statistical significance of each variable independently (construction activities, compost/mulch thickness, and tillage practices).

Based on factor analysis results, the TSS, turbidity, and BOD values were graphically categorized for comparative variables (construction activities, compost/mulch treatments, tillage practices) and presented in Figures 6.4, and 6.5. Generally, TSS trends and turbidity trends (Figure 6.4) were perfectly matched. Results showed that construction activities influenced TSS and turbidity even with 10 cm of compost/mulch treatment. By the end of the experiment, construction activities were completed and likely led to the substantial decrease in TSS and turbidity over the last two months (Figure 6.4a). Figure 6.4b indicated that plot S4A (control) had substantially higher TSS and turbidity values than plot S3A (5 cm compost/mulch), this illustrated the effectiveness of compost/mulch application. As expected, Figure 6.4c showed that 10 cm compost/mulch (S4B) decreased TSS and turbidity compared to 5 cm compost/mulch application (S3B), even with tillage incorporation. Finally, tillage increased the TSS and turbidity values in water runoff (Figure 6.4d) as it disturbs the soil surface, accelerates the flow rate, and increases the suspended particles in runoff.

The BOD was not highly influenced by the determined variables (construction activities, compost/mulch application, and tillage practices). Construction activities and 5 cm compost/mulch application increased the BOD values (Figure 6.5a and 6.5b). In order to quantify any significant differences in the water quality parameters due to the comparative variables (construction activities, compost/mulch applications, and tillage practices), parameters were statistically analyzed under each comparative variable.

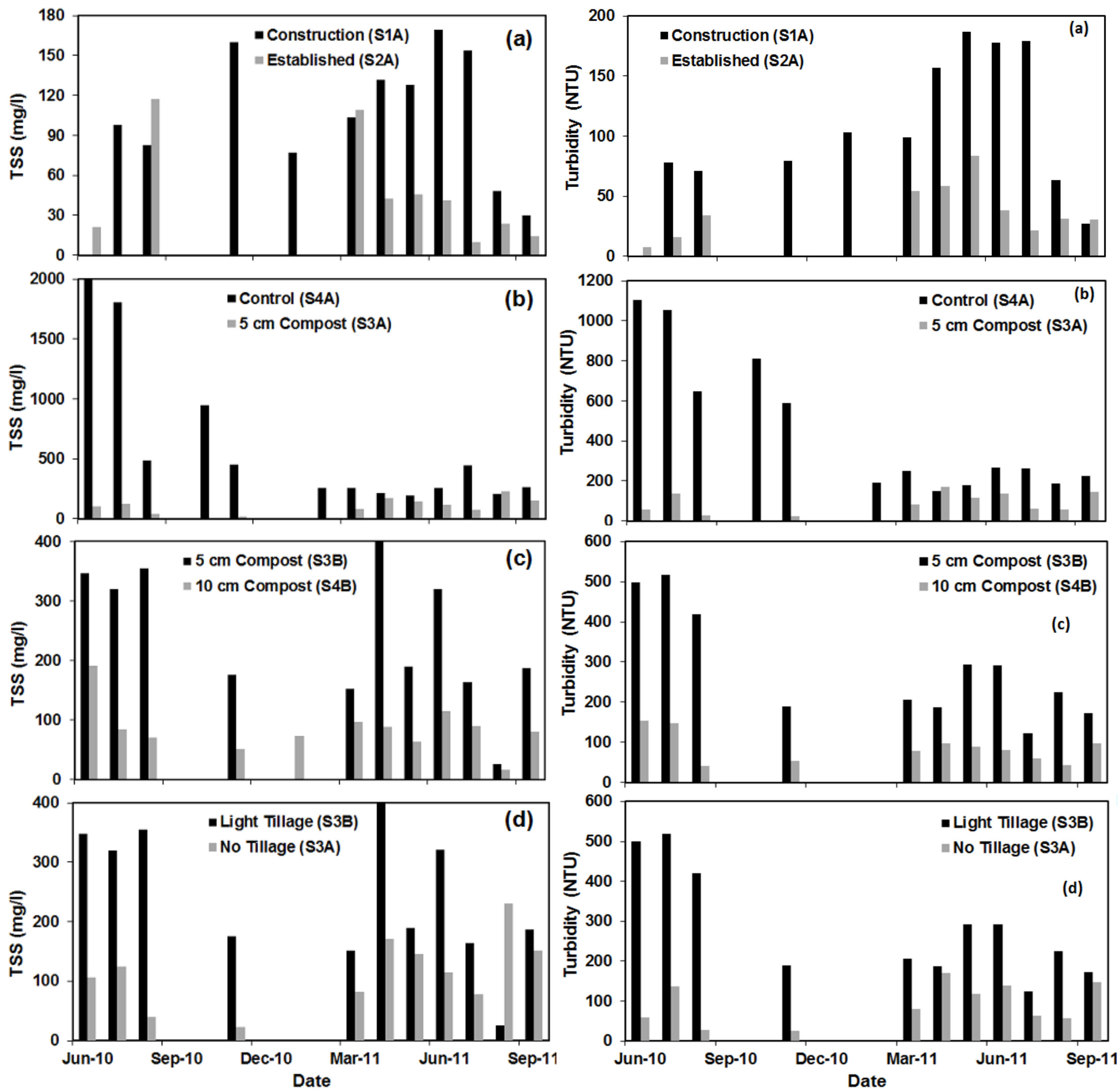

Figure 6.4. TSS (left) and turbidity (right) distribution; a) active construction vs. established areas, b) no vs. 5 cm compost/mulch, c) 5 cm vs. 10 cm compost/mulch, and d) non-tilled vs. light tillage plots in Louisiana.

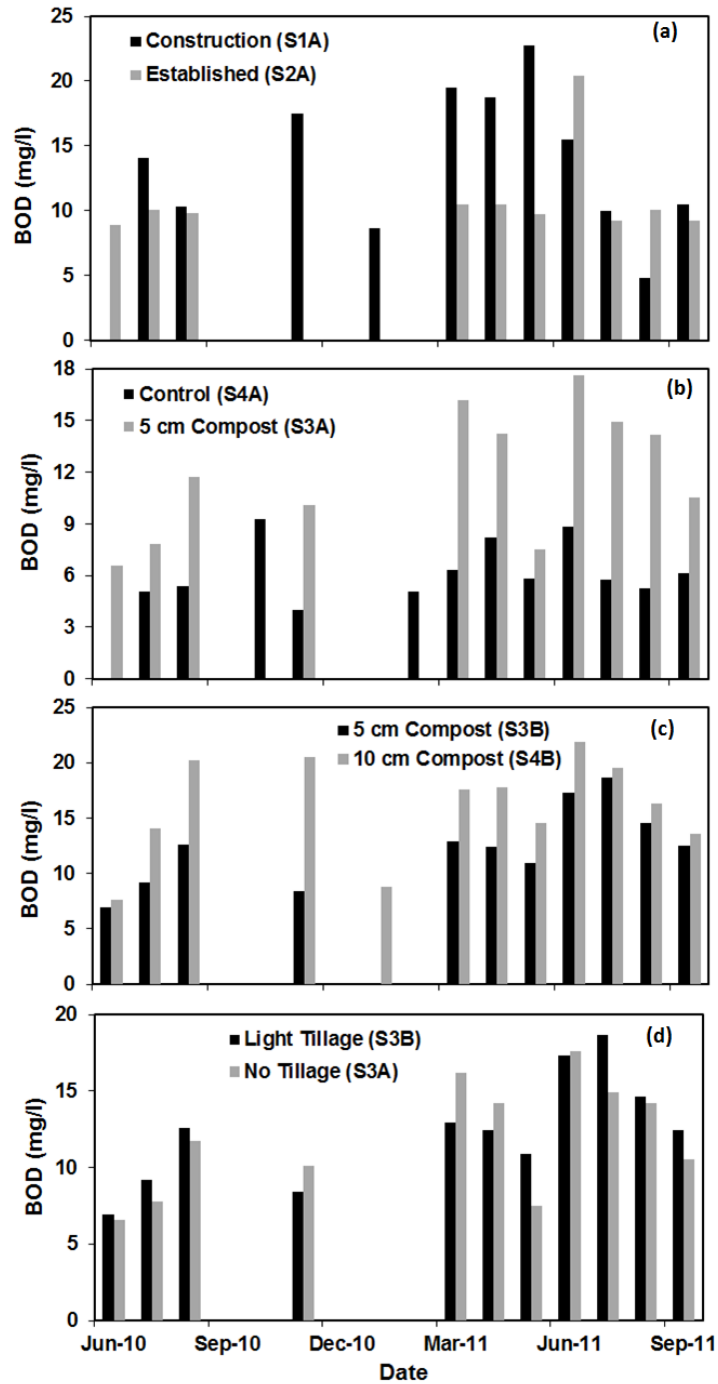

Figure 6.5. BOD distribution; a) active construction vs. established areas, b) control vs. 5cm compost/mulch, c) 5cm vs. 10cm compost/mulch, and d) non-tilled vs. light tillage, Louisiana.

6.3.4.2 Active construction versus established sites

In Table 6.6, the mean difference was used to estimate the variability of data due to construction activities. The mean difference of the TSS and associated turbidity, as an indication for soil erosion, between the two plots showed the influence of construction on sediment

contained in runoff. The results revealed that the S1A (active construction site) mean is higher than the S2A (established site) mean by 70 mg l^{-1} and 64 nephelometric turbidity units (NTU) for TSS and turbidity, respectively. Although the soil surface was covered by 10 cm compost/mulch (S1A), construction activities resulted in significant increases in TSS and turbidity by 79% and 70%, respectively. A similar trend can be observed for the BOD results since the mean difference value of the active construction site was higher (2.6 mg l^{-1}) than the established site by 21% increase in BOD. Additionally, the mean difference results for pH and EC showed that they were higher in S2A compared with S1A by 0.2 and 40 µS cm^{-1}, respectively. This may be related to the inherent soil properties rather than construction activities.

Table 6.6. Basic statistical measurements for runoff water samples from active construction site (S1A) vs. established site (S2A), 5 cm compost/mulch (S3A) vs. control (S4A), and 5 cm compost/mulch (S3B) vs. 10 cm compost/mulch (S4B), Louisiana.

Plot	TSS[a] (mg/l)	Turbidity (NTU)	BOD[b] (mg/l)	pH	EC[c] (µS/cm)
			Mean (SD[d])		
S1A	89.4 (39.8)	91.8 (47.3)	12.6 (5.9)	6.4 (0.3)	115.3 (37.6)
S2A	19.7 (15.0)	27.8 (21.4)	10.0 (1.9)	6.7 (0.4)	155.1 (83.5)
Mean difference	69.7	63.97	2.63	-0.24	-39.79
S3A	98.1 (62.7)	94.5 (60.8)	12.7 (5.2)	5.9 (0.4)	60.6 (15.3)
S4A	331.8 (142.5)	322.6 (194.6)	6.4 (2.8)	5.9 (0.3)	38.1 (13.7)
Mean difference	-233.7	-228.1	6.3	0.08	22.58
S3B	275.3 (132.8)	268.7 (133.4)	13.5 (4.6)	6.2 (0.4)	74.3 (24.2)
S4B	75.7 (25.5)	80.1 (54.3)	18.1 (7.0)	6.4 (0.4)	118.6 (39.3)
Mean difference	199.7	188.6	-4.59	-0.26	-44.22
S3A	98.1 (62.7)	94.5 (60.8)	12.7 (5.2)	5.9 (0.4)	60.6 (15.3)
S3B	275.3 (132.8)	268.7 (133.4)	13.5 (4.6)	6.2 (0.4)	74.3 (24.2)
Mean difference	-177.2	-174.2	-0.8	-0.21	-13.7

[a] total suspended solids, [b] biochemical oxygen demand, [c] electrical conductivity, [d] standard deviation

6.3.4.3 Compost/mulch application (5 cm) versus control (no compost/mulch)

The mean difference of the TSS and turbidity values between the two plots was greatly influenced by compost/mulch application (Table 6.6), which was consistent with Nunes et al.

(2011). They found that a bare surface (control plot) could serve as a good indicator for soil erosion while soil coverage (compost/mulch) could conserve those soils. Our results showed that the S4A mean was higher than the S3A mean for turbidity and TSS by 228 NTU and 234 mg l^{-1}, respectively. Thus, the TSS and turbidity decreased by 71% and 70%, for the 5 cm compost/mulch plot compared to the control plot, respectively. Conversely, the BOD mean in the control plot (S4A) was less than the 5 cm compost/mulch plot (S3A) by 6 mg l^{-1} which indicated that there was a 98% reduction in the BOD with no compost/mulch application. As expected, the TSS increase the light attenuation coefficient and reduce the amount of light available for photosynthesis, which leads to less DO production (Ji, 2008). Moreover, mean difference results in Table 6.6 show that pH was the same and EC was higher in plot S3A (5 cm compost/mulch) compared with S4A (control) by 23 µS cm^{-1}. This was mainly related to the compost/mulch properties applied to the S3A plot.

6.3.4.4 Compost/mulch application (5 cm) versus (10 cm)

In Table 6.6, TSS and turbidity mean difference values were reduced by 200 mg l^{-1} and189 NTU, respectively, for the 10 cm compost/mulch plot (S4B) compared to the 5 cm compost/mulch plot (S3B). Therefore, 5 cm compost/mulch increased TSS and turbidity by 73% and 70%, respectively, compared with 10 cm compost/mulch application. Additionally, decreasing the compost/mulch thickness to 5 cm reduced the BOD mean by 5 mg l^{-1}, which represented a 34% reduction compared with the 10 cm compost/mulch application. Although, using 10 cm compost/mulch decreased the TSS and turbidity values in water samples, it added more organic matter. This increasing of organic matter led to an increase in BOD values. This indicates that oxygen was depleted in the runoff, and less oxygen was available for the surface water (Ji, 2008).

6.3.4.5 Tillage versus no-tillage

With all other variables held constant (slope, construction activities, compost/mulch thickness, and soil characteristics), the tillage effect was examined for site 3 (Table 6.6). Turbidity and TSS results showed significant differences between the tillage practices; mean turbidity and TSS values of the tilled plot (S3B) were increased by 174 NTU and 177 mg l^{-1}, respectively, compared to the non-tilled plot (S3A). Therefore, the tillage application increased TSS and turbidity by 64% and 65%, respectively compared with the non-tillage application. Also, the BOD mean value was increased when tillage was applied by 0.8 mg l^{-1} (6% increases in BOD).

6.3.4.6 Two samples *t*-test results

Evaluation of construction activities, differential compost/mulch coverage, and tillage practices on water quality was accomplished using a two samples *t*-test. Table 6.7 provides summary results of *P* values for each assigned parameter between each pair of comparison plots.

The TSS and turbidity results indicated that there were significant differences due to construction activities (S1A vs. S2A), 5 cm compost/mulch application versus control (S3A vs. S4A), 5 cm versus 10 cm compost/mulch applications (S3B vs. S4B), and tillage practices (S3A vs. S3B). As all *P* values for all the aforementioned cases were less than the significance level of 0.05, TSS and turbidity were significantly impacted by all of the variables (construction activities, compost/mulch thickness, and tillage practices). Additionally, the compost/mulch thickness more strongly affected TSS and turbidity than construction activities and tillage practices. *P* values for TSS and turbidity in those cases (S3A vs. S4A and S3B vs. S4B) provided the highest significant differences of 0.0048 and 0.0001 for TSS and 0.0001 and 0.0006 for turbidity, respectively. This conclusion matches the results from factor analysis. For BOD, *P* values for the pairs (S1A vs. S2A and S3A vs. S4A) were significantly less than 0.05. Thus, we

reject the null hypothesis and conclude that there were significant differences in the BOD due to construction activities and the 5 cm compost/mulch treatment. Conversely, for the other two pairs (S3B vs.S4B and S3A vs. S3B), *P* values for BOD were higher than 0.05; thus, no significant differences exist. This indicated that BOD was not significantly changed by increased compost/mulch thickness to 10 cm, or when different tillage practices were imposed. Finally, the *P* values for pH and EC across all plots indicated that there were no significant differences due to construction activities, compost/mulch applications, or tillage practices.

Table 6.7. Two samples *t*-test results for significant difference between water quality parameters among different compost/mulch plots, Louisiana.

	TSS[a] (mg/l)	Turbidity (NTU)	BOD[b] (mg/l)	pH	EC[c] (µS/cm)
	--------------------------------------- *P*				------------------------------------
S1A vs S2A	0.0136*	0.0262*	0.0030*	0.1543	0.0028*
S3A vs S4A	0.0048*	0.0001*	0.0222*	0.2393	0.6661
S3B vs S4B	0.0001*	0.0006*	0.1292	0.5692	0.0698
S3A vs S3B	0.0103*	0.0075*	0.6810	0.8770	0.1087

*significantly difference at significant level (α=0.05)
[a] total suspended solids, [b] biochemical oxygen demand, [c] electrical conductivity

To quantify the correlation between the water quality parameters, PROC CORR in SAS was conducted and produced the Pearson correlation matrix (Table 6.8). The correlation matrix indicated a positive, very strong correlation between turbidity and TSS (0.94). Additionally, turbidity had a negative correlation with BOD, with a 0.32 correlation coefficient. Besides, negative correlations were observed between pH and turbidity as well as pH and TSS by 0.43 and 0.37, respectively.

Table 6.8. Correlation coefficient of the water quality parameters based on the four sites dataset.

	Pearson Correlation Coefficients				
	TSS[a]	Turbidity	BOD[b]	pH	EC[c]
TSS	1.00				
Turbidity	0.94	1.00			
BOD	-0.29	-0.32	1.00		
pH	-0.37	-0.43	0.11	1.00	
EC	-0.27	-0.36	0.40	0.55	1.00

[a] total suspended solids, [b] biochemical oxygen demand, [c] electrical conductivity

6.4 CONCLUSIONS AND RECOMMENDATIONS

Based on the results of this study, the effectiveness of compost/mulch cover in reducing runoff, TSS, and turbidity from soils susceptible to high-intensity storms in Louisiana was confirmed. This finding is attributed to the positive effect of compost/mulch coverage on the flow rate. The total flow increased when soils were not covered (control) or when the compost/mulch was incorporated with the top soil (light tillage). Statistical analyses showed that TSS and turbidity are significantly influenced by the compost /mulch thickness (0, 5, and 10 cm) at the plot scale. As such, compost/mulch thickness was the most influential variable on the two water quality parameters. Decreases in TSS were observed when compost/mulch was applied on the soil surface (70% and 74% reduction in TSS for the 5 cm and 10 cm compost/mulch applications, respectively). Significant differences were also observed between active construction plots versus established plots and light tillage versus non-tilled plots.

The incorporation of compost/mulch into the soil via light tillage decreased its effectiveness in reducing flow rate and sediment losses compared to compost/mulch which was surface applied with no tillage. Based on these observations, tillage incorporation of compost/mulch is not recommended, since it caused surface disturbance and increased erosion rates. Soils which are highly susceptible to erosion, such as those in the current study, require special attention during construction activities. Based on the results of the current study, compost/mulch coverage is recommended as a BMP in both active construction areas and established areas prone to soil erosion on roadsides. Furthermore, while 10 cm of compost/mulch coverage is superior in reducing the total flow and TSS, 5 cm of compost/mulch application may strike the most economical balance between benefits received and cost of BMP implementation.

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CHAPTER 7. SEDIMENT AND PHOSPHOROUS LOSS FROM COMPOST/MULCH TREATED ROADSIDE SOILS USING SIMULATED RAINFALL

7.1 INTRODUCTION

Roadside erosion is a serious issue which threatens surface water quality. Many approaches have been suggested to quantify erosion from roadsides including field-based roadside or stream monitoring, sediment tracing, and the use of roadside erosion models (Fu *et al*., 2005). The amount of sediment reaching surface waters is a direct result of erosion and sediment transport from the soil surface. Road construction causes extensive surface disturbance with adverse impacts on adjacent environments, especially with a soil highly susceptible to erosion, poor vegetative cover, and a fragile ecosystem. Typical site disturbances associated with road construction include removal of topsoil, destruction of native vegetation, and alteration of natural topography, which frequently result in severe surface runoff and water erosion (Xu *et al*., 2006). Sediment from construction sites can easily be delivered into adjacent streams, and become serious nonpoint source pollution (NPSP) for the receiving environment (Lane and Sheridan, 2002). To control roadside erosion and reduce its impacts on water quality, a careful evaluation of surface runoff, sediments loss, and application of best management practices (BMPs) is required. Greene et al. (1994) indicated that increased plant cover reduced runoff, but had no effect on sediment concentrations when rainfall for 30 and 60 minutes was applied. They also found that greater surface roughness of vegetated plots contributed to the reduction in runoff. These results were consistent with Freebairn and Gupta (1990), who pointed out that infiltration rates were controlled primarily by surface crusting and the amount of rainfall. Navas (1993) used a rainfall simulator for 15 minutes over 1.25 m X 1.25 m plots and determined that greater vegetative cover and rock outcrops reduced runoff and soil loss, whereas greater slope increased runoff and soil loss.

Mulch is an effective tool for controlling soil loss and erosion, since soil surface coverage increases the flow depth and reduces the runoff velocity and kinetic force of raindrops on soil (Mutchler and Young, 1975). Wischmeier (1984) reported that a 90% mulch cover reduces erosion by 93%. Adekalu *et al*. (2006) reported that mulching reduces surface runoff during and after rainfall, increases infiltration, and reduces soil loss. In their rainfall simulation experiment at an intensity of 100 mm h^{-1} for 1-hour, they found that soil loss and runoff were significantly correlated with the mulch cover. Jordán *et al*. (2010) conducted a 3-year experiment with five straw mulching rates under cultivated soils in semi-arid conditions. Their findings indicated that besides enhancing physiochemical properties of the soil, runoff rates and soil loss at a simulated rainfall intensity of 65 mm h^{-1} were greatly reduced with mulch applications. In simulated rainfall experiments, runoff and soil loss were observed to be reduced by increased mulch cover percentage (Osunbitan and Adekalu, 1997). Arthur *et al*. (2011) reported that compost has a positive influence on soil physical properties since its application improved the aggregate stability of a silt loam soil. Although compost application is generally considered a BMP for erosion control, there are still disagreements in the literature its effectiveness in reducing soil loss due to water erosion. For example, some studies have reported that the application of manure and compost does not have any effect on runoff and erosion under simulated rainfall (Gilley and Eghball, 1998; Edwards *et al*., 2000; Arthur *et al*. 2011). Conversely, significant reductions in soil loss and erosion rate were reported in other studies (Faucette *et al*., 2004; Persyn *et al*., 2004; Birt *et al*., 2007) when different types of compost were applied.

Surface runoff is a function of many variables including rainfall intensity and duration, soil type, soil moisture, land use, cover, and slope (Elhakeem and Papanicolaou, 2009). Several runoff studies have depended on natural rain variability in intensity, drop size, drop energy, spatial and temporal distribution. However, rainfall simulation, which has been used in the soil

erosion studies several decades, allows for rapid and reproducible collection of data in laboratory and field experiments (Miller, 1987; Esteves *et al*., 2000, Dunkerley, 2008). Rainfall simulation provides a convenient tool that allows uniform and controlled rain events to be created. Additionally, rainfall simulation provides a valuable tool for exploring processes in extreme events, which are difficult to study otherwise, owing to their relative rarity (Clarke and Walsh, 2007). The use of simulated rain is an acceptable way to extend field observations made under natural rain in a controlled way. The field plots areas of previous studies and rain duration range from <0.1 m^2 and <5 minutes duration (Casermeiro *et al*., 2004) to >300 m^2 (Lusby and Toy, 1976) and duration of 6–9 h (Armstrong *et al*., 1998; Scherrer *et al*., 2007). Rainfall intensity values of 60-100 mm h^{-1} are mostly commonly cited in rainfall simulation literature (Clarke and Walsh, 2007). In the US, many studies have utilized different rainfall simulator intensities and durations in different places across the country (e.g. Frauenfeld and Truman 2004; Keim *et al*., 2006; Green et al., 2000; Martin and Moody 2001; Smith et al., 2007; Neave and Rayburg 2007; Wan *et al*., 1996). The results from simulated rainfall only give relative, rather than absolute, erosion data; to correlate the simulations to natural events, data from similar long-term natural rainfall events must be available for comparison (Meyer, 1988). Although rainfall simulation over small plot has certain shortcomings, such as edge effects by the plot frame and relatively small runoff components, it is the most practical method available to study field rainfall runoff and erosion processes in many situations.

The contribution of phosphorous (P) to eutrophication of surface waters has been confirmed in many studies (e.g. Jarvie *et al*., 2002; Ji, 2008; Hecky and Kilham, 1988). Spivakov *et al*. (1999) reported that P is a limiting nutrient determining the biological productivity for most inland waters as the P fractions of environmental interest are often developed via eutrophication. Eutrophication negatively impacts the aquatic environment, since it can lead to algal blooms,

reduced dissolved oxygen levels, fish kills, and reduced biodiversity, which may sharply reduce surface water quality (Ji, 2008). In the US, eutrophication is one of the most widespread water quality impairments; with agriculture a primary source of P in the surface waters of many watersheds (United State Environmental Protection Agency, USEPA, 1996; Litke, 1999). The *National Phosphorus Research Project* (NPRP) was launched to assess the effects of soil properties, mainly soil test P, and management on P loss in overland flow (Sharpley *et al*., 2002).

Louisiana has a humid subtropical climate with abundant rainfall resulting from dominant winds from the south/southeast directing warm, moist air inland from the Gulf of Mexico. The annual average precipitation statewide ranges from 122 cm in northwest to 163 cm in the southeastern coastal plains (Louisiana Department of Environmental Quality, LDEQ, 2010). Around 16% of state surface area is covered with water resources to include; rivers and streams, lakes and reservoirs, fresh and tidal wetlands, and estuaries. Multiple nonpoint source pollutants have been recognized in Louisiana surface waters and are mainly associated with land-use activities. Typical impairments include sediment and total suspended sediment (TSS), nutrients (mostly N and P), metals, organic materials, and bacteria (LDEQ, 2010).

Although, there are well documented studies relating to runoff volume, sediment, and nutrient export from agriculture fields, comparatively fewer studies reported those parameters along roadsides. The overall goal of the study was to evaluate the impacts of compost/mulch application, on runoff and the export of sediment and P from highway roadsides using rainfall simulation in-situ. The specific objectives were to: 1) assess the influence of different compost/mulch thickness on runoff volume and flow rate; 2) quantify the effect of light-tillage incorporation and slope on runoff volume, flow rate, and total runoff; and 3) examine the sediment and P loss in the associated runoff.

7.2 MATERIALS AND METHODS

7.2.1 Study site and plots design

The rainfall simulation experiment was conducted along two highway roadside locations in Louisiana, USA (Figure 7.1). The first location was in West Feliciana Parish in an active construction area adjacent to the northbound lane of US Highway 61 (~8 km from St. Francisville, LA) and had only one site (site 1), with steepest slope of 34%. Conversely, another location in southern Rapides Parish along of IH-49 (~20 km from Bunkie, LA) was established in an area prone to erosive undercutting in many areas. This location had three sites; site 2 adjacent to the northbound lane, site 3 adjacent to the southbound lane, and site 4 in the center median; with slope percentages of 25%, 15%, and 10%, respectively. At each of the four sites, two plots (side-by-side) were constructed at a fixed size of 4.0 m X 4.0 m. The eight plots were bordered by heavy gauge steel edging on three sides to prevent overland flow from entering the plots. The downslope side directed runoff from the plots into 0.305 m depth H flumes. At each site, one of the two plots was lightly tilled and one kept non-tilled (Bakr *et al*., 2012).

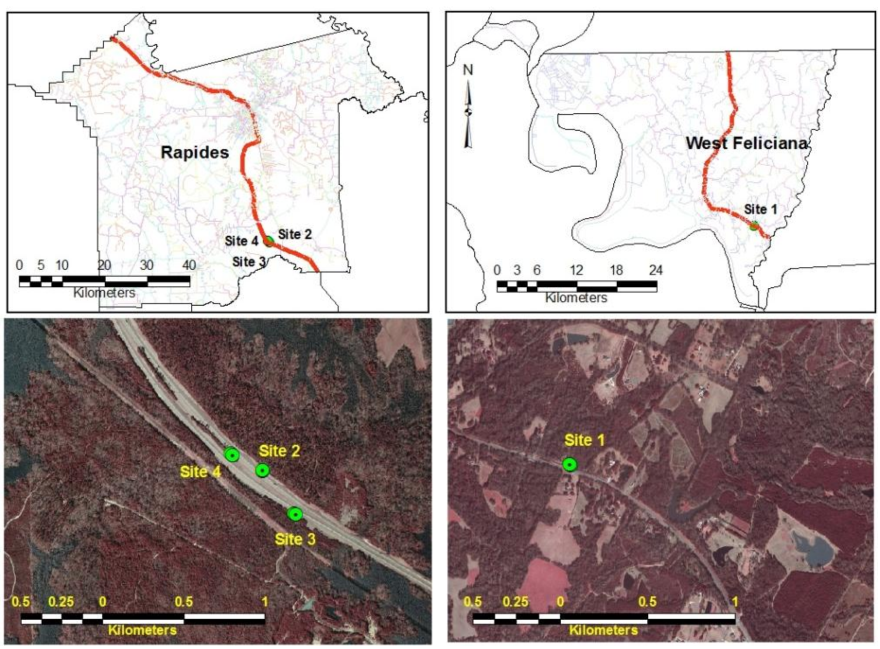

Figure 7.1. General locations of the study sides on US Highway 61 (right) and IH-49 (left) in Louisiana, USA.

Different compost/mulch applications were utilized over the plots, three compost/mulch thicknesses were applied; 10 cm, 5 cm, and no-compost/mulch (as a control). The mulch materials were locally harvested (70% hardwood and 30% pine trees). Compost blended into the mulch was a double-ground, screened, recycled wood fiber material, also harvested locally (Bakr *et al*., 2012). Local species were preferably used for erosion control to reduce the cost of construction and maintenance, and to protect the local ecosystem (Grace, 2002).

7.2.2 Rainfall simulation

In April 2011 and May 2012, two different rainfall simulation experiments were conducted based on the United States Department of Agriculture (USDA) National Phosphorous Research Project (NPRP) (2004) protocol for rain simulation. A Tlaloc 3000 portable rainfall simulator (Joern's Inc., West Lafayette, IN) was used. The simulated rain produced runoff from the release plots to investigate the effect of compost/mulch applications, tillage practices, and slope on water runoff, sediment, and P losses. The rainfall simulator featured a design suggested by Miller (1987), and had a ground cover area of 2.81m X 2.32m. One ½ HHSS50WSQ Tee Jet nozzle (Spraying Systems Co.,Wheaton, IL) was placed in the center of the simulator to apply rain over the target plots at 3.0 m above the surface. The simulator was calibrated by measuring the volume and distribution of rain and adjusting the pressure regulator to 3.5 psi for the desired rain intensity. Polyethylene tarps were used as a windscreen on the sides of the simulator frame to minimize wind disturbance. Regular tap water was pumped to the simulator from a 250-gal (946.35 L) trailer-mounted tank kept on the roadside. Due to the slope of the plots, the framework was leveled in order to allow for direct spray from the nozzle on the release plot. Rainfall simulator installation and application photos are presented in Figure 7.2.

Figure 7.2. Rainfall simulator installation and application in the study sites in Louisiana, USA.

Two rainfall simulation pulses were conducted for each plot in each year. Pulse 1 and pulse 2 were conducted in 2011, and pulse 3 and pulse 4 were conducted in 2012. The first pulses in both years (1 and 3) were applied during dry field conditions while the second pulses (2 and 4) were on soils which had exceeded field capacity; water present from the initial pulses. Plastic sheeting was placed on the end of the open edge at each plot to maintain only the desired ground area under the simulator. The simulated rainfall was generated at an intensity of 75 mm h^{-1} (±5) in 2011 and 80 mm h^{-1} (±5) in 2012 for 30 minutes on all plots. Runoff was directed to the flume, and then collected downslope in a 5-gal (19 L) bucket. Time was recorded as runoff

began to accumulate in the 5-gal bucket. A 500-mL of runoff sample was collected every one minute for the first five minutes from the recorded time then one sample was collected every five minutes until runoff time reached 40 minutes (10 minutes after the rain pulse was stopped). After the runoff was completely stopped (no more drip from flume to the bucket), the last sample was collected and the time was also recorded. The sample volume was recorded every one minute during the entire pulse to calculate the flow rate and cumulative runoff volume. The water samples were treated for laboratory analysis per the *American Public Health Association* (APHA, 2005). The pH was measured immediately after transferring samples to the lab using an Orion 2 Star pH meter (Thermo Scientific, Waltham, MA), and then the samples were filtered and acidified to lower the pH to ~2. The filtered and unfiltered runoff samples were kept refrigerated at 4ºC for further analyses. Besides the pH analysis, runoff samples were subjected to electrical conductivity (EC) measured by 4063 Traceable® Portable Conductivity Meter (Control Company, Friendswood, TX), TSS based on (APHA, 2005), turbidity using HF-Micro 100 laboratory turbidimeter (HF Scientific Inc., Fort Myers, FL), dissolved P, and total P testing based on (Pierzynski, 2000). A Ciros model (Marlboro, MA, USA) inductively coupled plasma–atomic emission spectrometer (ICP–AES) was used to measure the concentration of dissolved P, and total P. Before, during, and after each pulse, surface soil moisture and temperature were recorded each minute, 0-10cm, using soil moisture smart sensors (S-SMx-M005) and 12-Bit temperature smart sensors (S-TMP-M006) attached to HOBO micro station (On-set Computer Corp., Bourne, MA, USA).

7.2.3 Soil samples

Multiple soil samples were collected from each plot in both 2011 and 2012. The collected soil samples were air-dried and grounded to pass a 2-mm sieve. Laboratory analysis of soil properties included soil pH using an Orion 2 Star pH meter (Thermo Scientific, Waltham, MA),

soil organic matter (SOM) via loss-on-ignition (LOI) (Nelson and Sommers, 1996), soil texture via pipette method (Gee and Bauder, 1986), and available soil P via Mehlich 3 extraction (Mehlich, 1984).

7.2.4 Statistical analyses

One way analysis of variance (ANOVA) was performed on all water sample characteristics to determine the effects of construction activities and post construction activities, compost/mulch thickness (0- as control, 5-, and 10-cm), tillage incorporation, and slope factor on the flow rate, runoff, TSS, and P loss. Tukey's test was used to identify the significant differences between means at different rain pulses in different plots. The PROC MIXED in SAS® 9.3 software (SAS, 2011) was used where compost/mulch treatments and pulses were the main effects. The significant differences were determined using $p \leq$ significant level (α) = 0.05.

7.3 RESULTS AND DISCUSSION

7.3.1 General soil characteristics

The main physiochemical characteristics of the soils for each studied plot are given in Table 7.1. Soil pH (4.96-6.66) ranged from very strongly acidic to slightly acidic (Soil Survey Staff, 1993) in all plots. The results also revealed that the available P in soils in all studied plots was very low (1.3- to 5.6 $\mu g\ g^{-1}$). The soil texture was greatly affected by the compost/mulch application when monitor over the two years. With 10cm compost/mulch application and no-tillage incorporation, the change in sand, silt and clay content were minimal as observed in plots S1A and S2A (Table 7.1). However, when compost/mulch was incorporated into soil by tillage, and when the compost/mulch thickness was decreased, changes were noted among soil textural separates. The most substantial alteration was in the silt fraction, as soils with higher silt content are more susceptible to erosion (Wischmeier and Mannering, 1969). As expected, SOM increased from 2011 to 2012; a direct result of compost/mulch treatments. The increase in the

SOM was observed at all plots, even to a small extent within the control plots. Increases in control plot SOM were likely due to grass and weed germination in the control plots with some root residue and detritus present.

Table 7.1. Selected physiochemical soil properties for each plot along US Highway 61 and IH-49 roadsides in Louisiana, USA.

Property	Year	S1A	S1B	S2A	S2B	S3A	S3B	S4A	S4B
pH	2011	5.02	5.26	4.96	4.98	5.38	5.62	6.01	5.89
	2012	5.13	5.43	5.31	5.01	5.45	5.77	6.66	6.29
		-----------------------------------%-----------------------------------							
Sand	2011	40.98	45.86	27.09	26.00	36.66	25.34	58.74	64.23
	2012	35.65	45.90	30.58	39.34	27.88	28.01	59.57	61.65
Silt	2011	34.69	27.69	47.49	49.84	41.92	44.59	24.97	19.39
	2012	31.07	28.08	44.83	35.36	47.79	42.27	29.59	21.52
Clay	2011	24.33	26.45	25.42	24.16	21.42	30.03	16.29	16.38
	2012	33.28	26.03	24.60	25.30	24.33	29.72	10.84	16.82
OM	2011	1.09	0.88	1.89	1.71	1.50	2.05	0.52	1.11
	2012	1.84	1.69	2.87	2.00	2.37	2.92	1.20	1.64
		-------------------------------$\mu g\ g^{-1}$-------------------------------							
P	2011	4.18	1.90	2.32	2.18	2.53	2.75	1.32	3.49
	2012	4.19	3.84	5.61	3.39	3.50	4.26	2.42	4.47

7.3.2 Soil moisture/temperature

Soil moisture/temperature values were recorded each minute for the entire experiment. The initial values of soil water content ($cm^3\ cm^{-3}$), when starting the simulated rainfall at each pulse for each plot, are given in Table 7.2. Figures 7.3 and 7.4 show the change in soil moisture /temperature due to the application of the rain pulses over each plot in 2011 and 2012. Figures 7.5 and 7.6 present the boxplot of moisture for each pulse and the significant difference between pulses within each plot.

Table 7.2. Soil moisture at the beginning of each rain pulse in each plot, Louisiana, USA.

Year	Pulses	S1A	S1B	S2A	S2B	S3A	S3B	S4A	S4B
		Initial water content ($cm^3\ cm^{-3}$)							
2011	Pulse 1	0.3235	0.1390	0.3133	0.0076	0.2385	0.3003	0.1012	0.1819
	Pulse 2	0.3351	0.2487	0.3569	0.1630	0.2930	0.3184	0.2058	0.2095
2012	Pulse 3	0.2189	0.0991	0.1521	0.0010	---	0.0548	0.0134	0.0264
	Pulse 4	0.3090	0.2770	0.2676	0.1136	---	0.0548	0.2256	0.2705

For site 1, plot S1A soil moisture results indicated that during 2011 the soil moisture content was relatively higher compared to 2012 (Figure 7.3a). Additionally, no significant increase in soil moisture was detected between pulses 1 and 2 (Figure 7.5a). Conversely, in 2012 the soil surface was drier as the initial water content was lower. When pulse 3 was applied the water content sharply increased to a maximum of 0.35 cm^3 cm^{-3} during rain application, decreased to 0.31 cm^3 cm^{-3} between pulses, and then increased again with pulse 4 application. There were significant differences between pulses 3 and 4, pulses 1 and 3, and pulses 2 and 4. Surface soil temperature for the two years (2011 and 2012) had the same trend, however, 2012 was warmer than 2011 (as May 2012 was warmer than April 2011). The first pulses (1 and 3) did not affect significantly on soil temperature. However, pulses (2 and 4) increased soil temperature.

For plot S1B (5 cm compost/mulch incorporated via tillage), an irregular pattern was observed for soil moisture/temperature (Figure 7.3b). Before pulses 1 and 3 were applied on plot S1B the soil water content was very low (0.09 cm^3 cm^{-3}). The soil surface was drier in 2012 than 2011 for plot S1B. The soil moisture results from S1B indicated that there were no significant differences between pulses1 and 2, and pulses 3 and 4; while between 2011 and 2012 the soil moisture was significantly different (Figure 7.5b). Soil temperature for 2011 had the same trend that was previously observed for plot S1A. However in 2012, with a relatively high temperature of soil surface, rain application resulted in a reduction of soil temperature. By the end of the experiment there was a gradual increase in soil temperature.

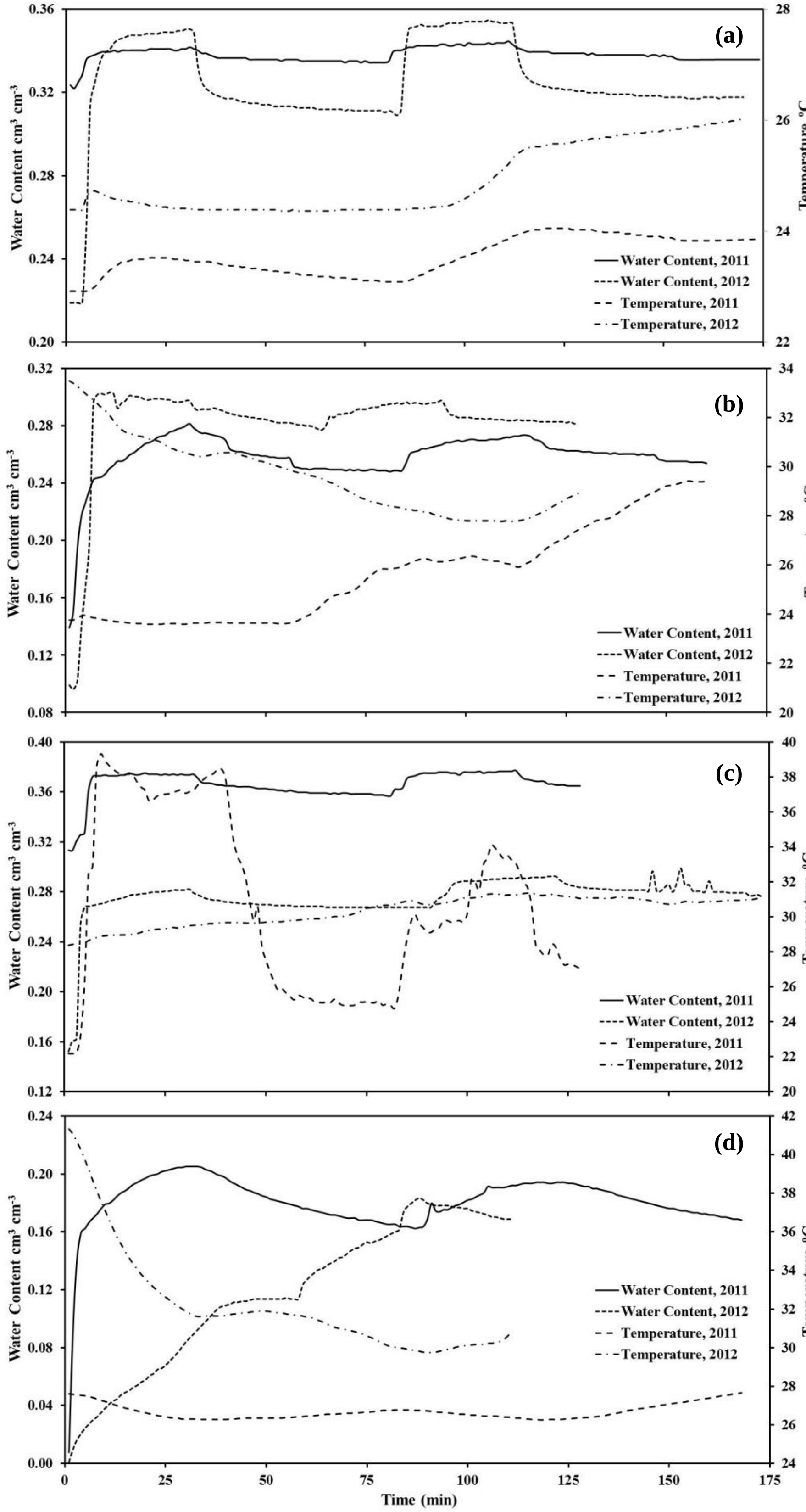

Figure 7.3. Soil water content and soil temperature during the entire rainfall simulation experiment period for sites 1 and 2, Louisiana, USA. (a) S1A, (b) S1B, (c) S2A, and (d) S2B.

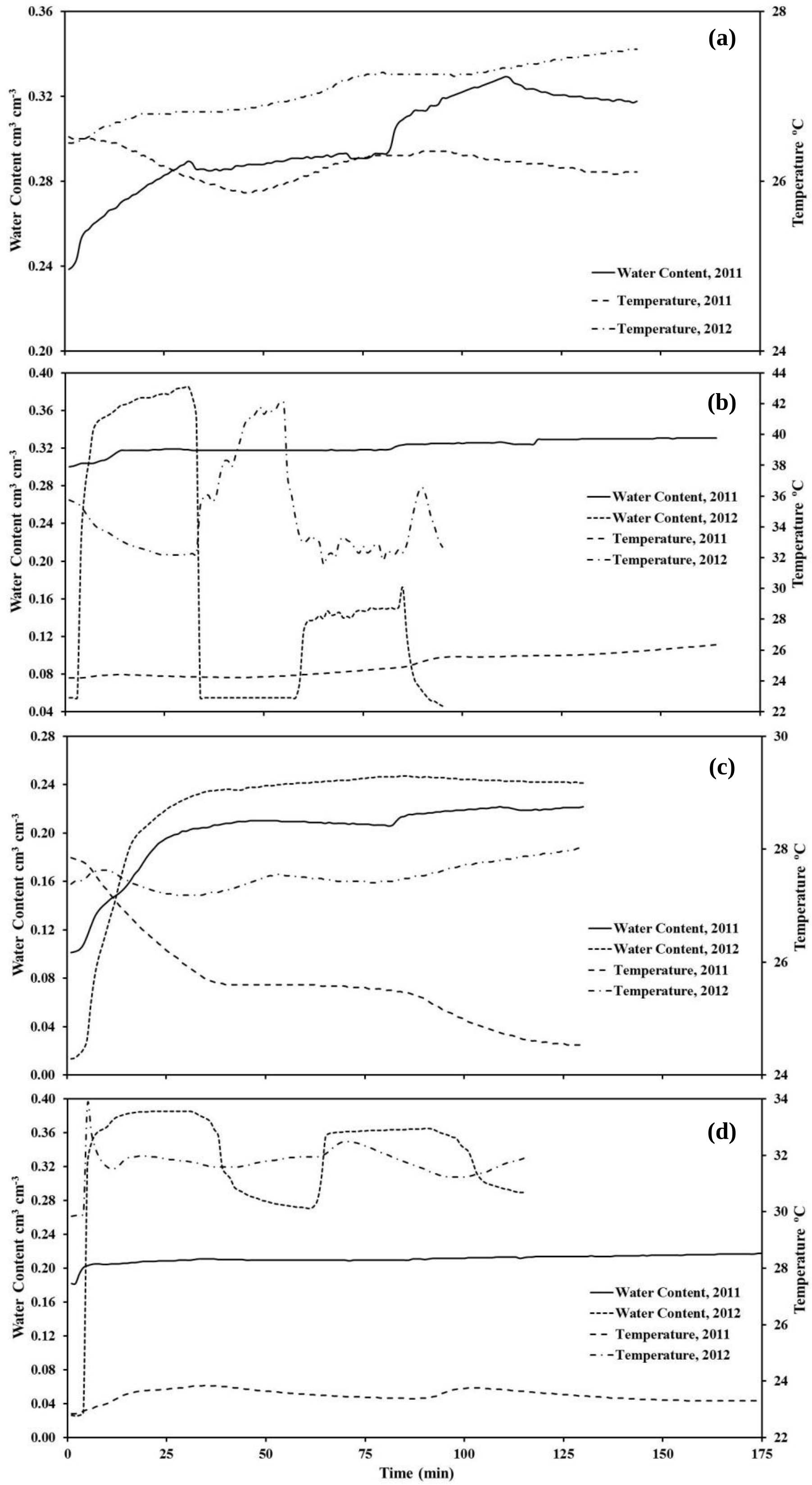

Figure 7.4. Soil water content and soil temperature during the entire rainfall simulation experiment period for site 3 and 4, Louisiana, USA. (a) S3A, (b) S3B, (c) S4A, and (d) S4B.

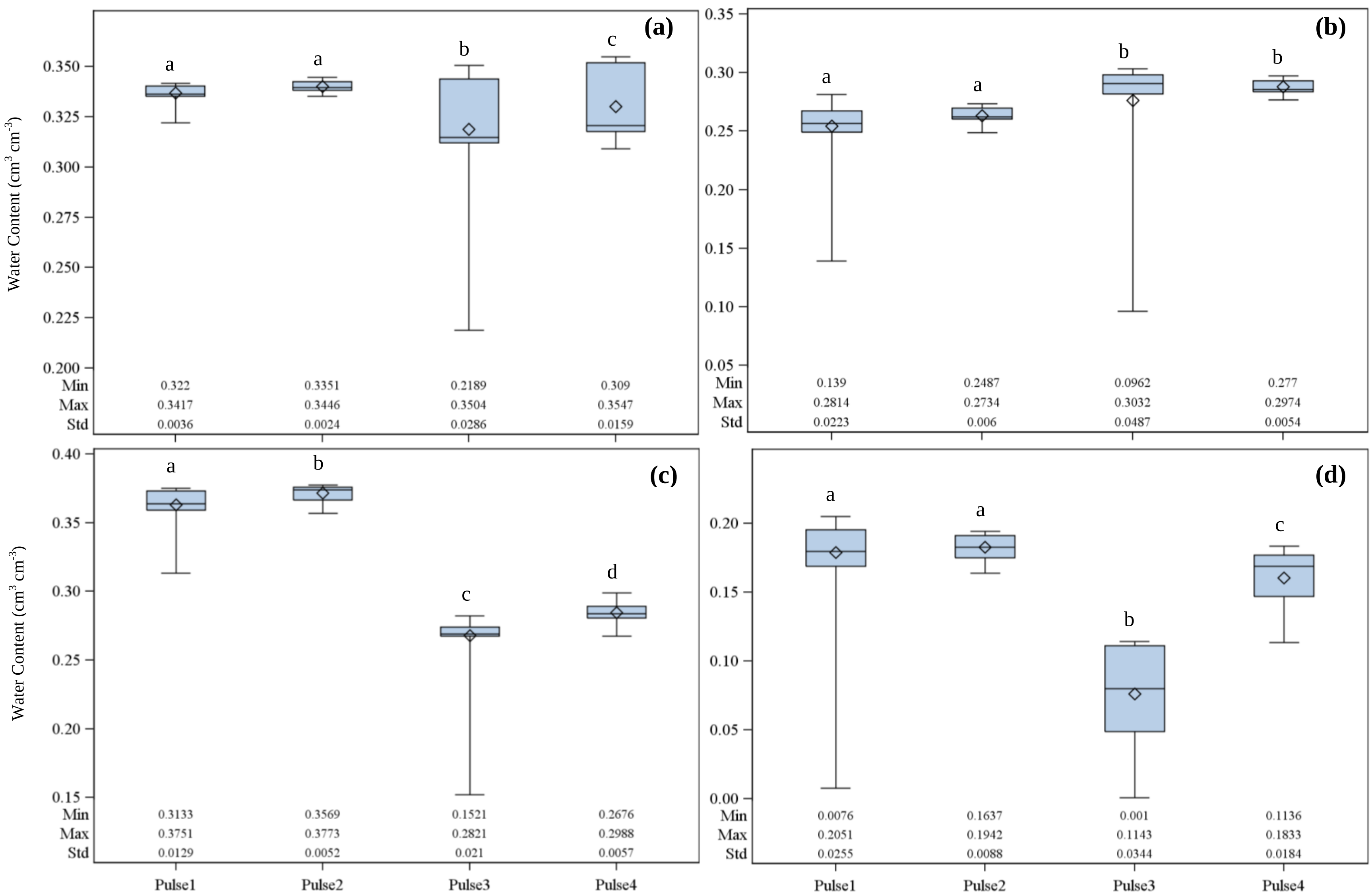

Figure 7.5. Soil water content boxplot, pulses 1 and 2 applied in 2011, and pulses 3 and 4 applied in 2012. Pulses with the same letter (a, b, c, or d) indicate no significant differences between them at α=0.05. (a) S1A, (b) S1B, (c) S2A, and (d) S2B.

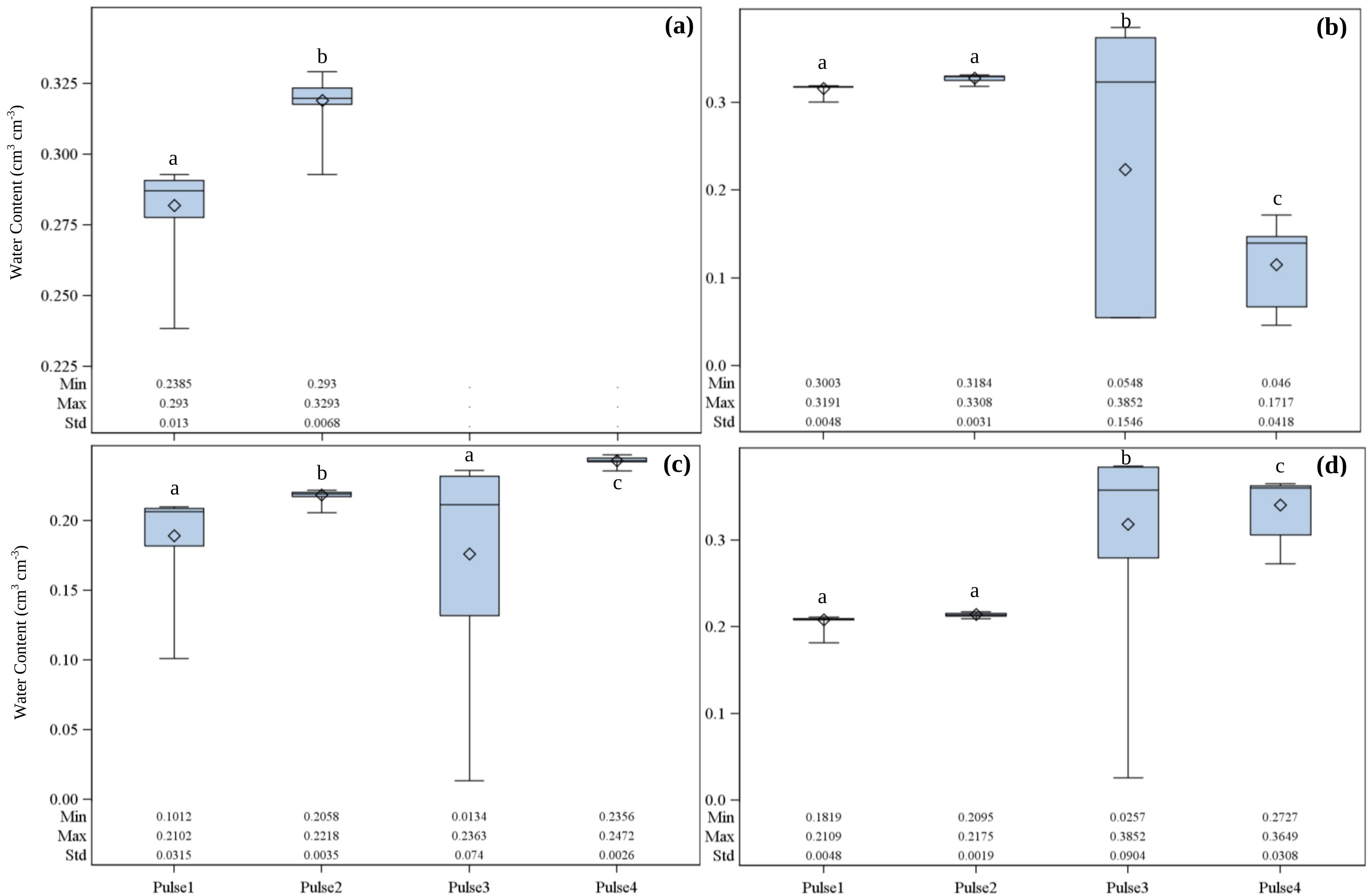

Figure 7.6. Soil water content boxplot, pulses 1 and 2 applied in 2011, and pulses 3 and 4 applied in 2012. Pulses with the same letter (a, b, c, or d) indicate no significant differences between them at α=0.05. (a) S3A, (b) S3B, (c) S4A, and (d) S4B.

At site 2, the soil moisture data for plot S2A had the same trend that existed in plot S1A since both plots received 10cm compost/mulch. During 2011, relatively higher water content was recorded in S2A compared to S1A (Figure 7.3c). All applied rain pulses were significantly different form each other (Figure 7.5c). In 2011, soil temperature increased with the first pulse, decreased between pulses, and then increased again with the second pulse. This trend is perfectly correlated with the soil moisture trend. In 2012, with the effect of 10cm compost/mulch coverage, relatively moderate temperature (30-32 ºC) was kept for the soil surface and slightly increased with the second pulse application.

With a bare soil surface, the S2B control plot provided the lowest soil moisture and highest soil temperature between all plots (Figure 7.3d). No significant difference was observed between pulses 1 and 2, while between pulses 3 and 4, 1 and 3, and 2 and 4, there were significant differences (Figure 7.5d). No change in soil temperature was recorded as a result of rain pulses (1 and 2). However, in 2012 as a temperature was over 40 ºC, the rain application roughly decreased soil temperature by 10 ºC.

At site 3, the soil moisture sensor for plot S3A failed to record data during the pulses that were applied in 2012. In 2011, soil moisture increased gradually with the application of pulse 1 then continually increased during application of pulse 2 (Figure 7.4a). Soil water content between the two applied pulses was significantly different (Figure 7.6a). No change in soil temperature was noted due to rain application for 2011 or 2012; a value of 27 ºC was the average soil temperature for the entire experiment.

In plot S3B, the soil moisture results showed no change in water due to the applied rain pulses in 2011. However, in 2012 a massive effect of rain pulses can be observed (Figure 7.4b). Thus, there was no significant difference in soil moisture for pulses 1 and 2 while pulses 3 and 4 were significantly different between each other and from pulses 1 and 2 (Figure 7.6b). Similar to

the soil moisture results, soil temperature slightly increased in 2011, while a huge difference in soil temperature was recorded in 2012. The application of simulated rain pulse decreased soil temperature. Between pulses, the temperature increased again until the application of the second pulse, which then reduced soil temperature to the same value during pulse 3 (32 ºC).

Site 4 featured the second control plot (S4A), and during the experiment on this plot the moisture content increased with time to a maximum of 0.22 and 0.24 cm^3 cm^{-3} for 2011 and 2012, respectively (Figure 7.4c). There was a significant difference between the two pulses for each year but no significant difference was observed between pulses 1 and 3 (Figure 7.6c). Although soil temperature was initiated from the same value (28 ºC), the reduction in soil temperature for 2011 was higher by 3 ºC compared to the reduction noted for 2012 (0.5 ºC).

With incorporating 10 cm compost/mulch with soil surface, the soil moisture in S4B was kept the same during the two pulses in 2011. In 2012, the soil surface was very dry. Due to rain application, soil moisture increased sharply to a maximum value of 0.38 cm^3 cm^{-3}, decreased between pulses, and then increased again to 0.36 cm^3 cm^{-3} during pulse 4 (Figure 7.4d). No significant difference was noted in soil moisture between pulses 1 and 2. However, pulses 3 and 4 were significantly different between each other and the 2011 pulses (Figure 7.6d). Soil temperature had the same trend as soil moisture for plot S4B since only a very slight change in soil temperature was observed in 2011. However, more substantial changes were noted in 2012.

7.3.3 Flow rate and runoff

The runoff flow rate (mm min^{-1}) was calculated for each simulated rainfall pulse for all plots and is shown in Figures 7.7 and 7.8. Generally, the flow rate increased with time and reached steady state conditions during the application of simulated rain, then decreased after rainfall cessation. The p values resulting from ANOVA are given in Table 7.3.

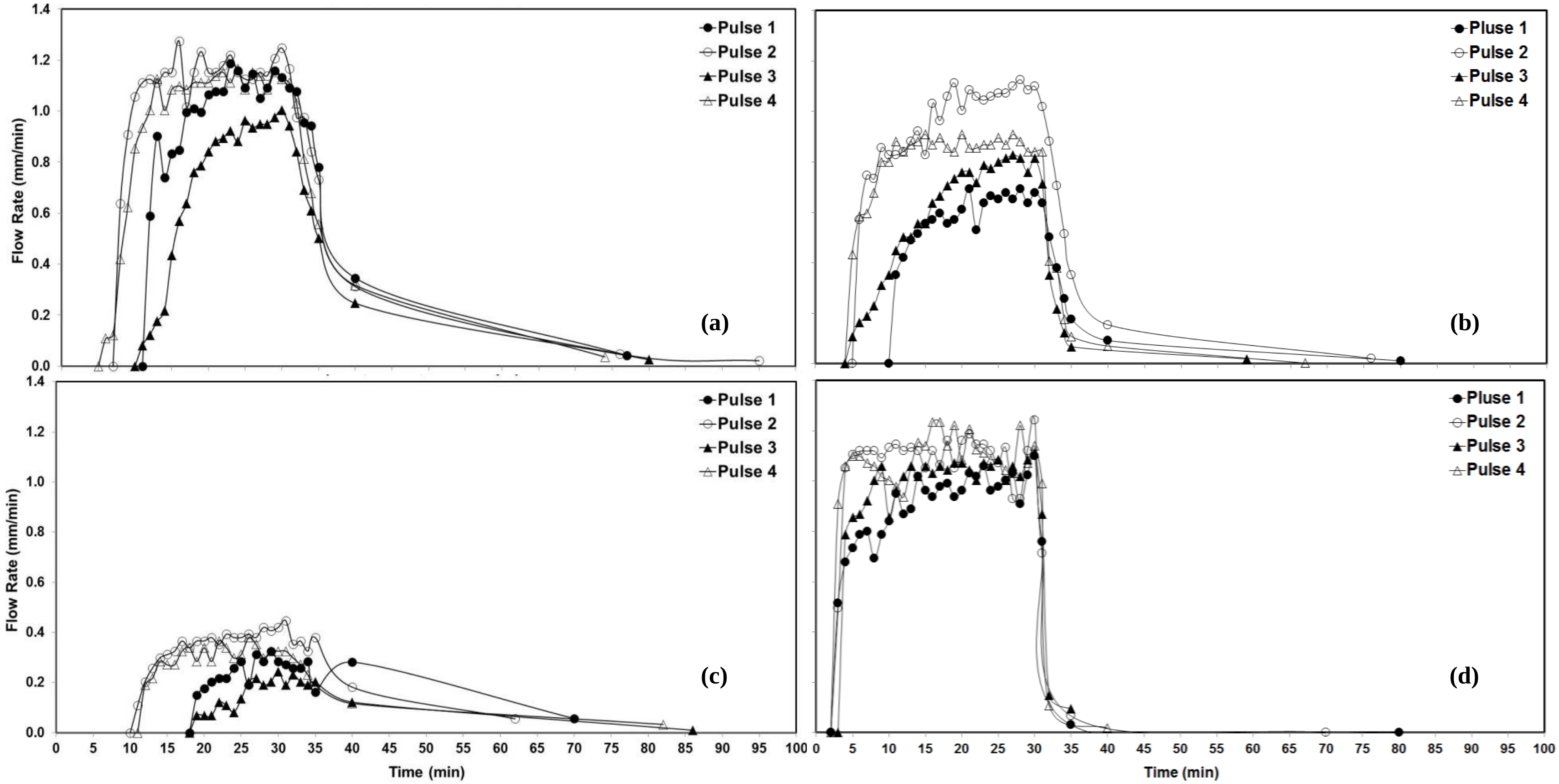

Figure 7.7. Flow rate (mm min^{-1}) for sites 1 and 2 as; (a) S1A, (b) S1B, (c) S2A, and (d) S2B. Pulses 1 and 2 were first and second rain pulses applied in 2011, respectively. Pulses 3 and 4 were the first and second rain pulses applied in 2012, respectively.

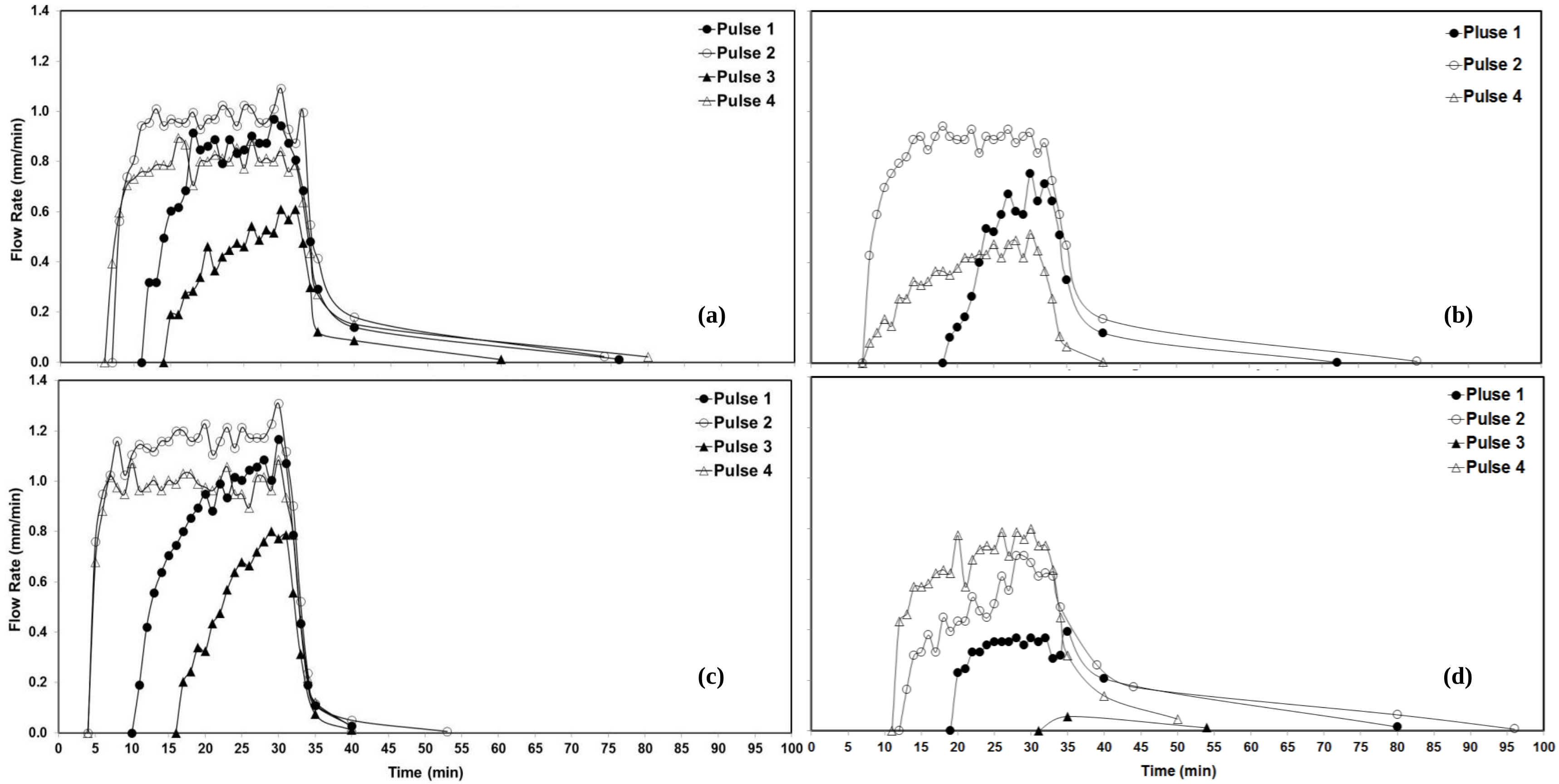

Figure 7.8. Flow rate (mm min^{-1}) for sites 3 and 4 as; (a) S3A, (b) S3B, (c) S4A, and (d) S4B. Pulses 1 and 2 were the first and second rain pulses applied in 2011, respectively. Pulses 3 and 4 were the first and second rain pulses applied in 2012, respectively.

Table 7.3. The p value (at significant level=0.05) resulting from ANOVA analysis to estimate the mean difference in flow rate between the applied rain pulses at each plot, Louisiana, USA.

Pulses		S1A	S1B	S2A	S2B	S3A	S3B	S4A	S4B
1	2	0.9094	<0.0001*	0.0041*	0.2038	0.0704	<0.0001*	0.0477*	0.1301
1	3	0.0295*	0.9965	0.0388*	0.6135	0.0007*	---	0.0408*	0.0119*
2	4	0.6881	0.1219	0.1972	0.9957	0.1033	<0.0001*	0.5558	0.0128*
3	4	0.0363*	0.0415*	<0.0001*	0.7707	0.0001*	---	0.001*	<0.0001*

* p is significant at (α) = 0.05

Commonly, the soil moisture values during the second pulses were higher compared to the first pulses as the soils were nearly saturated during the second pulses. Additionally, in 2012 the soils were likely drier compared to the soil water content values in 2011 especially for the sites 2, 3, and 4 along IH-49.

Cumulative runoff (as a percentage of the total applied rain) was calculated based on the amount and intensity of applied rain (Table 7.4). Runoff (%) generally increased for the control plots and decreased due to the compost/mulch application. Light tillage with compost/mulch application increased soil roughness and decreased the total runoff. These findings were consistent with Greene et al. (1994) and Freebairn and Gupta (1990). Based on the compost/mulch thickness (0-, 5-, and 10 cm), the plots were generally classed into three groups. Plots S2B and S4A were control; plots S1B, S3A, and S3B received 5 cm compost/mulch; and plots S1A, S2A, and S4B treated with 10 cm compost/mulch.

7.3.3.1 Control plots

Plot S2B, at 25% slope, was lightly tilled. Runoff was observed after only two to three minutes from simulated rainfall initiation for all pulses (Figure 7.7d). Flow rate increased rapidly when rain was applied to the maximum of 1.3 mm min^{-1}, and after 30 minutes it sharply decreased with time.

Table 7.4. Total runoff as a percentage from the applied rain, total P, turbidity, TSS and total soil loss at each pulse from each plot, Louisiana, USA.

Plot	Pulse	Simulated Rain l 30min^{-1}	Intensity mm hr^{-1}	Runoff %	Turbidity NTU	TSS mg l^{-1}	Total soil loss g	Total P mg l^{-1}
S1A	Pulse 1	249.81	74.69	74.17	96.54	111.27	14.29	1.74
	Pulse 2	257.38	76.96	89.99	65.34	65.28	12.59	1.45
	Pulse 3	283.88	84.88	46.90	255.75	277.85	27.62	1.02
	Pulse 4	283.88	84.88	73.53	187.82	205.74	33.51	0.89
S1B	Pulse 1	246.03	73.56	40.52	250.00	266.17	27.31	1.10
	Pulse 2	257.38	76.96	76.42	394.38	431.78	76.27	1.03
	Pulse 3	283.88	84.88	40.79	765.77	802.79	91.89	1.31
	Pulse 4	246.03	73.56	65.01	708.64	741.15	115.90	1.08
S2A	Pulse 1	253.60	75.82	19.46	110.95	139.16	4.00	1.97
	Pulse 2	264.95	79.22	27.53	112.89	96.75	5.18	1.68
	Pulse 3	264.95	79.22	9.66	65.43	81.58	0.97	0.45
	Pulse 4	264.95	79.22	23.35	18.86	30.97	1.82	0.44
S2B	Pulse 1	257.38	76.96	69.12	3696.36	5562.00	953.84	2.16
	Pulse 2	234.67	70.17	91.04	3762.27	5217.88	1269.39	2.17
	Pulse 3	264.95	79.22	73.38	1834.67	2214.94	458.12	0.87
	Pulse 4	242.24	72.43	89.92	1814.80	2197.09	489.19	0.83
S3A	Pulse 1	261.17	78.09	48.58	101.74	157.92	5.58	0.96
	Pulse 2	242.24	72.43	76.00	26.10	52.13	6.33	0.89
	Pulse 3	283.88	84.88	22.27	80.56	84.92	4.67	0.73
	Pulse 4	283.88	84.88	54.68	94.64	106.81	12.88	0.69
S3B	Pulse 1	246.03	73.56	24.52	118.00	249.19	8.43	1.23
	Pulse 2	246.03	73.56	65.73	96.59	131.67	11.12	1.10
	Pulse 3	272.52	81.48	0.03	882.21	882.21	0.08	1.31
	Pulse 4	283.88	84.88	21.92	525.20	556.94	37.29	1.18
S4A	Pulse 1	280.09	83.75	47.29	42.81	96.14	6.82	0.80
	Pulse 2	264.95	79.22	83.77	35.54	43.29	7.81	0.76
	Pulse 3	257.38	76.96	24.85	99.08	112.79	4.96	0.46
	Pulse 4	257.38	76.96	73.53	45.15	54.16	8.35	0.40
S4B	Pulse 1	264.95	79.22	17.94	36.16	119.27	2.68	0.87
	Pulse 2	227.10	67.90	45.26	36.07	56.96	4.43	0.90
	Pulse 3	257.38	76.96	1.37	98.70	110.63	0.33	1.04
	Pulse 4	272.52	81.48	41.64	30.95	41.63	4.22	0.97

The ANOVA results (Table 7.3) indicated that there were no significant differences in the flow rate between the four pulses ($P > 0.05$). Runoff percentage at Plot S2B was the highest

compared to the other plots; an average of 70% and 90% from applied rain were lost as runoff from first (1 and 3) and second (2 and 4) pulses, respectively (Table 7.4). Conversely, soil water content was the lowest as averages of 0.01 and 0.13 ($cm^3\ cm^{-3}$) were recorded from the first (1 and 3) and second (2 and 4) pulses, respectively (Table 7.2).

Plot S4A was non-tilled on a 10% slope. Runoff was observed after 10 and 16 minutes for pulses 1 and 3, respectively. The flow rate increased gradually to the highest value of 1.1 and 0.8 mm min^{-1}, respectively, and then sharply decreased with time when the rainfall simulator stopped (Figure 7.8c). Less flow rate was observed from pulse 3, which may be due to the reduction of soil moisture from 0.1 to 0.01 $cm^3\ cm^{-3}$ for pulse 1 and pulse 3, respectively. During pulses 2 and 4, the runoff was observed after only four minutes. The flow rate increased rapidly with the highest values of 1.3 and 1.1 mm min^{-1} for pulses 2 and 4, respectively, and after 30 minutes, the flow rate sharply decreased with time. The ANOVA results (Table 7.3) showed that there were significant differences in the flow rate between pulses 1 and 2, pulses 1 and 3, and pulses 3 and 4 since the p values were < 0.05. However, no significant difference was observed between pulses 2 and 4. Runoff percentage from S4A was still high compared to most compost/mulch treated plots but it was less than control plot S2B, owing to its lower slope. Runoff values of 47% and 25% were lost from pulses 1 and 3, respectively. While 83% and 74% were lost as runoff from pulses 2 and 4, respectively (Table 7.4). Although S2B and S4A were control plots, the higher slope in the S2B produced earlier runoff compared to S4A during pulses 1 and 3.Both control plots achieved virtually the same maximum of the flow rate value (1.3 mm min^{-1}), although the different in slope and tillage practices.

7.3.3.2 Adding 5 cm compost/mulch

Three plots were treated with 5 cm compost/mulch; S1B (light-tillage, 34% slope), S3A (no-tillage, 15% slope), and S3B (light-tillage, 15% slope). Generally, the compost/mulch

application reduced the flow rate relative to the control plots. For plot S1B, runoff was noted after ten and four minutes. The flow rate increased to 0.6 and 0.8 mm min^{-1} for pulses 1 and 3, respectively, and after 30 min, it decreased gradually with time (Figure 7.7b). During the second pulses (2 and 4), the runoff started after four minutes. The flow rate increased to maximum values of 1.1 and 0.9 mm min^{-1} for pulses 2 and 4, respectively, then sharply decreased with time. The ANOVA results (Table 7.3) showed that there were significant differences in the flow rate between pulses 1 and 2, and pulses 3 and 4 with p values <0.05. However, no significant difference was observed between pulses 1and 3, and pulses 2 and 4. Pulses 1 and 3 both lost 41% of applied rainfall to runoff, while pulses 3 and 4 had 76% and 65% runoff, respectively (Table 7.4). Additionally, the soil water content was nearly the same for pulses 1 and 3 with a value of 0.1 cm^3 cm^{-3}, while for the second pulses (2 and 4) the average water content was 0.25 cm^3 cm^{-3}.

In plot S3A, pulse 1 and 3 had the same trend since the flow rate gradually increased temporally to around 0.9 and 0.5 mm min^{-1} for the pulses 1 and 3, respectively, and after 30 min, it decreased gradually with time. However, runoff was noted after 10 and 14 minutes for pulses 1 and 3, respectively (Figure 7.8a). During the second pulses (2 and 4), runoff started after six minutes. The flow rate sharply increased to maximum values of 1.0 and 0.9 mm min^{-1} for pulses 2 and 4, respectively, and after 30 minutes, the flow rate sharply decreased. The ANOVA results (Table 7.3) showed that there were significant differences in the flow rate between pulses 1 and 3, and pulses 3 and 4. However, between pulses 1 and 2, and pulses 2 and 4, no significant difference was noted. Runoff percentages were different between the four pulses; less for pulses 2 and 4 compared to pulses 1 and 2. A malfunction was observed in the soil moisture sensor during 2012, so no moisture data is available for that time.

Plot S3B generated runoff after 18 minutes for pulse 1, but there was no runoff for pulse 3. For pulse 1, the flow rate increased to 0.7 mm min^{-1}, then after 30 min it decreased gradually

with time. During pulse 2, the runoff started after six minutes, flow rate increased to a maximum value of 0.9 mm min^{-1}, and after 30 minutes it sharply decreased. Flow rate for pulse 4 slowly increased to a maximum of 0.5 mm min^{-1} then gradually decreased with time after 30 minutes (Figure 7.8b). The ANOVA results (Table 7.3) showed strongly significant differences in the flow rate between pulses 1 and 2, and pulses 2 and 4 ($p < 0.0001$). Runoff percentage was very low for pulse 4 (22%), and soil water content was very low (Table 7.2).

7.3.3.3 Adding 10 cm compost/mulch

The application of 10 cm of compost/mulch occurred on three plots; S1A (non-tilled, 34% slope), S2A (non-tilled, 25% slope), and S4B (lightly-tilled, 10% slope). Generally, with a higher thickness of compost/mulch application the flow rate and runoff were significantly reduced compared to control and 5cm-treated plots. Conversely, plot S1A had higher flow rate and higher runoff (%) compared to the 5 cm-treated plot as this area was under active construction during 2011 which reduced the effectiveness of the applied compost/mulch (Figure 7.7a). In 2012, while conducting pulses 3 and 4, the runoff started after nine and five minutes. The flow rate increased to 0.9 and 1.1 mm min^{-1}, respectively, and after 30 minutes, the flow rate gradually decreased. The ANOVA results (Table 7.3) showed that there were significant differences in the flow rate between pulses 1 and 3, and pulses 3 and 4, but, no difference was observed between pulses 1 and 2, and pulses 2 and 4. These results were confirmed by the soil water content and runoff results (Tables 7.2 and 7.4).

In plot S2A, the highest level of soil conservation was achieved since the thick layer of compost/mulch drastically decreased the flow rate and runoff from this plot (Table 7.4). In pulses 1 and 3, the flow rate gradually increased with time to 0.3 and 0.2 mm min^{-1}, respectively and runoff was noted after 18 minutes. After 30 minutes, the flow rate decreased very gently with time. During the second pulses (2 and 4), runoff started after 10 and 11 minutes, respectively.

The highest flow rate was 0.4 mm min^{-1}, and after 30 minutes, the flow rate gradually decreased (Figure 7.7c). The ANOVA results (Table 7.3) showed significant differences in the flow rate between pulses 1 and 2, pulses 1 and 3, and pulses 3 and 4, however, no significant difference was observed between pulses 2 and 4. Runoff results were perfectly matched with the water content values (Table 7.2); with lower water content, lower runoff was generated and vice versa.

For plot S4B, almost no flow was observed during pulse 3 as a maximum of 0.06 mm min^{-1} was recorded and runoff was noted after stopping the rainfall simulator (31 minutes). In pulse 1, the runoff started after 19 minutes and the flow rate increased temporally to 0.4 mm min^{-1}, then decreased with time. In pulses 2 and 4, the runoff started after 12 and 11 minutes, respectively. The flow rate increased to maximum values of 0.7 and 0.8 mm min^{-1}, respectively, then after 30 minutes, gradually decreased (Figure 7.8d). The ANOVA results (Table 7.3) showed that there were significant differences in the flow rate between pulses 1 and 3, pulses 2 and 4, and pulses 3 and 4, but, no significant difference was observed between pulses 1 and 2. The runoff percentage was only 1% for pulse 3, which correlated perfectly with very low water content of 0.03 cm^3 cm^{-3} (Tables 7.2 and 7.4).

The runoff results in this study support the findings of Naslas *et al*. (1994). They used the modular system described by Guerrant *et al*. (1990) to evaluate runoff and erosion as influenced by different soil types, slopes and soil cover. They concluded that greater amounts of runoff and erosion occurring with greater slope, and less runoff yet increased erosion with increased plot disturbance. The rainfall simulation experiment results in the current study showed that the compost/mulch coverage increased surface roughness, delayed runoff generation, and reduced the runoff percentage and soil loss compared to the control plots. These results are consistent with the results from Jordán *et al*. (2010).

7.3.4 Total suspended solid and soil loss

The total suspended solids (TSS) concentrations in the runoff samples from each pulse in each plot with time are presented in Figures 7.9 and 7.10. Generally, TSS started with the highest concentration, in most cases, and then the concentration diminished with time to the end of each pulse. Additionally, the untilled plots tended to have higher TSS for the first pulses (1 and 3) compared to second pulses (2 and 4). Conversely, in the lightly tilled plots the second pulses exhibited higher TSS than the first pulses. Furthermore, tillage practices caused more soil disturbance and led to higher TSS concentrations in the tilled plots compared to those untilled. Also, the untilled plots had a homogenous pattern as the TSS concentration started high then gradually decreased with time. Conversely, the lightly tilled plots had a random trend especially with tilled control plot S2B (Figure 7.9d).

To study the importance of TSS and how it affected water quality, the relationship between TSS and the total amount of soil loss from each plot was investigated. Table 7.4 shows the average of TSS (mg l^{-1}) and turbidity (nephelometric turbidity unit, NTU), and the total soil loss (g) for each applied pulse at each plot. The results indicated that there was a strong relationship between TSS and turbidity values. Based on the current study data set a simple linear regression between turbidity and TSS can be expressed as TSS= 1.38057 X (Turbidity) - 8.58251; r^2 = 0.9858; and $p<0.0001$ at 0.05 significant level.

Figure 7.11 explains the effects of applying different thicknesses of compost/mulch (0-, 5-, and 10cm) on TSS in runoff and the amount of soil lost due to the simulated rain pulses. Generally, the control plot (S2B) had the highest TSS and the most erodibility by losing over 1000 g of soil during the application of 30 minutes of simulated rain (Figure 7.11a). This plot represented the worst case scenario as it had a steep slope of 25%, was lightly tilled, and did not receive any erosion control treatment. Another control plot, S4A, had a maximum of 112 mg l^{-1}

TSS and <10 g soil loss. This massive difference between the two control plots is likely due to the difference in slope, tillage practice, and inherent soil properties (e.g. texture) that affected the erodibility of the soil. Plot S4A, kept untilled, had the lowest slope of 10%, and had the highest sand percentage.

Adding 5 cm compost/mulch on the soil surface led to a 6- and 10- fold reduction in TSS and soil loss, respectively. Compared to the tilled plots (S1B and S3B), untilled plot S3A exhibited the best results as TSS concentration and the amount of soil loss were limited to 157 mg l^{-1} and 12 g, respectively (Figure 7. 11b). Although 10 cm compost/mulch was a double thickness of the previous treatment (5 cm), the reduction in TSS and soil loss was not the same when compared 5 cm vs. control. Approximately a 3-fold reduction in TSS and soil loss was observed when applying 10 cm compost/mulch compared to the 5 cm treatment.

Although, plot S4B was treated with 10 cm composted/mulch and incorporated in the soil by tillage, the value of TSS and soil loss was relatively low compared to the untilled plots, especially plot S1A. This may due to the low slope for plot S4B (10%) compared to the steeper slopes of S1A (34%) or S2A (25%). When comparing the two untilled plots (S1A and S2A), S1A exhibited higher TSS and soil loss as a result of construction and post construction activities that were taking place in that location during the experiment (Figure 7.11c). The runoff and soil loss results obtained from the current study were consistent with other findings reported by Adekalu et al. (2006).

ANOVA analysis was conducted on the soil loss calculated from the runoff collected from each pulse in each plot. The results are displayed in boxplot format in Figures 7.12 and 7.13. The results revealed that mostly there were no significant differences in the soil loss between each pair of consecutive rain pulses (1 and 2; 2 and 3), or between the same pulses for different years (1 and 3; 2 and 4).

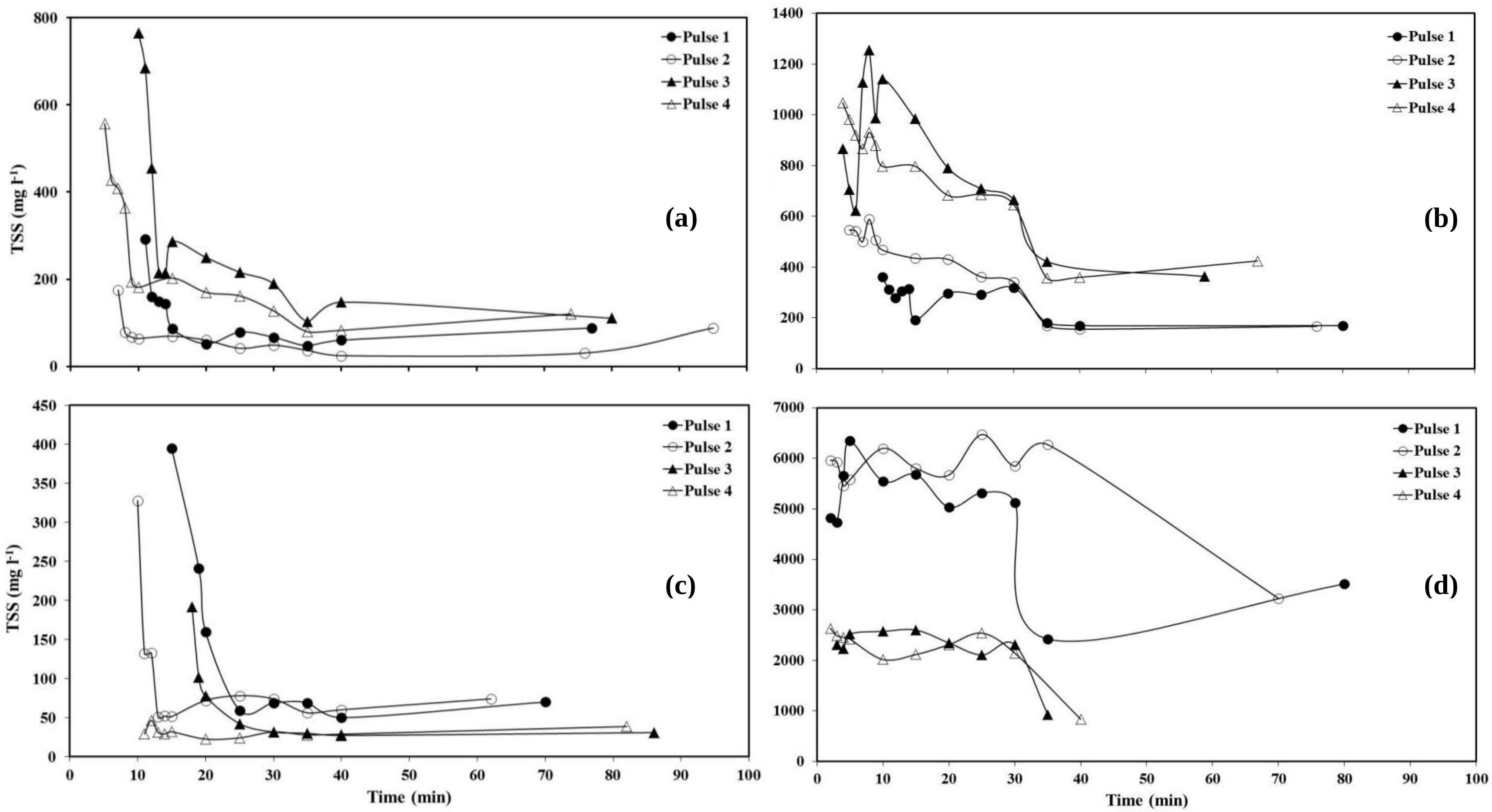

Figure 7.9. Total suspended solid (mg l^{-1}) with time for sites 1 and 2 as; (a) S1A, (b) S1B, (c) S2A, and (d) S2B. Pulses 1 and 2 were the first and second rain pulses applied in 2011, respectively. Pulses 3 and 4 were first and second rain pulses applied in 2012, respectively.

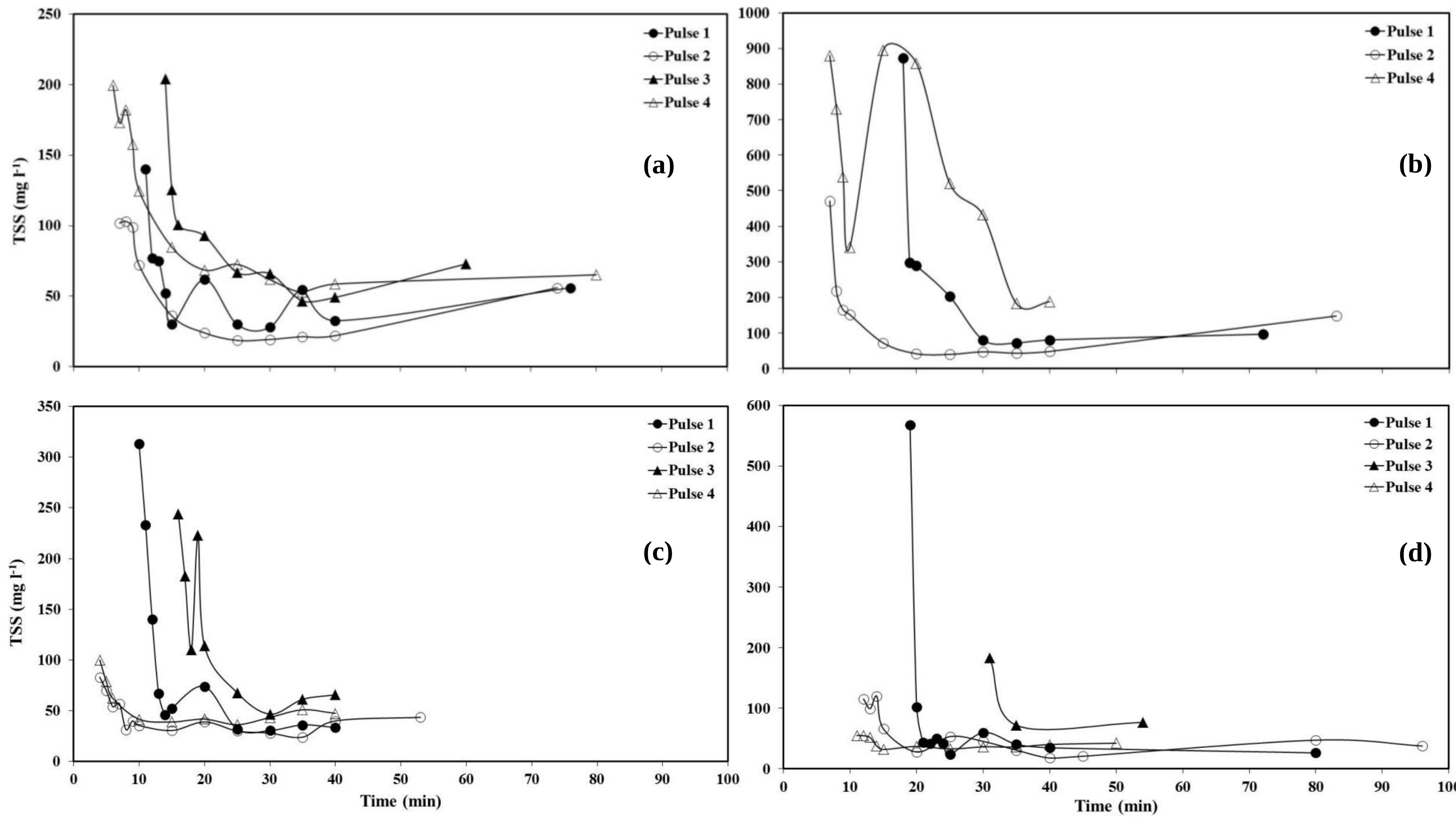

Figure 7.10. Total suspended solid (mg l^{-1}) with time for site 3 and site 4 as; (a) S3A, (b) S3B, (c) S4A, and (d) S4B. Pulse 1 and pulse 2 were first and second rain pulses applied in 2011, respectively. Pulse 3 and pulse 4 were first and second rain pulses applied in 2012, respectively.

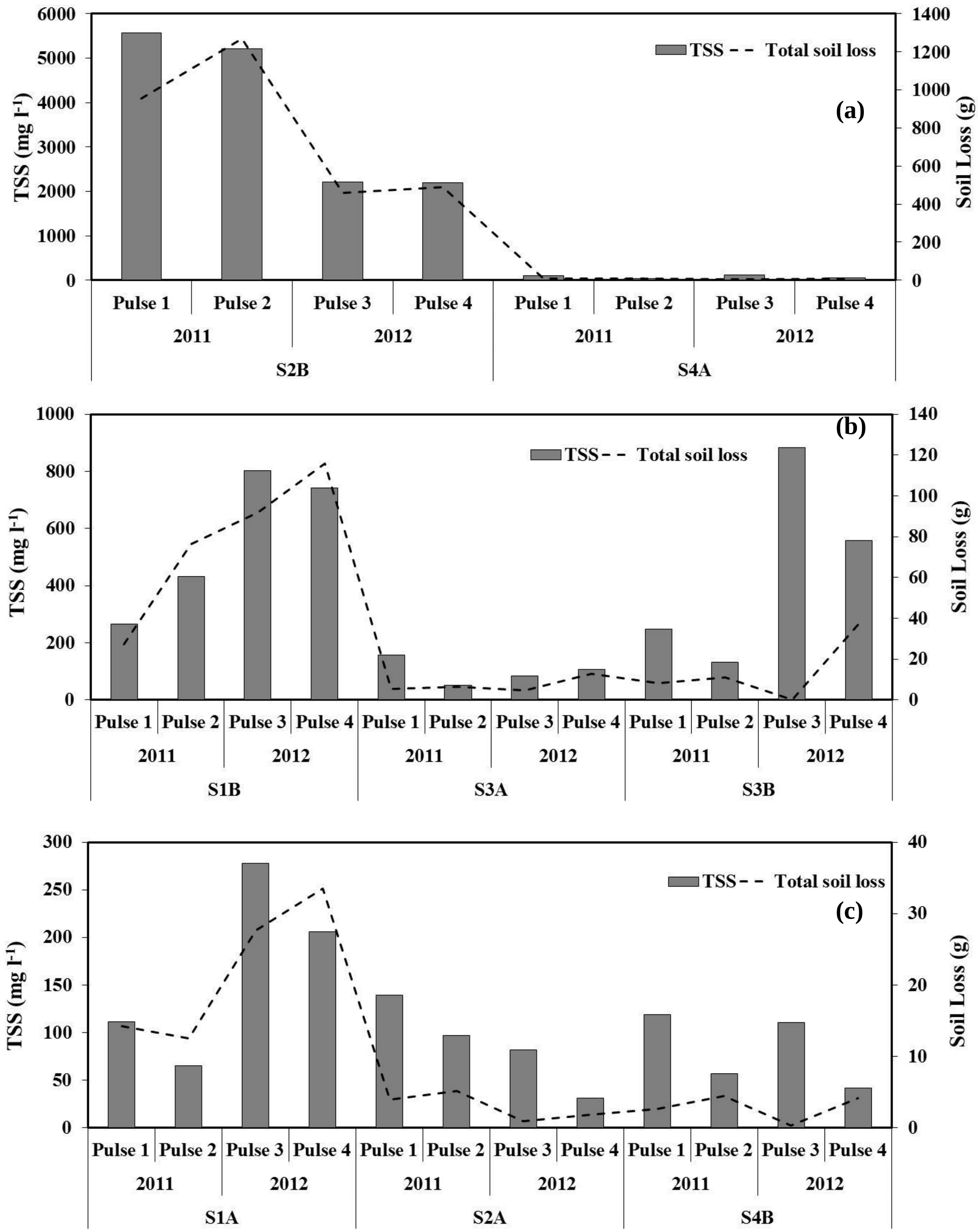

Figure 7.11 Effects of applying different thicknesses of compost/mulch on TSS in runoff and the amount of soil lost due to the simulated rain pulses. (a) Control plots, (b) 5 cm compost/mulch application, and (c) 10 cm compost/mulch application.

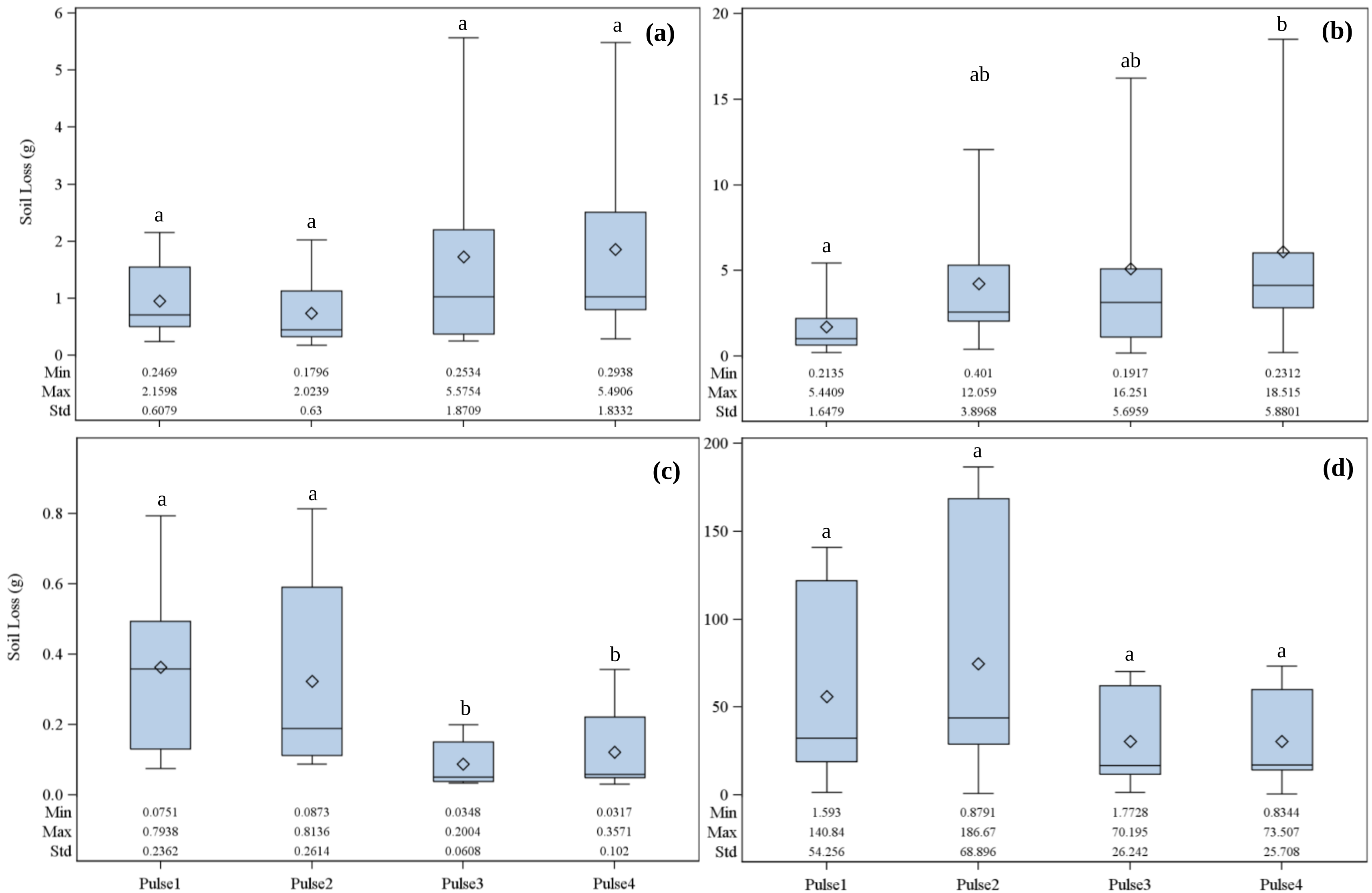

Figure 7.12 Soil loss boxplot, pulses 1 and 2 applied in 2011, and pulses 3 and 4 applied in 2012. Pulses with the same letter (a, b, c, or d) indicate no significant differences between them at α=0.05. (a) S1A, (b) S1B, (c) S2A, and (d) S2B.

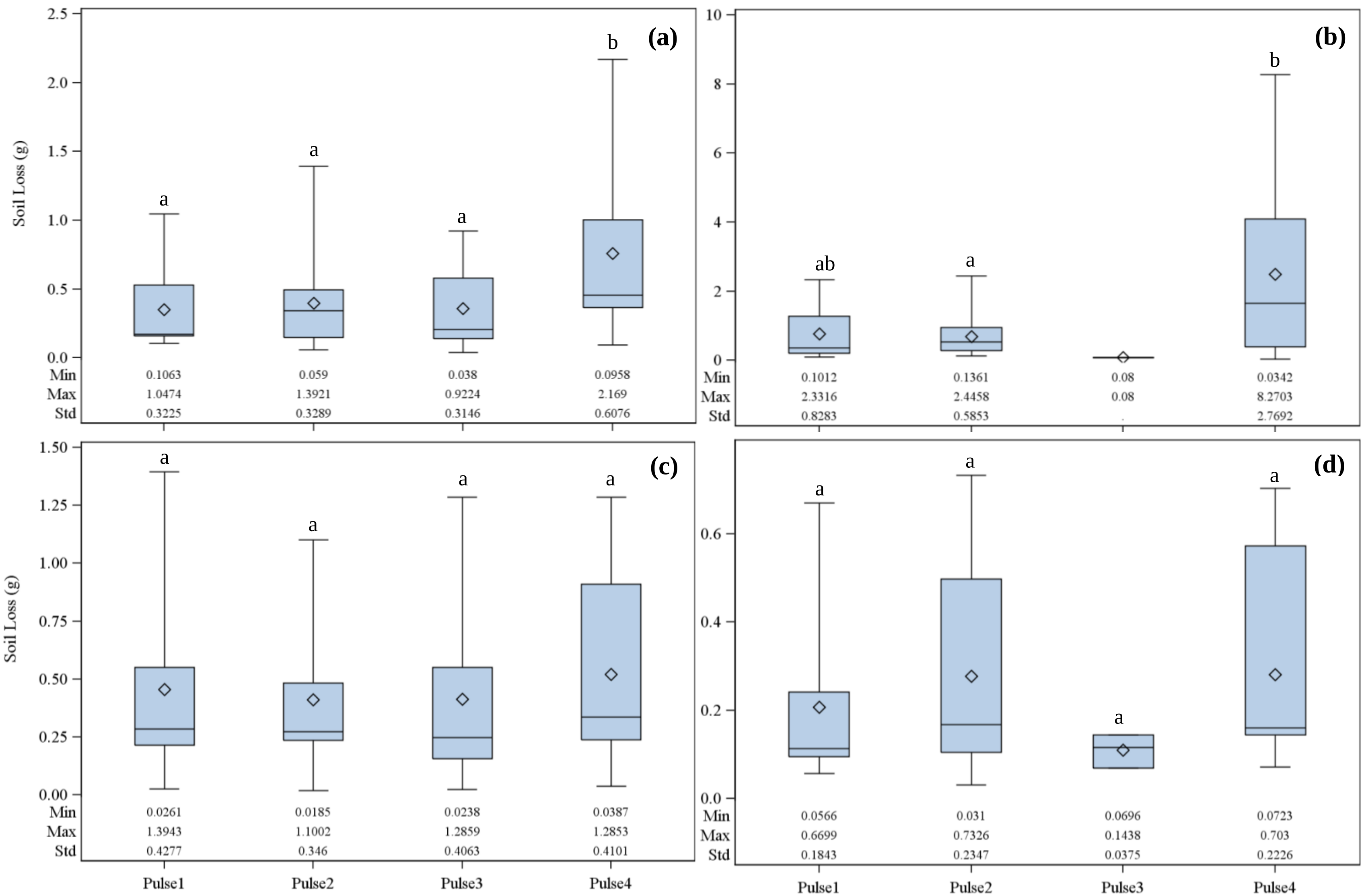

Figure 7.13 Soil loss boxplot, pulses 1 and 2 applied in 2011, and pulses 3 and 4 applied in 2012. Pulses with the same letter (a, b, c, or d) indicate no significant differences between them at α=0.05. (a) S3A, (b) S3B, (c) S4A, and (d) S4B.

7.3.5 Phosphorous in the runoff

The available P in the runoff, from all pulses in all studied plots, was very low with a mean value of 0.3 mg l^{-1}, which is consistent with the stated value from biosolids-amended soils (Sharpley, 1995). Given that the dissolved P is related to the amount of soil P that was extracted by Mehlich 3 (Sharpley, 1995), the low soil P (Table 7.1) resulted in low dissolved P in runoff.

Consequently, the total P was measured after digesting the unfiltered runoff samples from each pulse in 2011 and 2012. The average concentrations of total P from each pulse in each plot are presented in Table 7.4.

The total P results revealed that the concentration ranged from 0.4 to 2.2 mg l^{-1}, with higher values found in control plot S2B. Another control plot (S4A), had a lower value as the soil P was already low for this plot. The results indicated that there were no significant differences in the total P, between pulses or between plots, due to compost/mulch application, tillage practices, or different slope percentage. Phosphorous is often added to soils via agricultural fertilizer applications. However, since the location of the study sites were not near any agricultural activities or fertilization processes, there was no source of P added to the soil. These explained why the P content in soil and runoff samples was so low.

7.4 CONCLUSIONS

This study presents the comprehensive results of a rainfall simulation experiment that was conducted twice in two different years over eight plots on Louisiana roadsides treated with different compost/mulch thicknesses (0-, 5-, and 10 cm). Results showed very low P inherently in the studied soils, so the dissolved and total P measured in the runoff samples were consequently low. Variable slope and tillage practices on the plots influenced the effectiveness of compost/mulch as a soil erosion retardant. Construction activities also decreased the effectiveness of compost/mulch application by increasing TSS in the runoff and the amount of

soil lost by the application of simulated rain pulses. The best results were obtained using 10 cm compost/mulch with no-tillage, while 5 cm compost/mulch exhibited acceptable results. As such, 5 cm applications of compost/mulch may prove to provide the most effective cost/benefit ratio; ideal fodder for future study. To protect surface water quality from pervasive construction erosion, more caution should be taken during construction activities along roadsides. This study has confirmed that compost/mulch application can be successfully used as a BMP to reduce the erosion hazard on roadsides, though tillage incorporation of the material is not recommended.

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CHAPTER 8. CONCLUSIONS AND RECOMMENDATIONS

In this intensive research, the most advanced environmental techniques (remote sensing, geographic information system, and modeling) were used, besides field studies and laboratory analyses, to provide decision-makers with the most appropriate management practices for natural resource sustainability. Since the management practices are site specific, two studies were carried out throughout this research. The first was implemented in a semiarid region to assess agriculture sustainability. The second was conducted at the plot scale along roadsides in a humid region as a means of sustaining soil and water resources.

For the semiarid region, the *Bustan 3* area (341.27 km^2), western Nile delta, Egypt was chosen. This area represents a fragile agro-ecosystem which was targeted for reclamation in the 1990s. From 1984 to 2008, this area was radically changed from 100% barren land to 79% cultivated land. Using remote sensing, hybrid classification methods, and a vegetation index, this impressive change was successfully evaluated. The land cover maps produced from hybrid classification processes were more reliable and more accurate in detecting changes in land cover over this area compared to the vegetation index maps.

As the *Bustan 3* area is a fragile, vulnerable agro-ecosystem, high sensitivity to desertification exists. Using the land cover map from 1980 (while the *Bustan* 3 area was 100% barren land), and the land cover map from 2008 (when 79% of the area was cultivated), the most sensitive areas to desertification were identified. The Mediterranean Desertification and Land Use (*MEDALUS*) approach was used to evaluate the impacts of the reclamation processes that took place in the *Bustan 3* area on the sensitivity of the soil to desertification. With some adjustment in *MEDALUS*, such as adding new soil quality parameters and extending the quality parameters to include irrigation water quality index, the results were more reliable in assessing the most critically sensitive areas to desertification. The results of this study show that plant

cover, management, and irrigation water quality, which took place in 2008, dramatically impacted desertification. As irrigation water quality problems are more challenging, suitable management practices in such areas can improve the resilience of this fragile agro-ecosystem to desertification processes.

Besides the *MEDALUS* approach, land evaluation modeling was conducted for the *Bustan* 3 area to predict the land performance under specific use. With sufficient data and information related to soil, landscape, and climate, land capability analysis was predicted in the *Bustan 3* area. The Cervatana module of *MicroLEIS* software was used to predicate the general land use capability for possible agricultural uses. As geographic information systems have the ability to deal with attribute data and interpolate terrain attributes tables to spatially cover the entire area of interest, it significantly improved spatial data handling and analysis. In this study, land capability was examined for highly fragile soils in *Bustan* 3 area, Egypt. The main objective was to evaluate current soil resources and generate a wide range of possible scenarios based on different management practices to enhance agriculture sustainability. The land capability results concluded that 70% of the *Bustan 3* area, Egypt, had a good capability for agriculture production, while the three suggested management scenarios could increase the "good" capability class to cover up to 96% of the area.

Anthropogenic effects on the *Bustan* 3 area resulted in a massive change from desert soil to highly productive agricultural soil which could positively or negatively affect the environment. A coupling between remote sensing, geographic information systems, and modeling improved land use planning in this area, and could enhance the decision-making process especially in newly reclaimed areas in arid and semiarid regions. Additionally, land cover/use monitoring over long periods of time provided valuable information and is highly recommended for proper land use planning as well as sustainable development.

As agriculture sustainability was investigated in this research, sustainable natural resources on the roadsides were also considered. Roadside soils are often associated with slope and found in highly disturbed areas, which consequently increased the water erosion hazard especially in humid regions. Contrary to arid and semiarid regions, humid regions such as Louisiana, USA, are characterized by intense rainfall. Since the soils of roadsides are different in their properties and their response to management practices, four different sites were chosen on two different highways in Louisiana. With agreement in the literature about using the plot scale to study water erosion and soil loss, eight plots were constructed at the four study sites (two plots per site, side-by-side). The target plots were used to study the effects of compost/mulch thickness, tillage practices, slope, and construction activities on soil erosion and water quality for the runoff. Compost/mulch was used as a best management practice (BMP) to control soil erosion and water sedimentation. When the effect of compost/mulch on the soil properties was studied, the results confirmed that the compost/mulch was highly effective in improving surface soil water content and moderating surface soil temperature compared to the bare soils. Additionally, as a result of adding organic materials to the soil surface by using compost/mulch, the soil organic matter content of roadside soils was enhanced.

Besides the effects of compost/mulch on soil properties, it also affected surface water quality. The effect of compost/mulch cover in reducing runoff, total suspended solid (TSS), and turbidity from soils susceptible to high-intensity storm water in Louisiana was confirmed. The TSS and associated turbidity are the most problematic imparities in Louisiana surface water. Sediments transported from the roadsides are considered a common nonpoint source of water pollution in Louisiana. Results showed that TSS values were significantly decreased when compost/mulch was applied on the soil surface (70% and 74% reduction in TSS for the 5 cm and 10 cm compost/mulch applications, respectively).

Furthermore, compost/mulch also moderated runoff flow rate. Specifically, the application of 5- and 10-cm compost/mulch significantly reduced the flow rate. Based on results from a rainfall simulation experiment, 10 cm compost/mulch with no-tillage application exhibited the best results while 5 cm compost/mulch produced acceptable results. The total runoff volume from compost/mulch treated plots was significantly reduced compared to control plots.

As the effects of construction activities were evaluated in the current study, the results showed that active construction areas deserve more attention for conservation strategies since those areas are more exposed to erosion hazards via soil disturbance. The effectiveness of compost/mulch was significantly reduced and TSS and runoff were increased in the active construction area even with 10 cm compost/mulch application. Also the slope and tillage practices influenced the effectiveness of compost/mulch as an erosion retardant by increasing the flow rate and sediment losses compared to no tillage. The results of this study raised the importance of understanding the linkages between inherent soil properties, rainfall, and soil erosion in order to improve land use planning and identify better sustainable management practices.

From the current study results, some final recommendations should be considered:

1. Sustainable natural resource management planning is complex process requiring intensive information related to soil, water, climate, and other environmental parameters that could help the decision maker to select the best management plan.
2. For the purpose of achieving sustainable agricultural production, knowledge of past and current land use/land cover are essential for predicting future sustainability plans which could conserve land functionality as well as increase land productivity.

3. Sustainable management can be applied gradually by giving attention to the most critically sensitive areas first (subject to faster degradation), followed by less sensitive areas.
4. For roadside sustainability, 10 cm of compost/mulch coverage proved superior in reducing the total flow and TSS. However, 5 cm of compost/mulch application may strike the most economical balance between benefits received and cost of best management practice implementation.
5. To protect surface water quality from pervasive construction erosion, more caution should be taken during construction activities along roadsides.
6. Based on current study results, tillage incorporation of compost/mulch into the soil surface is not recommended, since it caused more disturbance to the soil surface and increased erosion rates.
7. Compost/mulch coverage is recommended as a best management practice in both active construction areas and established areas prone to soil erosion on roadsides.

APPENDIX A

PLOT CONSTRUCTION AND EQUIPMENT INSTALLATION

Figure A.1. The chosen site at roadsides of IH-49, Rapids Parish, Louisiana, USA.

Figure A.2. Plot preperation.

Figure A.3. Adding heavy steel edging.

Figure A.4. Adding compost/mulch treatments.

Figure A.5. ISCO auto-sampler unit with energy source and rain guage.

Figure A.6. H-flumes, 0.305 m depth.

Figure A.7. Data downloading from ISCO.

Figure A.8. HOBO® Micro Station (H21-002) Data Logging.

Figure A.9. Soil Moisture Smart Sensors (S-SMx-M005).

Figure A.10. 12-Bit Temperature Smart Sensors
(S-TMP-M006).

Figure A.11. Data Downloading from HOBO® Micro Station.

APPENDIX B

FLOW RATE FROM ISCO AUTO-SAMPLER

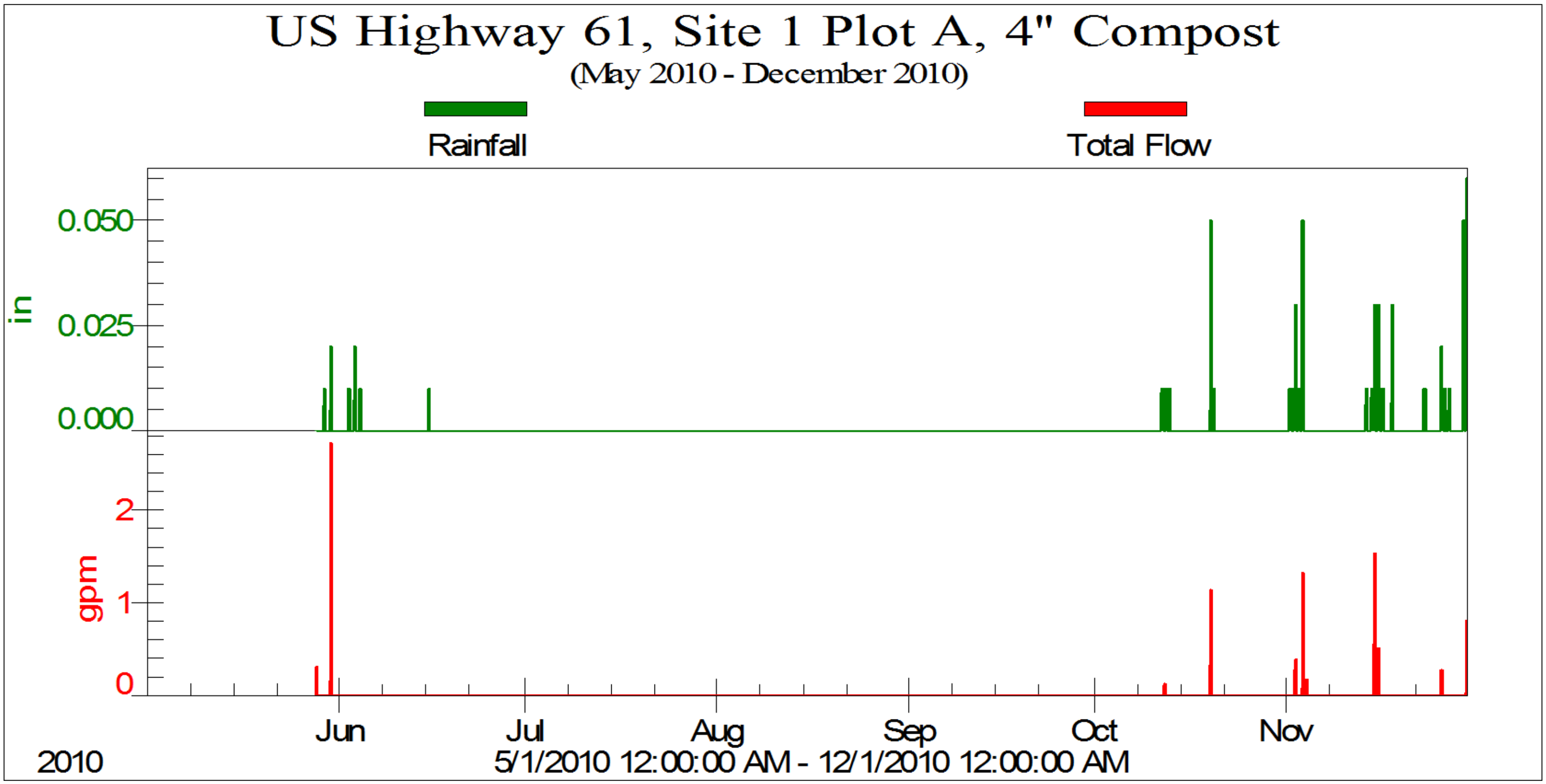

Figure B.1. US Highway 61, West Feliciana Parish, Site 1 Plot A, 4” compost/mulch application with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

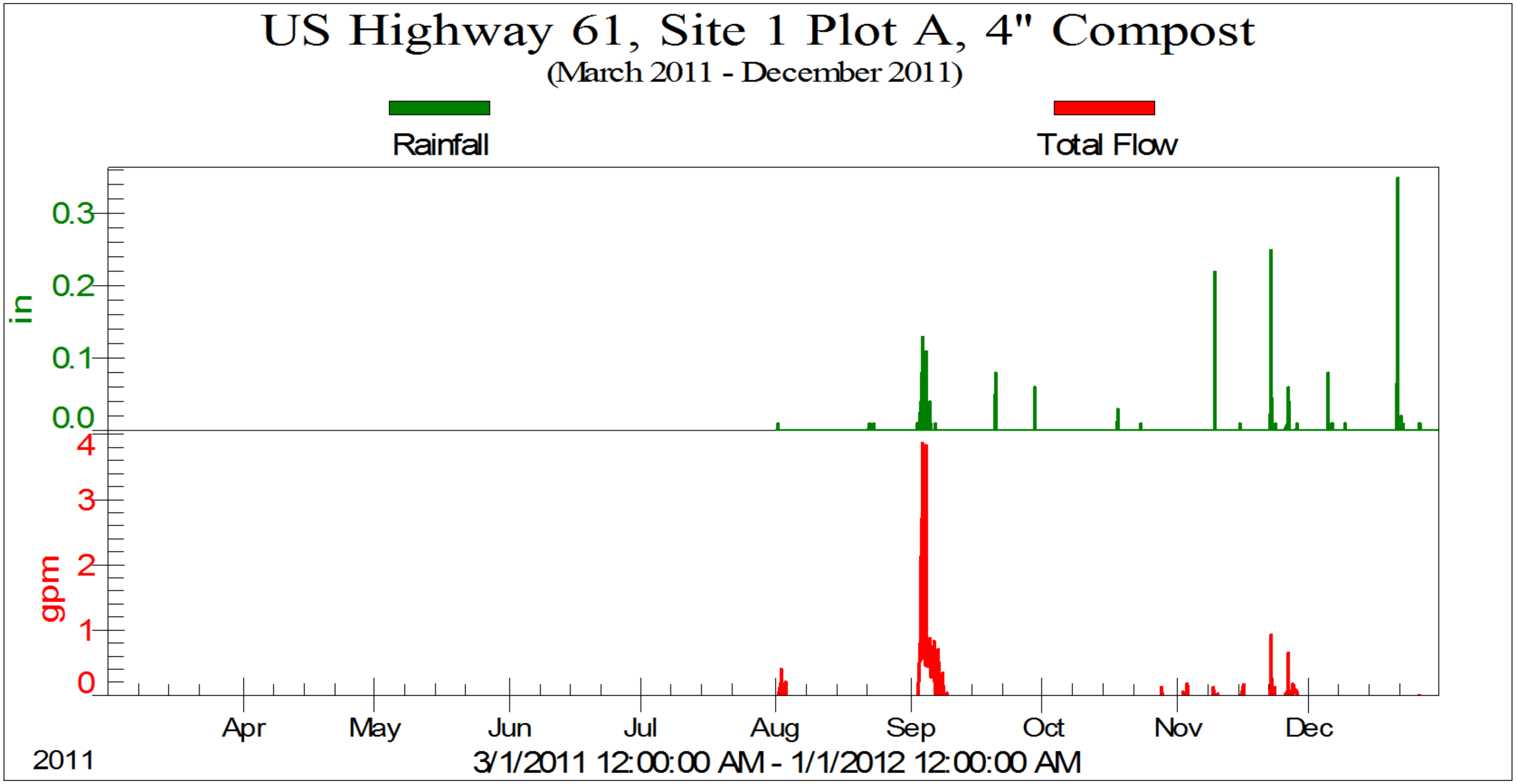

Figure B.2. US Highway 61, West Feliciana Parish, Site 1 Plot A, 4" composted mulch application with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

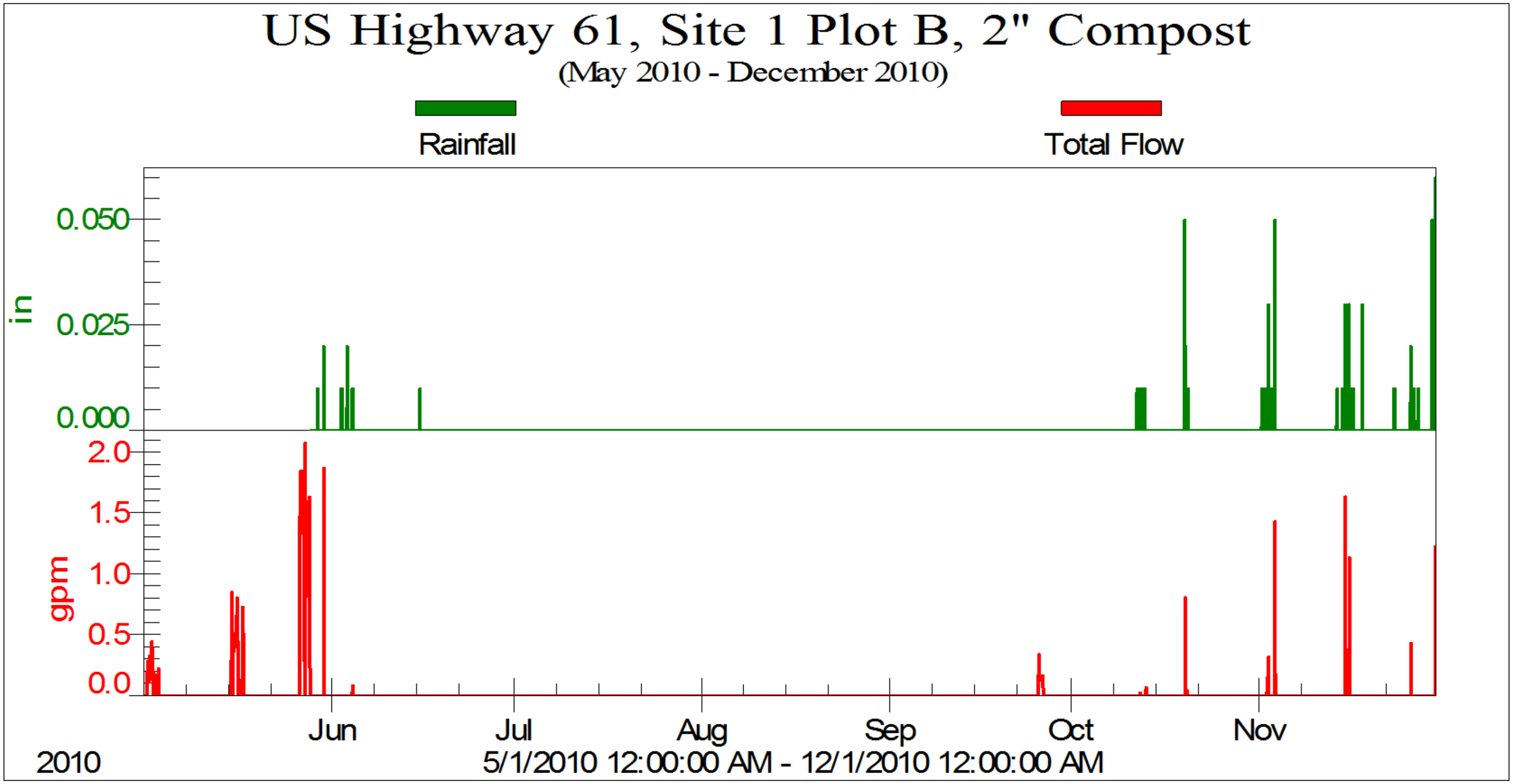

Figure B.3. US Highway 61, West Feliciana Parish, Site 1 Plot B, 2" composted mulch application with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

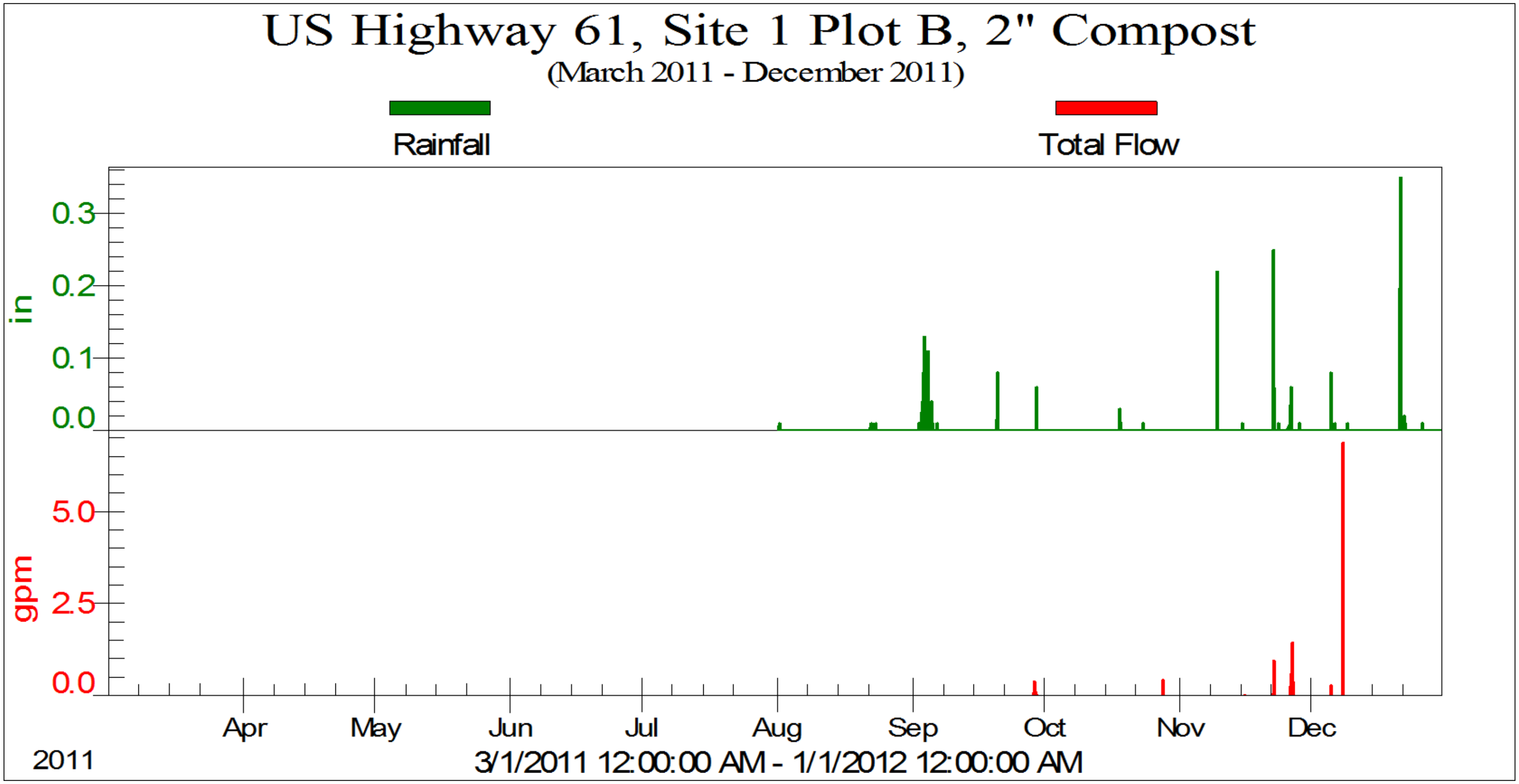

Figure B.4. US Highway 61, West Feliciana Parish, Site 1 Plot B, 2" composted mulch application with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

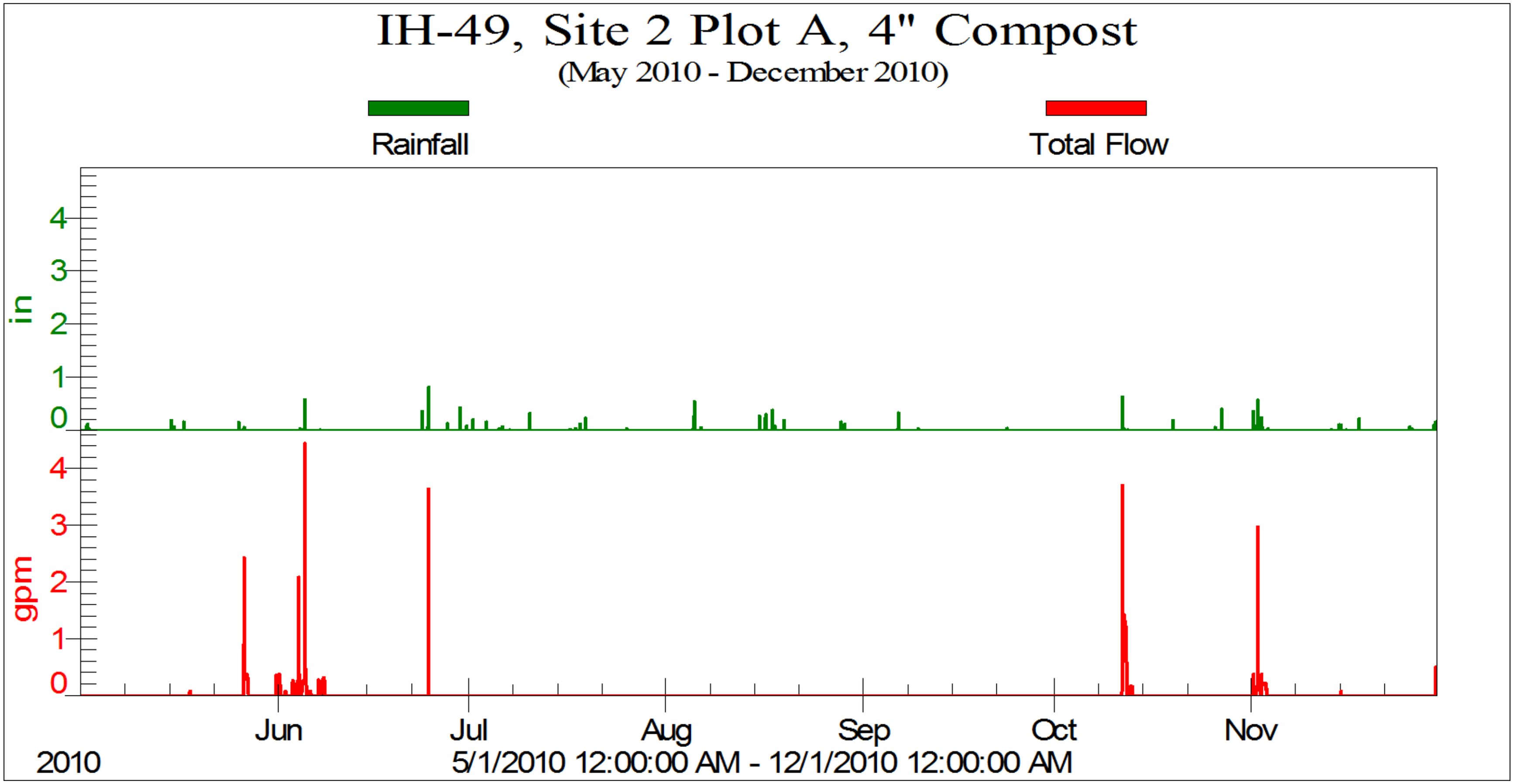

Figure B.5. IH-49, Rapids Parish, Site 2 Plot A, 4” composted mulch application with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

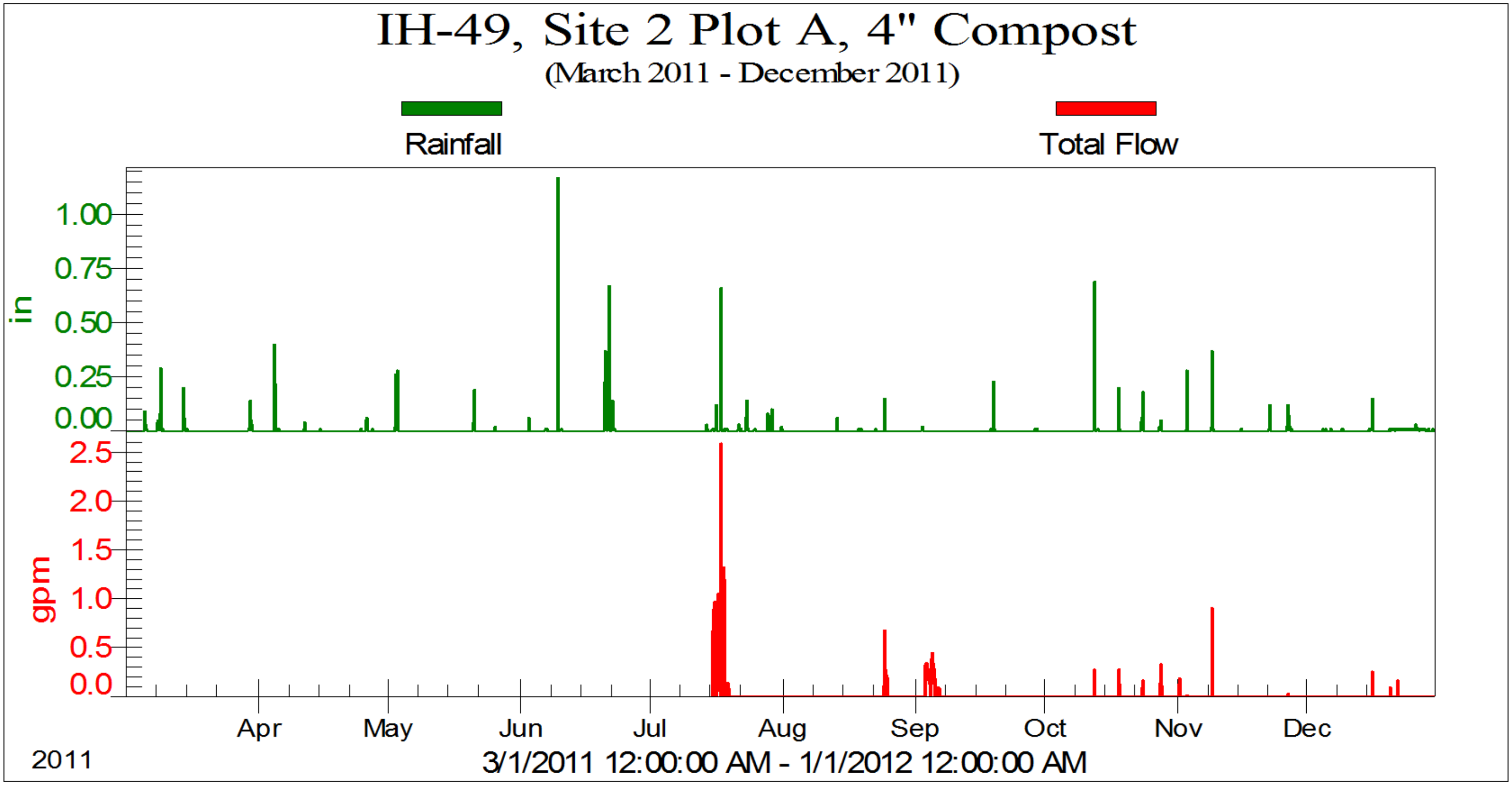

Figure B.6. IH-49, Rapids Parish, Site 2 Plot A, 4" composted mulch application with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

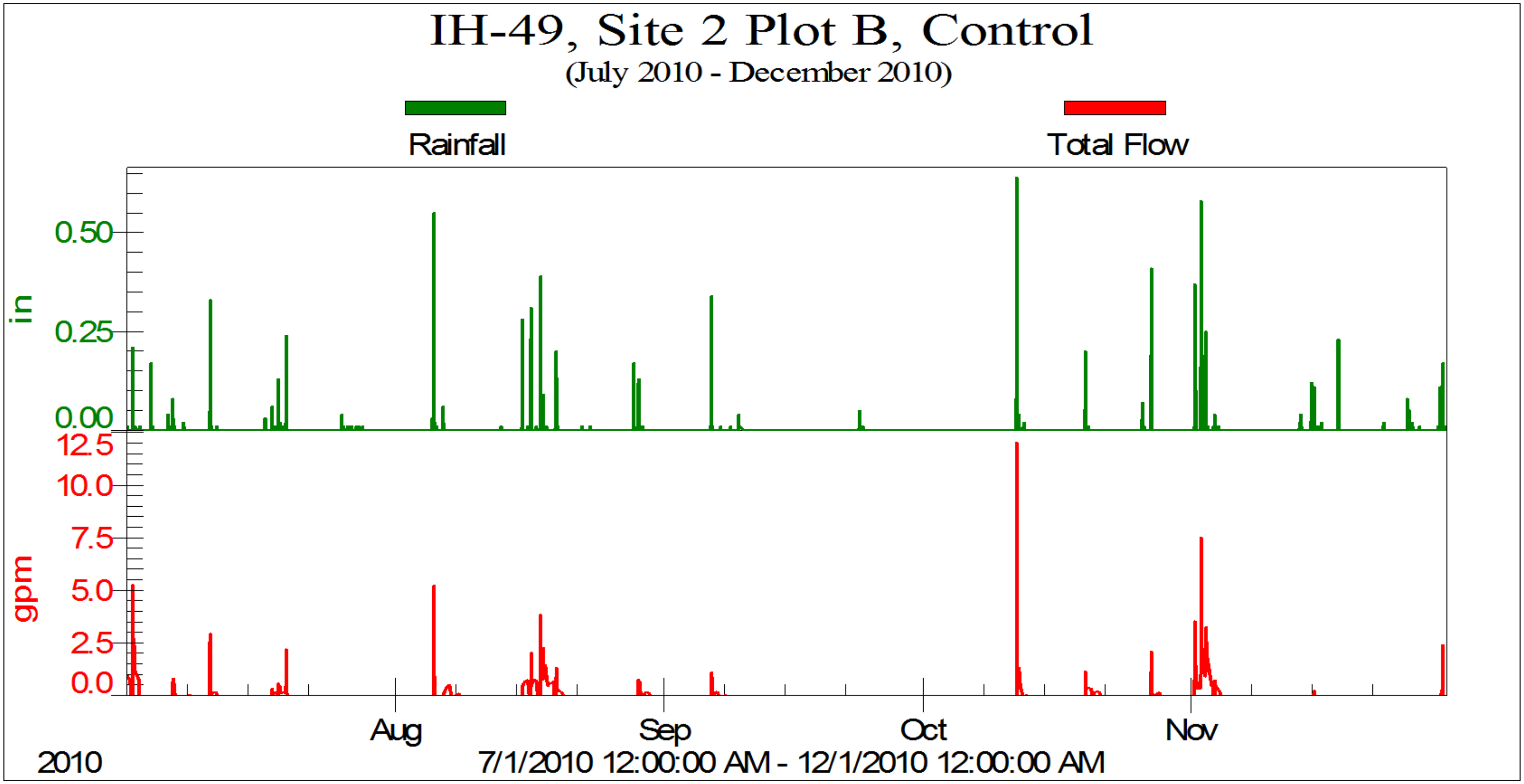

Figure B.7. IH-49, Rapids Parish, Site 2 Plot B, control with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from July 2010 to December 2010.

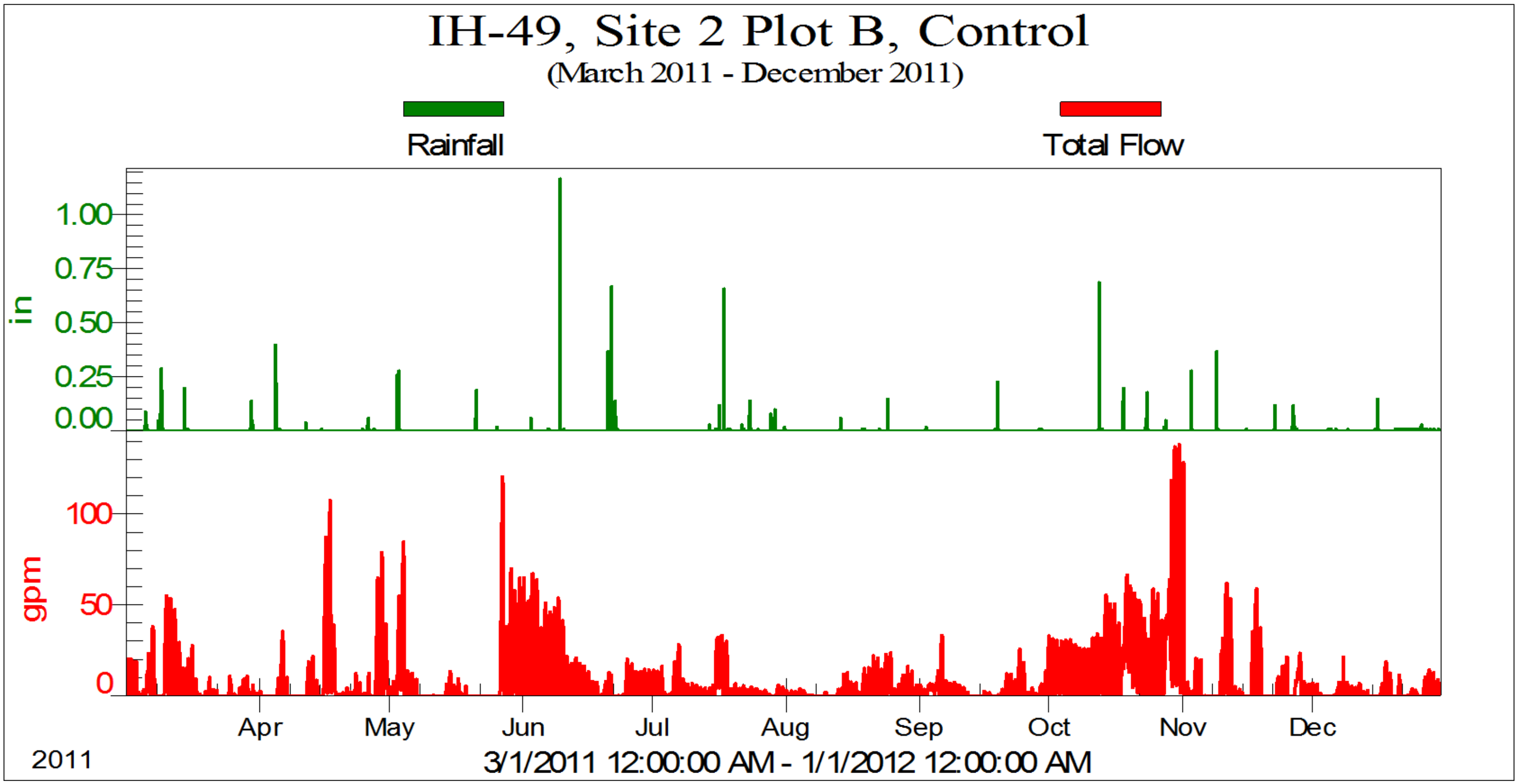

Figure B.8. IH-49, Rapids Parish, Site 2 Plot B, control with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

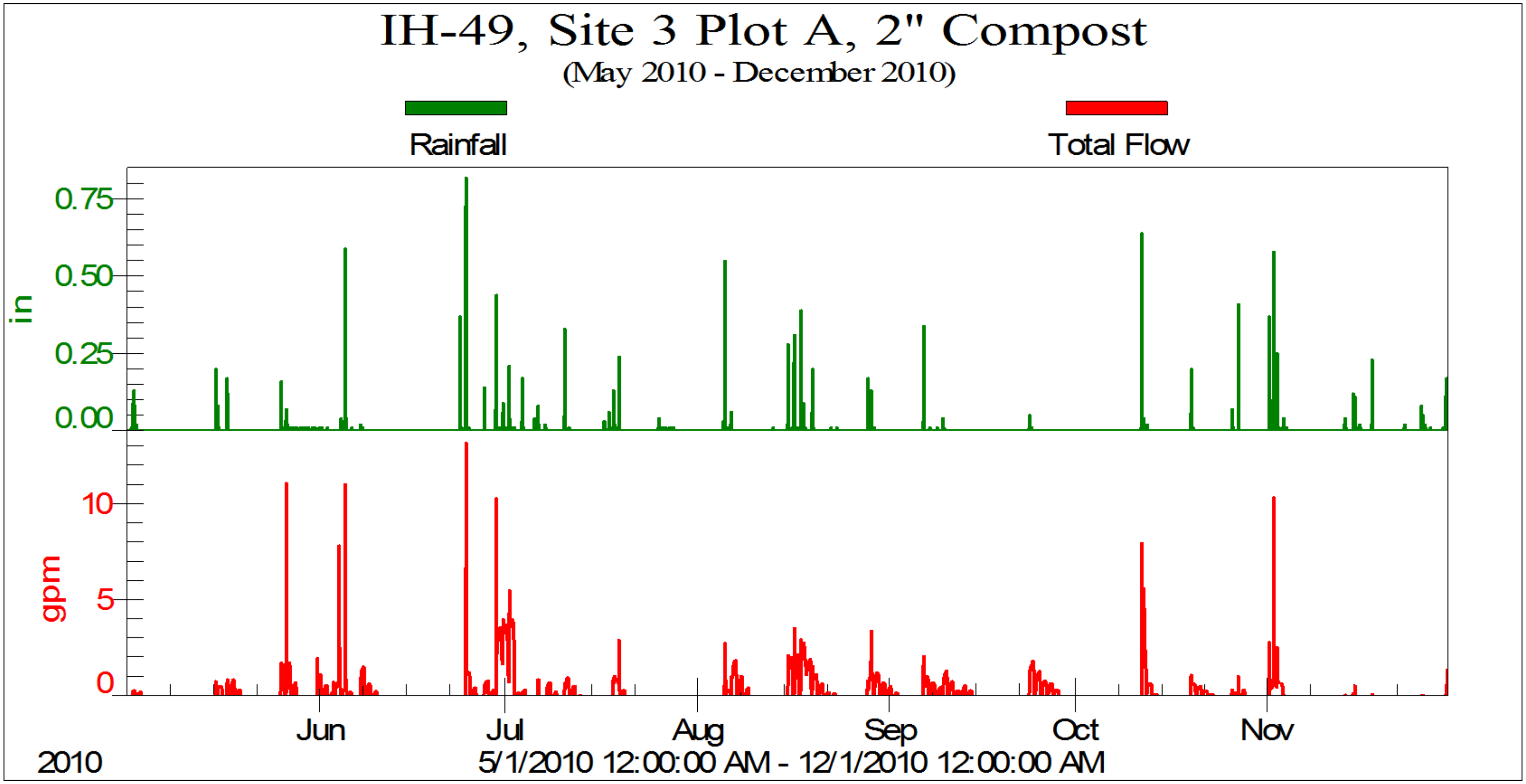

Figure B.9. IH-49, Rapids Parish, Site 3 Plot A, 2” composted mulch application with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

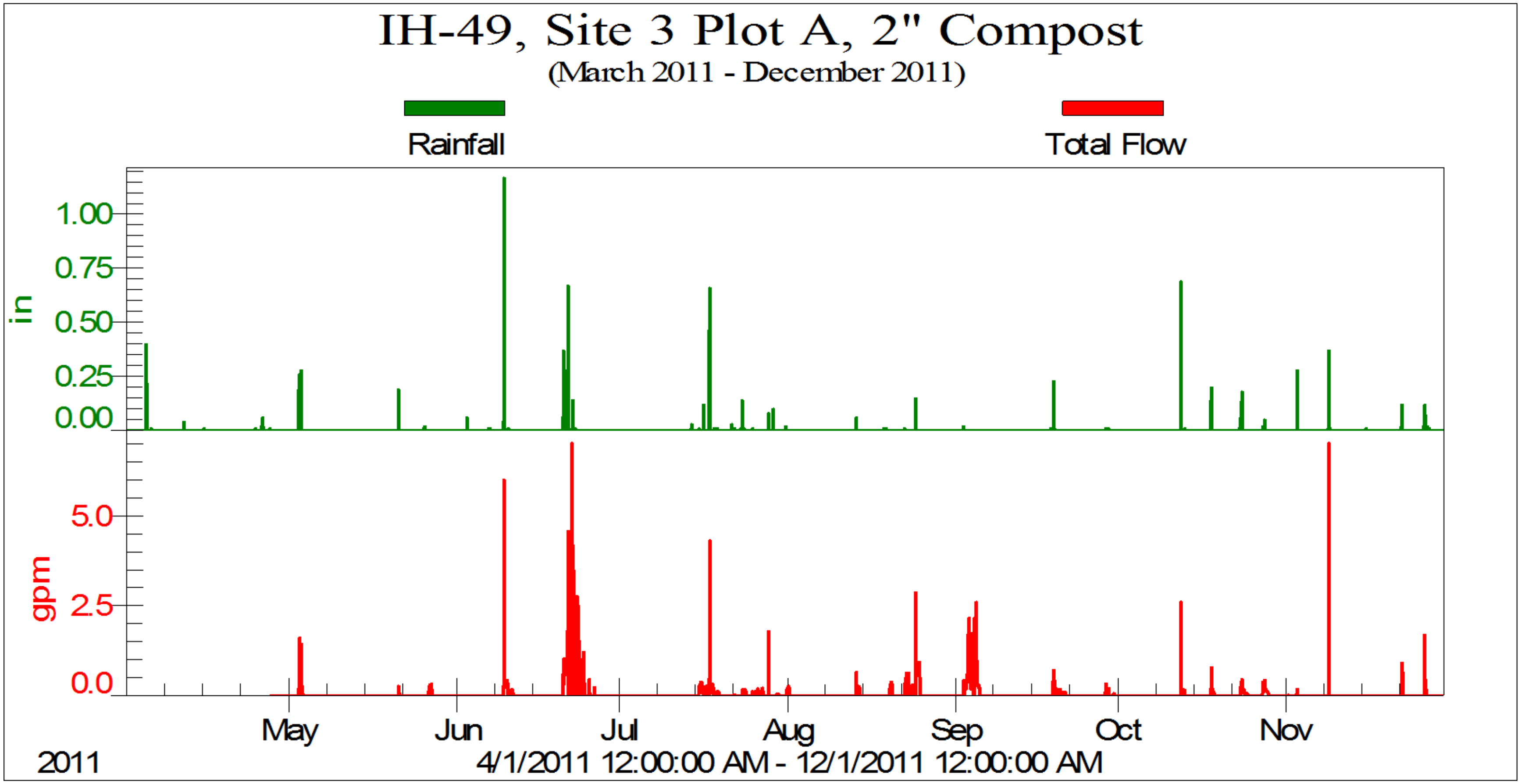

Figure B.10. IH-49, Rapids Parish, Site 3 Plot A, 2" composted mulch application with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

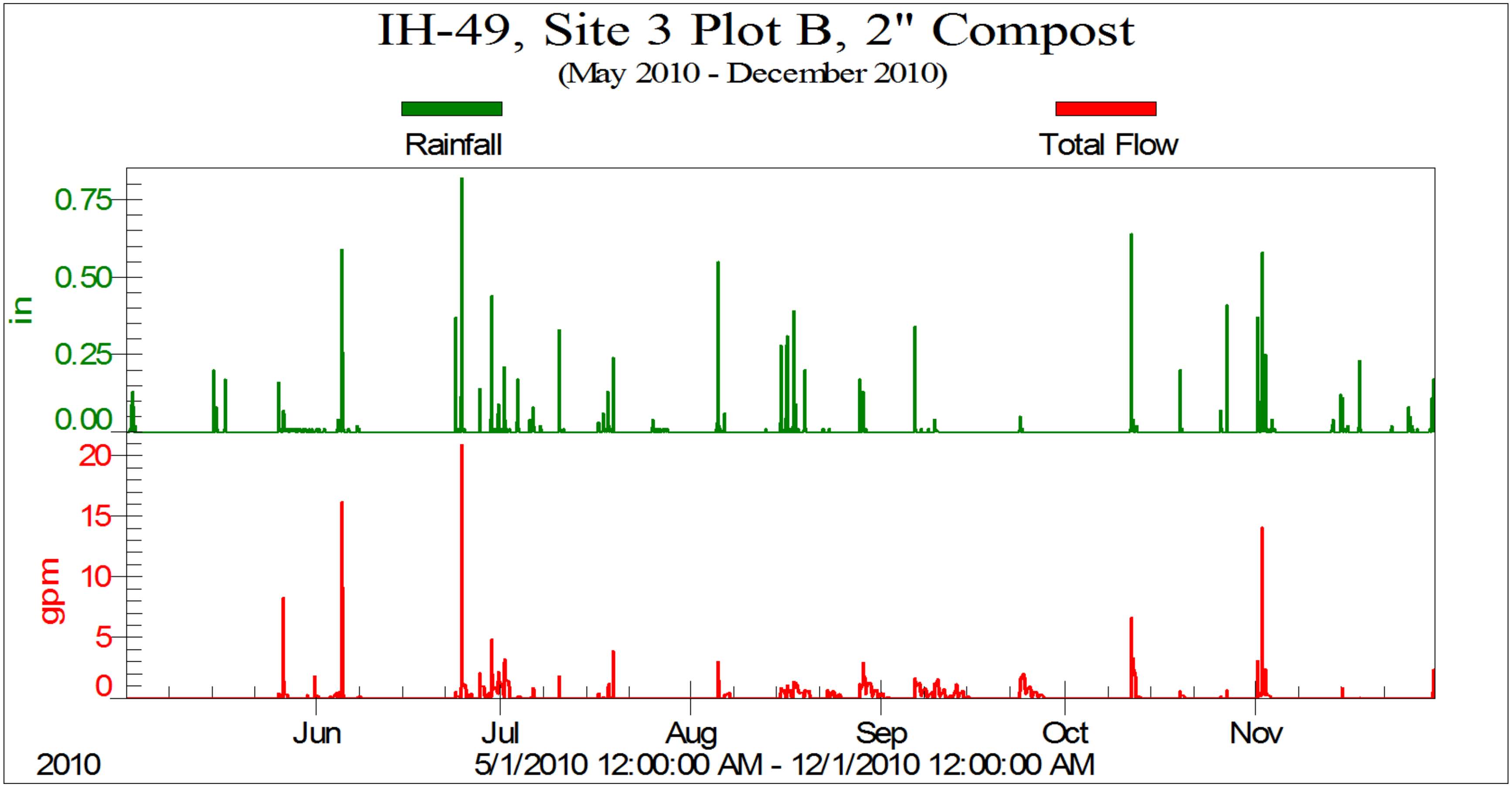

Figure B.11. IH-49, Rapids Parish, Site 3 Plot B, 2” composted mulch application with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

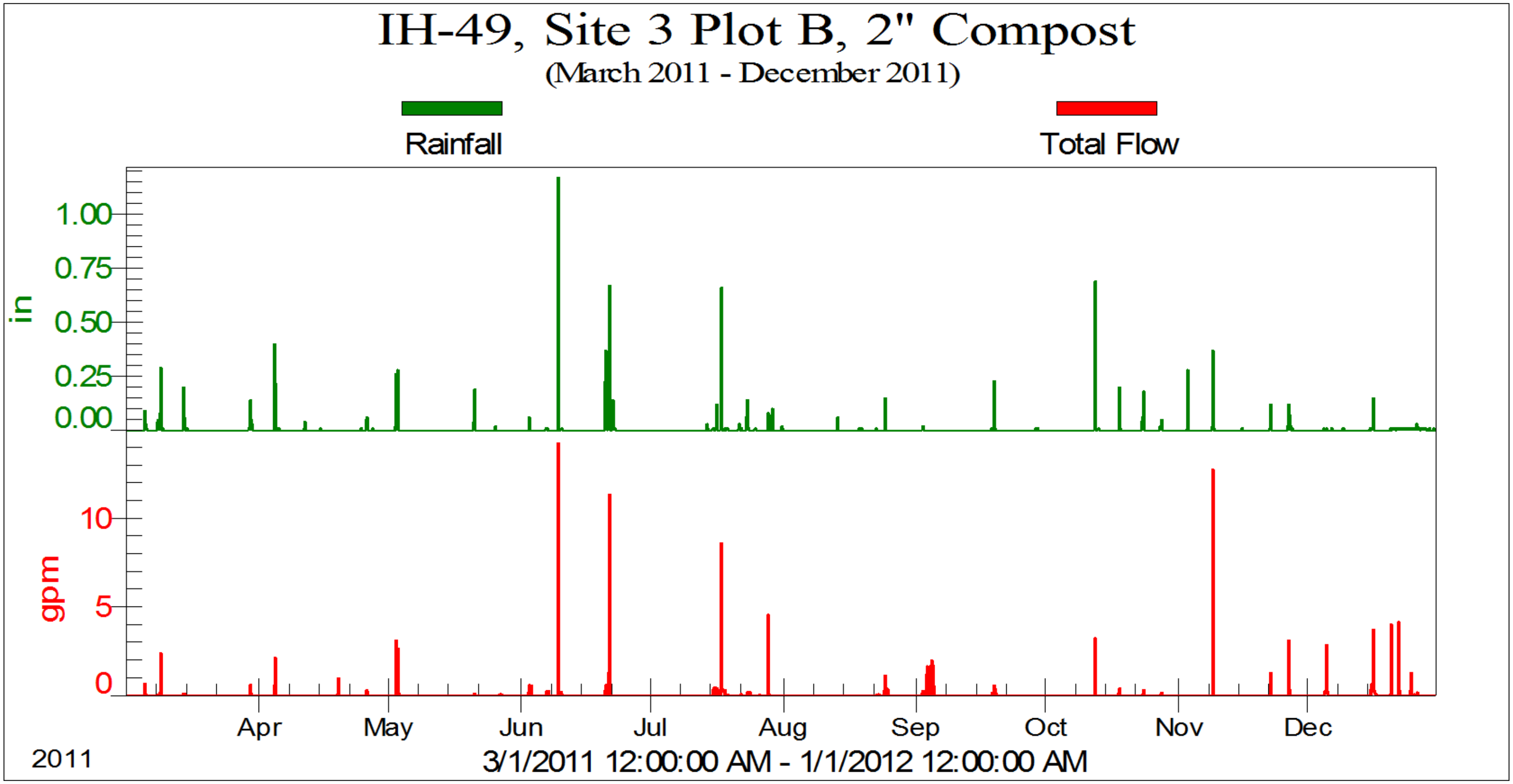

Figure B.12. IH-49, Rapids Parish, Site 3 Plot B, 2” composted mulch application with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

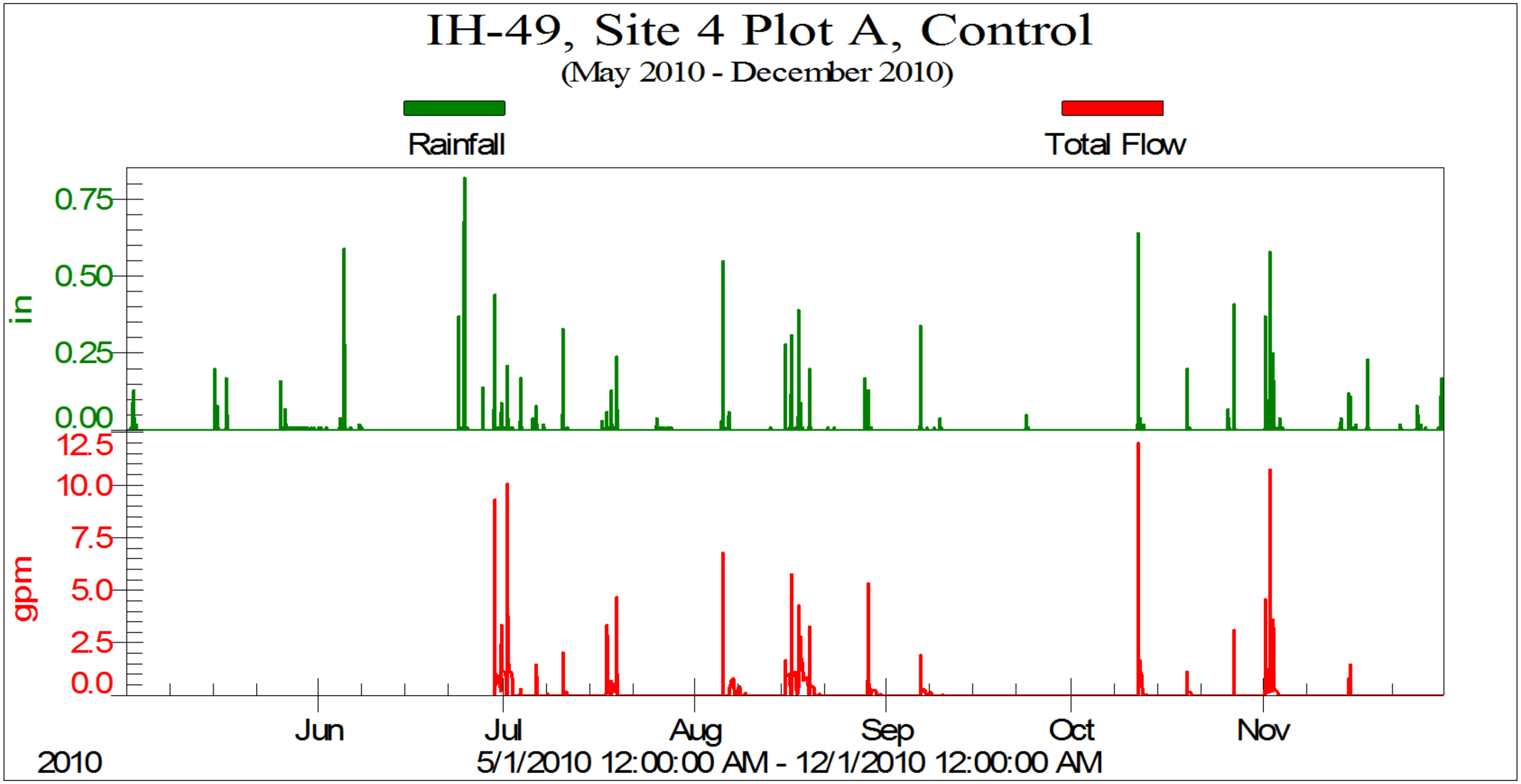

Figure B.13. IH-49, Rapids Parish, Site 4 Plot A, control with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

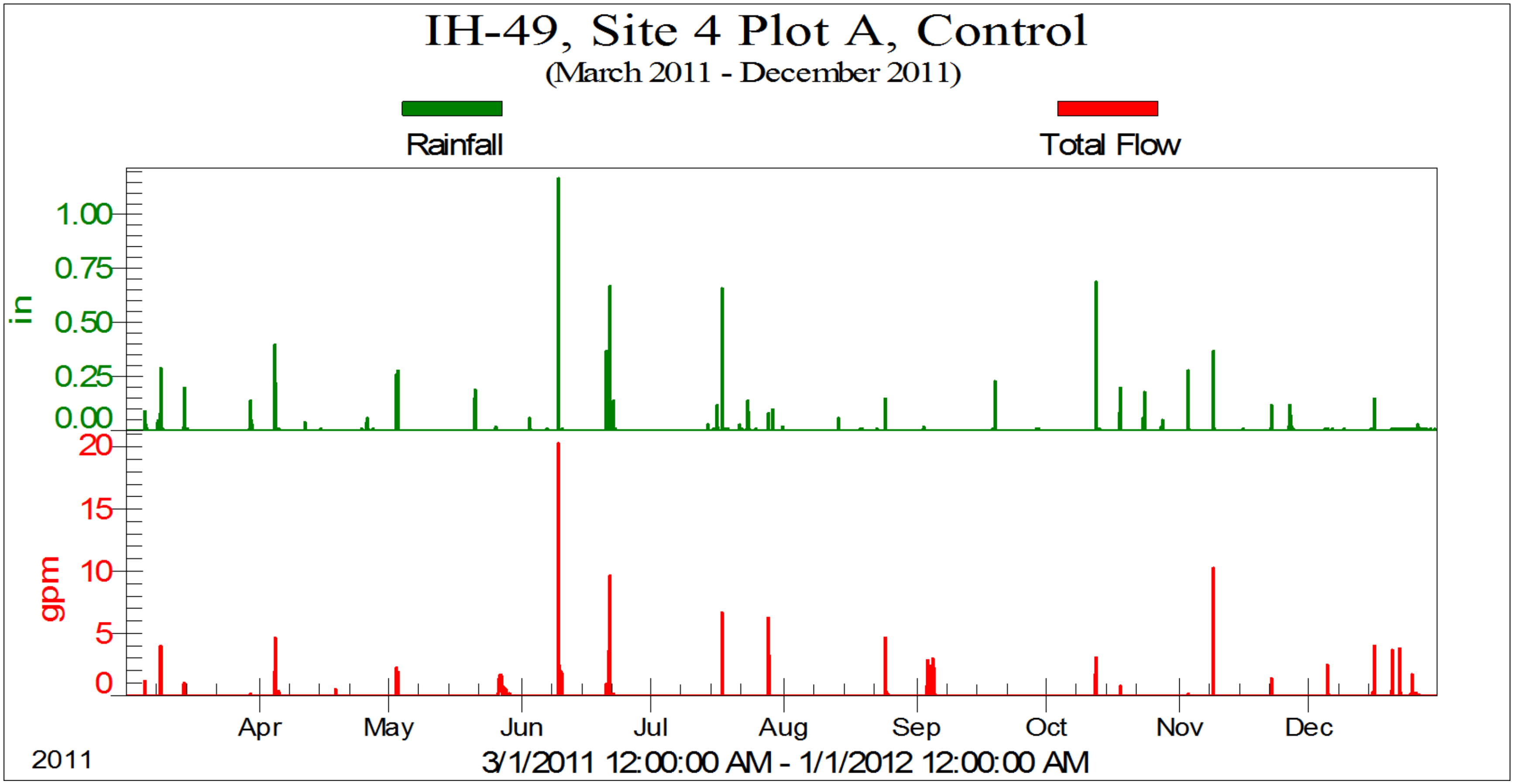

Figure B.14. IH-49, Rapids Parish, Site 4 Plot A, control with no-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

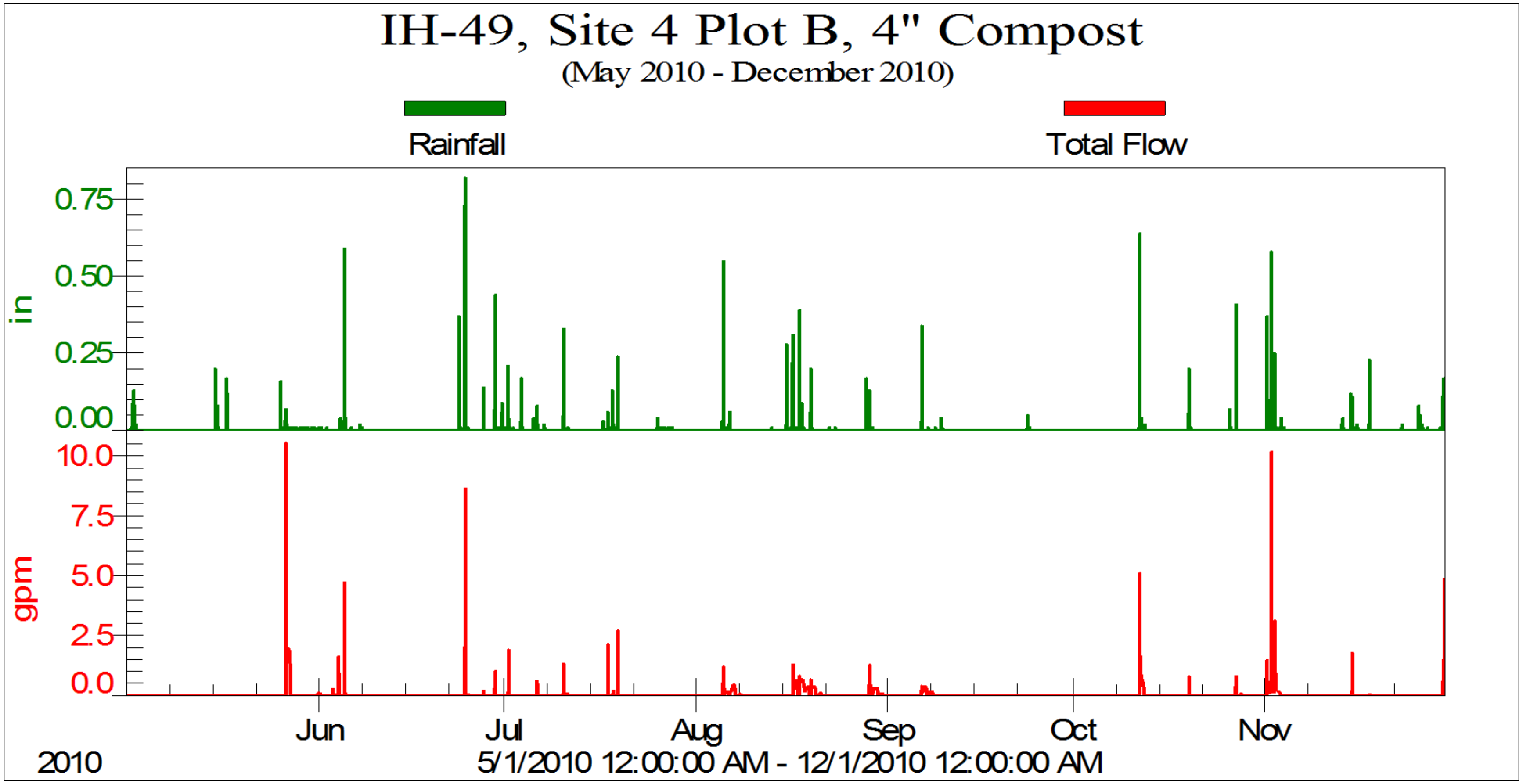

Figure B.15. IH-49, Rapids Parish, Site 4 Plot B, 4” composted mulch application with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from May 2010 to December 2010.

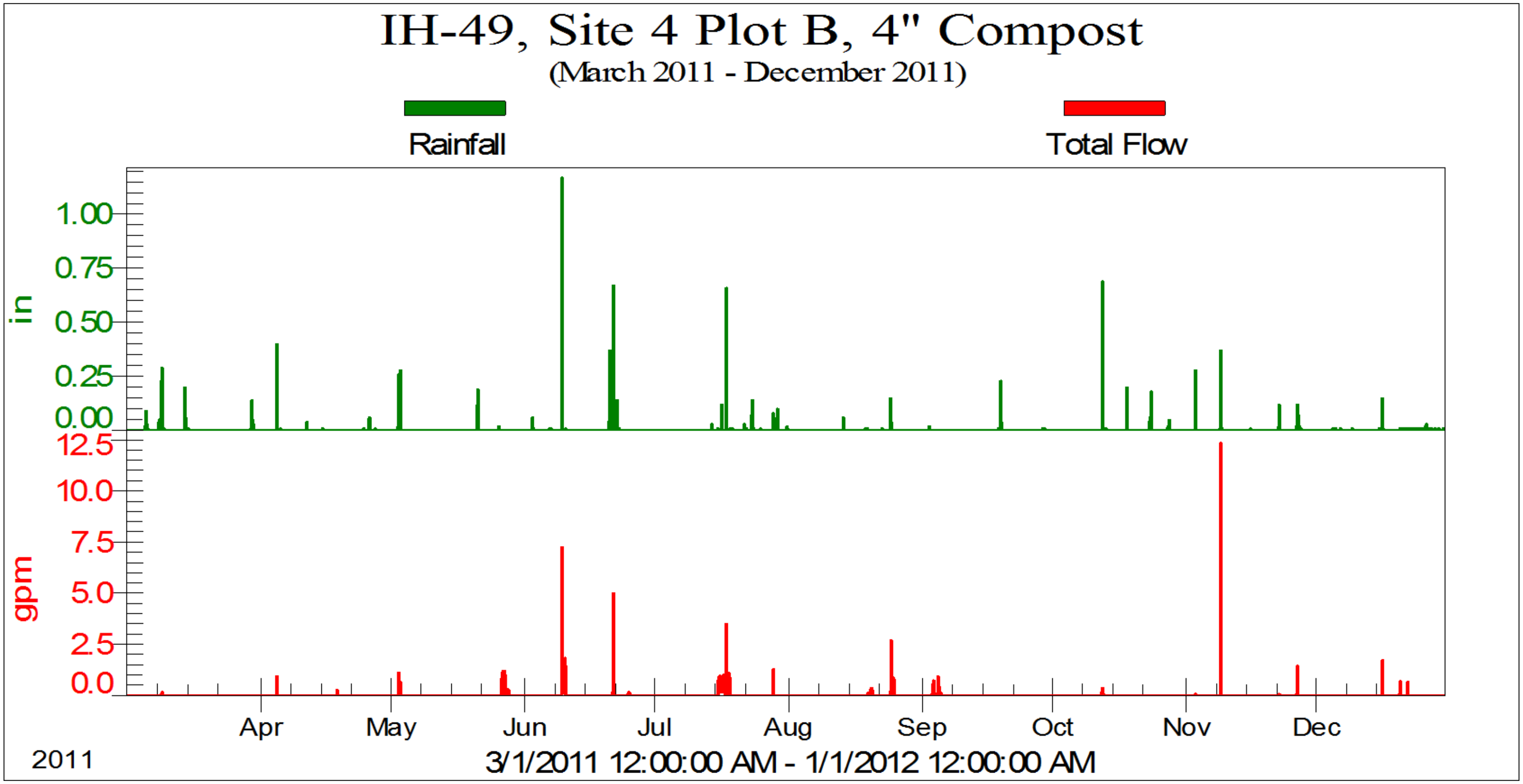

Figure B.16. IH-49, Rapids Parish, Site 4 Plot B, 4” composted mulch application with light-tillage. Rainfall (inches) and calculated flow rate (gallon per minute) from ISCO, from March 2011 to December 2011.

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Subject: RE: Question...

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David C. Weindorf, Ph.D.
Associate Professor of Soil Classification/Land Use, SPESS Graduate Advisor
2011 Fulbright Scholar
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Noura Bakr was born in Alexandria, Egypt, where she also grew up. She graduated from Faculty of Agriculture, Alexandria University in 1999, with a Bachelor of Soil and Water Sciences. She got her Master of Science in Soil and Water Sciences from the Faculty of Agriculture, Alexandria University in 2003. She worked as a test debugger in Fujitsu Services Limited, Egypt, from 2003 to 2004. She was employed as a research assistant in the National Research Center (NRC), Giza, Egypt in 2004 and re-enrolled at Alexandria University to pursue her doctoral degree in 2005. In 2008, she won a chance to be a visiting research associate in the School of Plant, Environmental, and Soil Sciences, Louisiana State University Agricultural Center, and was funded by the Egyptian government for two years. Her major professor, Dr. David Weindorf, offered her an assistantship to complete a full doctoral degree at Louisiana State University in 2010. She was married to Tamer Elbana in 2004 and has two children, Yehia and Janna.