

Impacts of Upstream Disturbances on Downstream Sediment Yield and Stream Morphology where Best Management Practices are Present

by

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Abstract

Attaining high quality water has always been a big concern for humankind. Forested watersheds are known to provide the cleanest form of water. However, conversion of forested lands to agricultural and/or urban use, as well as disturbances created in forested watersheds lead to degradation and deterioration of our water resources. To minimize the disturbance impacts on water quality various best management practices (BMPs) such as streamside management zones (SMZs) are implemented in managed forested watersheds. On the contrary, any upstream urban and agricultural activities where BMPs are not present or are inadequate can negatively impact downstream water quality regardless of the presence of downstream BMPs. In a recent study, two small paired watersheds located near Auburn, Alabama were examined for streamflow and sediment yield in 2009 and 2010 to evaluate the efficacy of SMZs at trapping sediment yield from a clearcut area. Recent urban activities upstream of the study watersheds and poorly designed BMPs around these activities provided us an opportunity to observe and document the impacts of upstream disturbances on downstream stream water quality and morphology. Six monitoring stations were established to observe flow and sediment yield. Sediment data collection began in January 2014, and proceeded until June 2015. In addition to sediment concentration measurements, cross-sections of the channels have also been surveyed at several locations across the streams, following each significant storm events in order to assess the effects on channel morphology. Collected data indicates substantial increase in sediment load. Sediment

concentrations were up to two orders of magnitude higher compared to the levels from the previous study where sediment concentrations were monitored following a clearcutting. Furthermore, channel morphology was altered visibly following almost every significant rain event (>25 mm). Stream channels were subject to instream erosion and deposition over time. Decreasing capacity of SMZs to protect water resources were also observed under the impacts of upstream disturbances. This study clearly showed that assessment of watersheds as a whole is needed in order to define the origin of problems and mitigate them more effectively.

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CHAPTER I - GENERAL INTRODUCTION

1. Water quality

The term ‘Water Quality’ refers to the assessment on suitability of water for its designated use based on selected physical, chemical, and biological characteristics. Uses of water are defined by considering the use and value of the water for public water supply, protection of fish and wildlife, recreational, agricultural, industrial, and navigational purposes (Clean Water Act, 2002). Water quality properties vary with color, conductivity, dissolved oxygen, hardness, pH, salinity, suspended sediment and turbidity based on designated use of the water bodies (USGS, 2014).

The importance of having high quality water has been acknowledged throughout history. For many years, scientists as well as non-governmental organizations have increased their concern on the unstable use of the earth’s available water resources (Gleick *et al.*, 2006). Uncontrollable rise of the world’s population and consumption have been increasing the human demand for domestic, industrial, and agricultural water (Zimmerman *et al.*, 2008). This growing human population will further increase the demand on freshwater (Defries *et al.*, 2004).

Rapid growth in population has another big negative effect; water pollution. Water pollution can be explained as making water unclean to the point where beneficial uses are harmed. Water resources can be polluted by human practices in two ways; point source (PS) e.g. sewage discharge, and non-point source (NPS) e.g. runoff from urban and agricultural areas (Sliva and Williams, 2001). NPS typically results from multiple contaminant sources over

broader area than a PS. It is difficult to reveal NPS because they usually contain large drainage basins with complex interactions (Sliva and Williams, 2001).

Many factors influence water quality such as erosion, sedimentation, runoff, dissolved oxygen, pH, temperature, toxic and hazardous substances and litter and rubbish (USEPA, 2012). Water and wind erosion causes 3.9 billion metric tons of soil loss each year in USA, and majority of the soil eroded comes from agricultural lands (National Resources Inventory, 1987). Runoff is the main cause and carrier of sediment reaching the streams, lakes and other bodies of water as a NPS (USEPA, 2012). The factors contributing to water contamination have arisen with the increasing urbanization, deforestation, inappropriate land use activities such as agriculture and livestock, and global warming. Scientists have shown that land use changes and many kinds of land cover disturbances have important role in deterioration of water quality (Eshleman, 2004).

2. Impacts of land use and land cover (LU/LC) on water quality

All land uses/land covers (LU/LC) have some degree of influence on water quality, either positive or negative. In forests and other areas covered by intense vegetation, most rainfall infiltrates into the soil rather than running off the ground, stream flows are fairly steady, and water quality is usually high. In areas lacking vegetation such as urban places, little rainfall is infiltrated into the soil resulting in high runoff, stream flows have higher peaks, and water quality is poorer. In fact, urban LU/LC practices may be the most damaging factors for the impairment of water quality.

LU/LC changes can be categorized in six groups; deforestation and reforestation, agriculturalization, urbanization, wetland drainage, water resources development, and surface

mining and reclamation (Defries *et al.*, 2004). Changes in LU/LC or any kind of disturbance such as logging, insect defoliation, invasion or entering of exotic species into the ecosystem, volcanic eruption, fire, disease, or overgrazing may reduce stream water quality in watersheds. This can be explained by the close relationship between streams and their watersheds (Webster *et al.*, 1990).

Impacts of three different land covers: agricultural, urban and forest on water quality were examined by Lenat and Carwford (1993), and it was found that runoff not only caused an increased sediment concentrations, but also increased nutrient concentrations in urban and cultivated land while sediment concentrations and runoff were fairly low in forested land.

Activities associated with urban and residential development may be significant sources of nutrients, sediment and other chemicals that influence water quality (Carpenter *et al.*, 1998). Ren *et al.* (2003) analyzed the relationship between urbanization and water quality, and found that the outcome of rapid urbanization was rapid degradation of water quality. Nelson and Booth (2002) also stated that human activity, especially urban development was one of the most important sediment contributors. In addition, Mallin *et al.* (2009) stated that impervious surfaces following urbanization caused an increased concentration of biochemical oxygen demand, fecal coliform bacteria, orthophosphate, and surfactants in stream water.

Many human activities such as construction in urban areas, forestry and agricultural practices significantly alter the LU/LC in watersheds (Isik *et al.*, 2012). Urban, silvicultural and agricultural activities may cause a change in vegetation within a watershed resulting in differences in streamflow pattern. Clearance of natural vegetation for land cultivation is known to cause an increased rate of soil erosion (Walling and Fang, 2003). Impairment of water quality

in Great Lakes Coastal Wetlands was monitored by Morrice *et al.* (2007), and it was concluded that reduction in water quality was positively correlated with anthropogenic stress related to agriculture. Mishra *et al.* (2007) evaluated the mixed land use effect on stream water quality, and stated that cultivated land generated much more runoff and sediment yield in comparison to forested land use.

The literature consists of many studies indicating that vegetation removal and alteration of forested land use to agricultural land use can negatively affect water quality by increasing the sediment yield (O'Loughlin *et al.*, 1980; Chang *et al.*, 1982). Expansion of croplands (460%) and pastureland (560%) from forest and grasslands around the world during the past 300 years increased streamflow, but increasing water quantity decreased water quality due to mobilization of salts and salinization caused by shallow water tables (Scanlon *et al.*, 2007). Restrepo and Syvitski (2006) sought the important drivers that increase sediment yield, and found that decreasing acreage of forest, conversion of forested areas to pasture and cultivated land, mining practices, and increasing urbanization were the main factors of increasing sediment yield. On the other hand, afforestation of grasslands typically results in reduced streamflow and runoff following afforestation (Fraley *et al.*, 2005).

Deforestation, increased agricultural activities and urbanization have also made changes on runoff and amount of sediment, contributing to flood plain deposition, channel erosion and unstable stream channels (Brooks, 1988; Clark and Wilcock, 2000; Grant *et al.*, 2003). Channel erosion and channel instability may result in water quality degradation. Soil eroded by intensive land use such as deforestation and plowing has been considered a concern in the USA since 1920s (Happ *et al.*, 1940; Laften and Moldenhauer, 2003; Triplet and Dick, 2008). Rakovan and Renwick (2011) observed the role of sediment supply in channel instability in Ohio, US, and

found that deforestation for agricultural practices increased sediment yield by causing the stream channel to enlarge laterally in the past. Agricultural lands with effective soil conservation practices have recently decreased sediment yield resulting in the cutting downward.

3. Sediment

The amount of soil material carried by surface water increases rapidly when vegetation is removed from a land surface (Leopold, 1956). In the event of any land disturbances, exposed soil particles are transported by surface water. Sediment is defined as transported, suspended or deposited soil particles by water; and the fragmented materials derived from weathering and eroded rocks (USEPA, 2014). The Environmental Protection Agency (EPA) catalogs sediment as the most common pollutant for streams, lakes and reservoirs (USEPA, 2014). The natural process of erosion, transport and sedimentation may result in severe engineering and environmental issues. Potential impacts of sediment on water quality may decrease water clarity causing reduced visibility for fish, damaged fish gills, changed benthic structure of the stream bed, decreased number of invertebrate species and increased contaminants because sediments are able to transport attached pollutants such as nutrients, e.g. phosphorous, bacteria and chemicals (Lowrance, 1985).

Sediment is considered one of the biggest threats to water quality (Grace, 2005). Changes in natural systems such as volcanic eruptions, human activities such as clearcuts and alterations on land management are the basis of increased sedimentation (Castro and Reckendorf, 1995). Human activities may accelerate the process of erosion by plowing, tillage and by opening the canopy with cutting and burning (Julien, 2010). In some situations, the erosion rate may be 100 to 1000 times harmful than the geological erosion rate of 0.25 tons/ha/year (Julien, 2010).

Agriculture, forestry, urban development, mining and road construction are the major sources of sediment (Waters, 1995). Clearing forest cover creates increased bed sedimentation and turbidity (Dudgeon, 2000). Logging roads can cause a large percentage of sediment entering to stream water at an accelerated rate (Everest *et al.*, 1987). Agricultural land with minimal conservation measures contributes agricultural runoff and sediment yield to water bodies. Because of the increasing demand for agricultural products, double cropping will be more recurrent, and as a result, a greater erosion and runoff will occur (CIEATFWH, 1982). Sediment caused by urban development is generally related to construction of buildings and roads which may lead to erosion problems (Castro and Reckendorf, 1995).

Disturbances on the land surface produce sediment and also influence stream ecosystems (McDonnel and Picket, 1990; Waters, 1995; Delong and Brusvern, 1998). In the USA, 45 % of all the streams and rivers are affected by excessive amount of sediment (Judy *et al.*, 1984). Sedimentation and turbidity are noted as the main contributors of reductions in aquatic fauna (Richter *et al.*, 1997). Andrew *et al.* (2002) observed sedimentation in both disturbed and undisturbed streams, and concluded that sedimentation was much higher in the disturbed stream, fish density and species diversity was greater in undisturbed stream with much lower sedimentation. Increases in suspended sediment levels in streams may clog feeding structures and reduce feeding capacity of aquatic invertebrates, or kill these organisms (Hynes, 1970). Impacts of suspended sediment on algae are correlated with its influence on light penetration (Newcombe and McDonald, 1991). Additionally, contaminants absorbed by suspended sediment may alter growth rates and biomass of algae (Newcombe and McDonald, 1991).

4. Forested watersheds and water quality

Forests are essential for water supply, quality, and quantity everywhere in the world (Bates *et al.*, 2008). Waters from forested watersheds usually include less sediment, nutrients and chemicals when compared to other lands (Wear and Greis, 2002; Chang, 2003; Brinkly *et al.*, 2004). The most abundant and cleanest water are provided by forests, because forests can store, filter and release the water for downstream (Furnis *et al.*, 2010). Forested watersheds have high evapotranspiration (ET) rates, and the soil in forests has high infiltration capacity (Giambelluca, 2002). In the Western USA, forests are the source of 65 % of the water supply; and 18 % of the Nation's water need is provided by National Forests (Brown *et al.*, 2008). Streams in forested watersheds are also high quality habitats in comparison to streams draining other LU/LC (Wallace, 1988; Kratzer *et al.*, 2006).

Since the roots of trees are substantial and fairly deep in forests, forest soils have high macroporosity, low bulk density and high infiltration rates which help mitigating surface runoff (Neary *et al.*, 2009). Understory and organic matter in the forest floor help keep nutrients in undisturbed forests (Swank and Waide, 1988). Forest soils enable retention of most nitrogen inputs even after timber harvesting (Likens and Bormann, 1995; McBroom *et al.*, 2008).

Although forests are unique and long-lived, they are exposed to structural alterations such as management practices or natural disturbances (Lockaby *et al.*, 2013). In order to sustain and benefit from all the services that forests provide, forests are managed using silvicultural practices. These practices may contribute to NPS pollution by negatively affecting water quality due to soil disturbances during the operations (Saleh *et al.*, 2004). Many studies have indicated that the main sources of sediment caused by silvicultural practices are roads, landings and skid trails (Trimble and Weitzman, 1953; Megahan and Kidd, 1972; Riverbark and Jackson, 2004).

Forest roads generate large surface runoff and high peak flows (Wemple and Jones, 2003), and soil is usually most erodible just after the construction of roads (Megahan and Kidd, 1972).

Forest harvesting also influences and increases soil compaction (Lockaby and Vidrine, 1984; Guo and Karr, 1988) and compacted soils cause much more storm runoff with lower infiltration rates (Dickerson, 1976). Clearcutting usually induces a decline in evapotranspiration and increases streamflow (Douglas, 1980); as well as increases concentration of suspended sediment loads (Binkley and Brown, 1994). Patric (1980) pointed out that clearcut increased the temperature of the forest floors and increased streamflow as well as exporting plant nutrients in West Virginia. Similarly, clearcutting resulted in excessive amounts of suspended solids, total nitrogen and phosphorus in southeastern North Carolina (Ensign and Mallin, 2001).

5. Best Management Practices (BMPs) and water quality

Against all the negative impacts of urban, agricultural and forestry disturbances mentioned above, Best Management Practices (BMPs) are developed as a result of the need to protect water quality. BMPs are devices, systems and procedures designed to protect water quality from NPS (AL Forestry Commission, 1999). They are usually divided into three groups (Conservation Technology Information Center, 2008);

- Urban BMPs
- Agricultural BMPs
- Forestry BMPs

Urban BMPs are usually referred to as storm water treatment practices. Pollutants related to NPS are controlled by slowing, retaining and absorbing pollutants generated by surface runoff in urban areas (Mandelker, 1989). Commonly used BMPs are infiltration trenches, dry wells,

infiltration basins, grass swales, porous pavements, extended detention dry ponds, artificial marshes, water detention/retention basins and recharge basins (Tsihrintzis and Hamid, 1997). Installing porous pavements can clear away pollutants carried by urban runoff (Galli, 1992). Detention basins are also very effective practices for managing storm-generated pollutions (Oliver and Grigoropolus, 1981).

Agricultural BMPs may be both structural such as terracing, impoundments and fencing, and nonstructural such as conservation tillage and strip filters (Park *et al.*, 1994). Impoundments slow the flow to downstream receiving waters (Laften *et al.*, 1978). Conservation tillage not only significantly reduces runoff, soil loss and nutrient loss, but also increases the infiltration (Loehr *et al.*, 1979). Park *et al.* (1994) observed the effectiveness of agricultural BMPs during pre- and post-implementation of no-till and critical area treatment in eastern Virginia, and concluded that BMPs were very effective at reducing N and P concentrations.

BMPs are also very successful in reducing NPS pollution from silvicultural practices in forested watersheds (Kochenderfer *et al.*, 1997; Stuart and Edwards, 2006). Forestry BMPs generally comprise of the devices eliminating the impacts of forest clearing, tree removal and stream crossing. Streamside Management Zones (SMZs), culverts, bridges, turnout ditches, broad-based dips, water bars and chemical site preparation are the types of forest BMPs to prevent NPS pollution (AL Forestry Commission, 1999). Forestry BMPs contribute to maintaining water quality following forestry practices (Grace, 2005). Wynn *et al.* (2000) observed the effectiveness of BMPs including SMZs, water bars and landing seeding with grass after clearcutting, and concluded that in case of no BMP implementation; forest clearcutting causes highly significant increases in sediment and nutrient concentrations. In another study, water quality was assessed by Arthur *et al.* (1998) in three watersheds which were undisturbed,

harvested with no BMP implementation, and harvested with BMP implementation, and it was found that compared to the undisturbed watershed, sediment rates were 14 times greater on the harvested watershed with BMPs and 30 times greater on the watershed with no BMPs. Sediment from forest roads is another source of erosion in forested watersheds (Ursic and Douglas, 1978). Waters bars or broad-based dips may be installed to reduce the effects of forest roads on water quality. Vegetation establishment can also be useful to prevent the sediment generated by forest roads (Grace, 2002).

5.1. Streamside Management Zones (SMZs)

Streamside Management Zones (SMZs) are a strip of land along a stream or adjacent to a body of water such as lakes (Simpson, 2002). They are very effective in protecting water quality, aquatic ecosystem, and wildlife habitat from the management practices near or adjacent to those water sources (Daniels *et al.*, 2012). SMZs have several benefits and functions (Daniels *et al.*, 2012);

- they maintain water quality by reducing surface runoff, and filter sediment that is carried to the water bodies by surface runoff during storm events,
- protect stream channel and banks,
- maintain stream temperature,
- provide shelter for wildlife animals,
- protect wetlands,
- and help control flooding.

Effectiveness of SMZs depends on several factors such as width of the buffer, density of the vegetation, and slope (Alabama Forestry Commission, 1999). It is suggested that width of a SMZ

should be at least 10 meters from each side of the stream slope (AL Forestry Commission, 1999). Even though SMZs have been effective to protect water sources from near and adjacent forestry practices (Daniels *et al.*, 2012), any disturbance in the watersheds may influence their effectiveness.

Several studies have been reported on the effectiveness of streamside buffers on water quality (Keim and Schoenholtz, 1999; Ensign and Mallin, 2001; Lakel *et al.*, 2006). Studies usually follow distinct methods to monitor the efficacy of SMZs under different conditions. Impacts of intensive forestry practices such as clearcutting, thinning, site preparation, forest fertilization, prescribed burning were monitored, and SMZs were found to have positive effects to reduce sediment movement to the water bodies (Keim and Schoenholtz, 1999; Ensign and Mallin, 2001; Lakel *et al.*, 2006).

Lakel *et al.* (2006) monitored the effectiveness of SMZs on water quality in sixteen watersheds using three different forestry practices; clearcut harvesting, prescribed burning, and planting with loblolly pine (*Pinus taeda* L.), and stated that all SMZs used with varying widths were effective to protect water quality from forestry practices. Contrary to this, Fraser *et al.* (2012) observed the effectiveness of SMZs at trapping sediment following harvesting and regeneration operations, and found that smaller width SMZs had been insufficient to protect water quality. However, they also stated that using wider SMZs than initial width increased the efficacy of the buffer zones.

As stated before, SMZs have other benefits in addition to protecting water quality. Macroinvertebrate populations were observed under the impacts of forestry practices by McCord *et al.* (2007), and it was found that SMZs had strong positive influence in protecting both water

quality and aquatic life. SMZs also protect stream channel and banks by reducing the velocity of stream water. Diverse and healthy vegetation has positive effects on stream channel shape and size. Due to the presence of woody species with extensive root systems along the SMZs, better streambank protection is provided by SMZs. Well vegetated streams are usually narrower and deeper than the streams with poor vegetation cover because of the binding capacity of plants and their roots (Comfort, 2005).

Although SMZs are highly recommended and effective at protecting water quality, in some situations they may be inefficient. This inefficacy can be explained by buffer width, slope or other factors. Ensign and Mallin (2001) monitored the efficacy of SMZs on water quality during some forestry practices. Water quality was observed before harvest for 2 years, during clearcut and the following clearcut for two years. Results showed that higher sediment concentration was monitored in spite of a 10 m uncut buffer zone compared to neighboring control watershed. They concluded that 10 m buffer zone is not enough to prevent effects of clearcut on water quality in their study area. Corner and Bassman (1993) suggested that width of a buffer zone should be determined as a function of physical parameters (e.g. slope, soil permeability, soil erodibility) and intensity of management practices. In addition, Kara *et al.* (2012) observed the efficacy of a stream buffer against a clearcutting area, and concluded that the buffer zone was incapable of trapping sediment yield from adjacent clearcut and road.

6. Research needs

Many studies have been conducted to analyze the impacts of LU/LC changes and impacts of watershed disturbances on stream water quality and aquatic life. (Webster *et al.*, 1990; Lenat and Carwford, 1993; Morrice *et al.*, 2007; and Isik *et al.*, 2012). The need for BMPs to mitigate and prevent the effects of LU/LC disturbances on water quality, and their effectiveness in protecting

water quality has been well studied (Loehr *et al.*, 1979; Mandelker, 1989; Arthur *et al.*, 1998; Grace, 2005). SMZs are designed to protect water quality, aquatic system, and wildlife habitat from the silvicultural management practices near or adjacent to water sources (Daniels *et al.*, 2012). However; even if all necessary measures are taken into account to protect stream water quality and morphology, these measures may not be adequate to protect water quality if there is any other disturbances in watersheds. Positive influences provided by BMPs may decrease or BMPs may lose their effectiveness over time under the impacts of human interventions. To our knowledge, reasons for the decreases in the efficiency of BMPs, and degradation of stream channel structure and stream water quality caused by this reduction in the efficiency have not been well understood. There is a need to highlight the importance of watershed management in a holistic fashion in order to maximize the services provided by watersheds.

7. Objectives

The overarching goal of this research is to highlight the importance of managing watersheds as a whole because focusing on the entire watershed contributes to a balance among efforts to control NPS pollution and maintain clean water sources. Taking this approach into account, the main objective is to evaluate the impacts of upstream disturbances on stream water quality and morphology, and to assess the capacity of BMPs to sustain their effectiveness under these disturbances. The specific objectives of the study are to;

1. Assess the impacts of upstream urban practices on downstream sediment yield and stream morphology in two small watersheds with SMZs,
2. Determine whether the upstream urban disturbances negate the benefits gained by SMZs.

8. General outline

This thesis is divided into two main chapters (Chapters II and III) that address the influences of human impacts on stream water quality and stream channel morphology in the presence of BMPs. This chapter serves as a literature review covering the topics in chapters II and III. It discusses the importance of water quality, stresses on water quality and the measures taken to minimize the impairments on water quality.

Chapter II examines the relationship between upstream urban practices and downstream sediment yield where BMPs are present or are inadequate to protect stream water quality. It also discusses the stream flow trends under the impacts of upstream disturbances. Chapter III presents the changes on downstream channel morphology in the presence of BMPs when intensive urban practices are present in the upstream. Chapter IV synthesizes the results presented in the two main chapters of the thesis.

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CHAPTER II – IMPACTS OF UPSTREAM URBAN ACTIVITIES ON DOWNSTREAM SEDIMENT PRODUCTION IN THE PRESENCE OF BMPs

Abstract: Although higher quality of water is usually provided by forested watersheds, intensive forestry practices are known to negatively affect water quality if best management practices (BMPs) such as streamside management zones (SMZs) and well-designed logging roads are not present. Any upstream disturbance due to urban, agricultural or silvicultural activities where BMPs are not present or inadequate may negatively impact downstream water quality and stream ecology regardless of the presence of downstream BMPs. The objectives of this research were to evaluate the effects of upstream urban practices on downstream sediment yield where SMZs are present, and the effectiveness of BMPs in controlling sediment yield from the disturbed areas in two small forest-dominated watersheds near Auburn, Alabama. Streamflow and sediment yield were examined over a fifteen-month period from January 2014 to June 2015. Total sediment load increased at downstream monitoring sites as a result of upstream disturbances. Substantial amounts of sediment load were monitored across all monitoring sites in both watersheds. SMZs were found to be ineffective at mitigating the impacts of upstream disturbances. This study emphasizes the necessity of watershed management as a whole, and the importance of BMP applications within the entire watershed.

Key words: Best management practices, disturbance, sediment, stream buffer zones, urban, erosion.

1. INTRODUCTION

Water is a vital natural resource because life cannot exist in the absence of it. Rapid population growth and changes on land use/land cover (LU/LC) are strongly associated with water quality degradation (Smith *et al.*, 1987). Land use activities often result in deterioration of water quality due to increasing nonpoint sources of pollution (NPS) (Rhodes, 2001). The nature of NPS depends on the impacts of LU/LC on water quality (Waneilista *et al.*, 1977). Contaminated runoff from urban areas, agricultural fields, animal feedlots, roadways, abandoned mines, silvicultural and construction activities are major contributors to NPS (Baker, 1992). Controlling NPS is fairly difficult because it originates from a broad area rather than a particular point (Nikolaidis, 1988). Therefore, NPS has been defined as the main cause of surface water impairment in US (ASIWPCA, 1985).

Conversion of forested land to agriculture and pasture, mining practices and increasing urbanization significantly increase NPS (Restrepo and Syvitski, 2006). Positive correlations are usually found between suspended solids and urbanization (Freemen and Schorr, 2004). In addition, increasing imperviousness as a result of urbanization may increase sediment loading to rivers and water bodies (Randhir, 2003). Sediment has been shown to be a fundamental NPS pollutant in rivers and lakes of the US (USEPA, 1190b). Excessive sediment negatively influences fish habitat, recreational use and water storage capacity (Baker, 1992). Alterations of natural systems, human activities such as forest clearcuts and changes in land management result in erosion and sediment transport to downstream waters (Castro and Reckendorf, 1995). For instance, removal of forest cover has been shown to cause increased stream bed sedimentation and turbidity (Dudgeon, 2000).

Forests are unique for water supply, quality and quantity (Bates *et al.*, 2008). Because of the low sediment transport from forests (Chang, 2006) and positive effects of understory on sediment trapping in forested watersheds (Grace, 2002), cleaner water is usually provided by forests (Furnish *et al.*, 2010). Forests are often managed through silvicultural practices (Silvicultural guidelines, 2009). However, these practices may contribute to NPS pollution, and therefore negatively impact water quality (Saleh *et al.*, 2004; McBroom *et al.*, 2007). For instance, forest harvesting results in soil compaction that may eventually causes increasing runoff and sediment transport (Guo and Karr, 1988).

Even though nature is able to recover from human-caused disturbances, wise application of essential knowledge regarding the water is needed to ensure the sustainability of quality and quantity (Tebbutt, 1998). Herein, BMPs are known to be very effective tools to protect and sustain water quality under the potential impacts of urban, agricultural and forestry disturbances (AL Forestry Commission, 1999; Wynn *et al.*, 2000). They are designed to protect water quality by reducing NPS (AL Forestry Commission, 1999). Harvested forests may generate more total suspended solids (TSS) in the absence of BMPs than harvested forests with BMPs in place (Wynn *et al.*, 2000).

Stream Management Zones (SMZs) are the most common type of forestry BMPs, and water quality, aquatic ecosystem integrity, and wildlife habitat are well preserved with the help of SMZs during silvicultural practices (Daniels *et al.*, 2012). In addition to their positive influence on water quality, stream buffers also support more diverse macroinvertebrates compared to unprotected streams (Newbold *et al.*, 1980). SMZs have been successful in trapping sediment carried in concentrated flow (Lacey, 2000; Ward and Jackson, 2004). Effectiveness of SMZs

depends on several factors such as width of the buffer, density of the vegetation, and slope (Alabama Forestry Commission, 1999).

Human-caused land disturbances following urban, agricultural and forestry practices are the major cause of increasing sediment loads in rivers and lakes (Guo and Karr, 1988; Waters, 1995; Randhir, 2003); thus, BMPs are needed to prevent sediment movement into streams (Loehr *et al.*, 1979; Wynn *et al.*, 2000). However, if BMPs are not properly prescribed and installed, they may not operate as efficient as they were intended for.

Kara *et al.* (2014) had studied the effectiveness of SMZs on protecting water quality from a clearcut area on two small watersheds where this study was conducted. They found that SMZs were not effective enough to trap the sediment from the clearcut area. Further, they partially cut the SMZ in one of the watersheds to increase its effectiveness by creating higher roughness and a well-developed understory, and reported that the partial cutting significantly increased the sediment yield on the treated sections over the next 6 months period (Kara *et al.*, 2014). This increase in sediment load was expected to phase out as understory starts growing and generating higher roughness. The initial objective of this study was to monitor recovery of harvested watersheds, and changes in sediment yield in the long term following the clearcut and the partial harvesting within that SMZ. However, the preliminary data had unexpectedly high sediment concentrations. Further exploration of the study watersheds indicated that the high amount of sediment movement could be attributed to intense urban practices in the upstream section of the study watersheds, and consequently led to a change in the study objective. Thus, this study provides further evidence to the significance of watershed management as a whole, and the importance of BMP applications in the entire watershed.

This study's objectives are 1) to assess the impacts of upstream urban activities with no or poor BMPs on downstream water quality, and 2) to observe the efficacy of downstream SMZs to protect water quality from upstream urban disturbances. It is hypothesized that upstream urban disturbances can negatively affect downstream water quality by increasing sediment yield even in the presence of downstream BMPs.

2. METHODOLOGY

2.1. Study Site

The study area is located on Mary Olive Thomas Demonstration Forest (MOT) near Auburn, Alabama (Figure 2.1). The total area of MOT is 158 ha (MacKenzie, 1999). The property belongs to Auburn University School of Forestry and Wildlife Science. The property includes different harvesting and regeneration combinations, stand maintenance through thinning and prescribed burning, fire control and fire lanes, environmental protection with BMPs are demonstrated in MOT as part of undergraduate classes. MOT is the toe of an upland area, and is on a transition zone from a Piedmont upland to a bottomland. Average annual rainfall is 1480 mm, and 50 % of the rainfall occurs during the growing season from April to September (Dubois *et al.*, 2000). The average daily temperature is 16.4 °C. Average humidity is around 50 % during the afternoon, and greater during night (McNutt *et al.*, 1981). The dominant soil types, Pacolet and Toccoa, are fairly productive for forests (McNutt *et al.*, 1981). Average slope is less than 6 %, but there are also steep slopes in some parts of MOT. Pine sawtimber, hardwood sawtimber, loblolly pine (*Pinus taeda* L.) plantation, and natural longleaf pine (*Pinus palustris* Mill.) regeneration are the stand types in MOT (MacKenzie, 1999). Deciduous species such as white oak (*Quercus alba*), sweetgum (*Liquidambar styraciflua*), yellow poplar (*Lirodendron tulipifera*

L.) are found within the SMZs. The SMZs are wider than 10 m recommended by the state of Alabama guidelines (AL Forestry Commission, 1999).

Two small adjacent watersheds named West watershed (W_W) and East watershed (W_E) were used in this study. The total area of W_W is 37 ha and the area of W_E is 50 ha. Both watersheds were divided into 3 sub-watersheds from upstream to downstream as W_{W1} , W_{W2} , and W_{W3} for W_W ; and W_{E1} , W_{E2} , and W_{E3} for W_E (Figure 2.2). Delineation of the study watersheds and stream network were completed by using 10 m Digital Elevation Model (DEM) with ArcSWAT. The highest and lowest elevations are 230 m and 180 m, respectively.

2.2. Land use / Land cover

Land use map of the study area was created using ArcGIS. An aerial photo taken in 2014 (Figure 2.1) was used as the base layer for digitizing. The northern part of (W_{W1}) is mostly pasture and forest, and there is also a pond in the middle of this section. The central section of W_W (W_{W2}) is almost completely covered by dense forest, and the downstream section of W_W (W_{W3}) is entirely forested. North of W_E (W_{E1}) is mostly residential area and forest with a pond in the middle. The middle and downstream section of the W_E , sub-watersheds (W_{E2} and W_{E3}) are also mainly forested (Figure 2.2). The LU/LC in the watersheds varies as follows: residential area (8.3 ha), forest (65.7 ha), and pasture (11.2 ha). SMZs (12 ha) are located throughout the streams within study area. The sub-watersheds W_{W1} and W_{E1} are not included within MOT. Intense urban practices were observed at the upstream sections (W_{W1} and W_{E1}) of the study watersheds (Figure 2.3). Based on personal communication, landowners' intend was to enlarge and deepen the ponds.

The clearcut area harvested in 2008 is not visible on the current aerial photo taken in 2014 because the clearcut area has totally recovered by natural loblolly pine trees. The aerial photos from 2008 and 2014 present changes in the clearcutting area over 6 years following the harvesting (Figure 2.3).

2.2. Monitoring Sites

Monitoring sites were selected and defined as W_1 , W_2 , W_3 , E_1 , E_2 , and E_3 from upstream to downstream for each sub-watershed (W_{W1} , W_{W2} , W_{W3} , W_{E1} , W_{E2} , and W_{E3} , respectively) in order to obtain stream stage and sediment samples (Figure 2.2). W_1 and E_1 are located on the south boundaries of W_{W1} and W_{E1} , and were selected to capture the sediment movement from the disturbed area to the forested area. W_2 and E_2 are installed near the north boundaries of the previous clearcut area, and were selected to examine the influence of forest cover on sediment in the stream. W_3 and E_3 are at the south end of the study watersheds, and were selected to evaluate the changes in sediment transport through the SMZs 6 years after the clearcut and the partial harvesting within the SMZ of W_w .

3.3. Sediment and Hydrologic Sampling

Water stage measurements and sediment sampling started in January 2014, and completed in June 2015. Solinst Levellogger Gold Model 3001 pressure transducers were used for continuous monitoring of water stage. The transducers were set to collect stream stage levels every 15-minutes. They were mounted to a permanent T-post at each monitoring site where the channel cross section is not likely to change over time. These locations are suitable for obtaining discharge measurements as well. The actual water level is found by compensating for variation in barometric pressure (Levellogger User Guide, 2014). Barometric pressure from the Auburn-Opelika Airport Station (72228403892) for every 15 minutes was acquired using National

Oceanic and Atmospheric Administration (NOAA) website (<http://www.ncdc.noaa.gov/cdo-web/datasets>). Hourly rainfall data for the study period was also acquired from State Climate Office of North Carolina website (<http://climate.ncsu.edu/>).

Stream discharges were measured at each monitoring site during significant rain events. Stream velocity was recorded at 0.6 m depth by using a Marsh-McBirney Inc. Model 2000 portable flowmeter. When water level depth was deeper than 0.3 m at any monitoring site, flow measurements were also taken at 0.8 m and 0.2 m, and then averaged to make the results more accurate. The stream cross-sectional velocity profile method was used for the calculation of discharge (Hewlett, 1969). Stream depths with different intervals (10 or 20 cm) based on stream width were measured to calculate stream cross-sectional area. Following formula was used for the calculation of total discharge.

$$Q = \sum_{i=1}^n (W_i * h_i * V_i)$$

where W_i is width of each sub-section (m), h_i is mean depth of each sub-section (m), and V_i is average velocity of each sub-section (m/sec).

Rating curves indicate the relationship between water levels and discharge from each site. Measured discharges taken during each site visit were associated with actual water levels to obtain discharge–water level relationships. These relationships were used to generate discharge time series at 15 minutes intervals by creating rating curves (Appendix A). The web based hydrograph separation model (WHAT) (<https://engineering.purdue.edu/~what/>) was used to estimate direct runoff from each sub-watershed (Lim *et al.*, 2005). This model predicts baseflow

based on observed streamflow data. The hydrograph separation was used to observe the trend in direct runoff under the impacts of upstream disturbances.

Water samples were collected during each significant rain event at each monitoring site. Polypropylene bottles were rinsed with stream water before water samples were collected. Samples were stored in a cooler at 4 °C until they were analyzed. All water samples were analyzed in the School of Forestry and Wildlife Science Laboratory within a week after they were collected. Using the 2540 Total Suspended Solids Method, TSS concentrations were measured. Primarily, this method requires that a standard glass-fiber filter should be washed with 100 ml distilled water, and then the filter should be dried in a 100-105 °C oven for one hour, and finally each sample should be weighed. At least 2 more repetitions of this process are conducted to increase the accuracy of the results. After the filters are washed, 100 ml of well-mixed water sample is filtered through the pre-washed and weighed filters, and then they are dried at least one hour in a 103-105 °C. Eventually, they are cooled for 15 minutes and weighed. This process is repeated 2 more times and the results averaged to improve accuracy. Then, following formula was used for the calculation of TSS concentration.

$$\text{TSS (mg/l)} = [(\text{Mass filter (mg)} + \text{Dried residue (mg)}) - (\text{Mass filter (mg)})] / (\text{Sample volume (ml)} * 1000)$$

Calculated TSS concentrations were used for the prediction of sediment load by running the LOADEST software, which is a FORTAN program designed to estimate constituent loads in streams and rivers. Time series of streamflow and constituent concentration, such as sediment, are required to generate a regression model for the prediction of constituent load in LOADEST (USGS, 2014).

3. RESULTS

3.1. Streamflow

During the study, more than 30 rain events over 12.7 mm (0.5") occurred (Figure 2.4). Total precipitation during the study period was about 1650 mm. In total, eighteen sediment samples were collected at each monitoring station during the study period. Water samples were collected during large rain events over 12.7 mm.

Figure 2.5 and 2.6 show the streamflow time series at W_W and W_E . Rain events smaller than 15 mm during the summer period did not significantly increase streamflow (Figure 2.6 and 2.7). Flow per unit watershed area (defined as specific discharge, i.e. $q = Q/A$, where A is drainage area and Q is discharge) at each site was calculated by dividing the flow at the relevant monitoring site to the upstream area of that site; i.e. q of W_{W1} was calculated by dividing q at W_1 to the area of W_{W1} , while q of W_{W2} was calculated by dividing the q at W_2 to total area of W_{W1} and W_{W2} , and q of W_{W3} was calculated by dividing the q at W_3 to total area of W_W . The specific discharge at each site of W_E was calculated using the same method. The specific discharge at sites W_1 , W_2 and W_3 did not significantly change on W_W (Table 2.1). Average flow per unit area was 0.090 (SE=0.005), 0.082 (SE=0.007), and 0.096 (SE=0.009) l/sec/ha at W_1 , W_2 and W_3 , respectively. At W_E , a slight increase in specific discharge was seen in downstream direction (Table 2.1). Average flow per unit area was 0.055 (SE=0.007), 0.069 (SE=0.008), and 0.074 (SE=0.08) l/sec/ha at E_1 , E_2 and E_3 , respectively.

As stated before, magnitude of rainfall event is the biggest factor driving the hydrology in watersheds. The flow per unit area generated by each site was examined for two significant rain events, 2/21/14 and 4/7/14. The total rainfall depth was 31 mm and 71 mm in 2/21/14 and 4/7/14, respectively. As stated above, average flow per unit area did not change much from upstream to

downstream during the study period at W_w . Flows per unit area during these two rain events showed the same pattern with the average flow in downstream direction. During the rain event of 2/21/14, higher amount of flow per unit area was observed at W_3 , and it was followed by W_1 and W_2 (Figure 2.7). During the rain event of 4/7/14, W_3 produced most water again, and it was followed by W_1 , and W_2 . (Figure 2.8). At W_E , E_3 generated the highest flow per unit area and followed by E_2 and E_1 both during the event of 2/21/14 (Figure 2.7), and 4/7/14 (Figure 2.8).

3.2. Direct Runoff

Both direct runoff and baseflow are the components of streamflow at each monitoring site. Direct runoff is the combination of overland flow and water flow in the top of the soil layer. It is also called quick flow. Direct runoff peaked at all monitoring stations during the rain events (Figure 2.9 and 2.10). Conversely, direct runoff was very low or even absent during the dry periods at each monitoring station. The highest direct runoff per unit area was usually observed at the most downstream monitoring sites of each watershed (Figure 2.9 and 2.10). On W_w , average direct runoff per unit area was 0.038 (SE=0.003) and 0.039 (SE=0.005) l/sec/ha at W_1 and W_2 , respectively during the study period, but increased to 0.062 (SE=0.008) l/sec/ha at W_3 . On the other hand, pattern of direct runoff per unit area only slightly increased from upstream site to downstream site at W_E ; average direct runoff per unit area was 0.036 (SE=0.005), 0.038 (SE=0.006), and 0.044 (SE=0.007) l/sec/ha at E_1 , E_2 , and E_3 , respectively, during the study period.

3.3. Sediment Load

Initial sediment monitoring indicated an active sediment source at upstream sections of the watersheds. Since the sub-watersheds, W_{w1} and W_{E1} are in privately owned lands, the sediment source was unknown. Later, on based on personal communication and visual observation, it was

determined that disturbances were at the south edge of the ponds due to deepening and enlarging of the ponds using heavy equipment, and construction of a residential playground near the pond, on the east side of the pond, at W_{E1} (Figure 2.11). In order to mitigate the impact of the disturbance on water quality, some BMPs such as silt fences and straw bales were used at the south boundary of W_{E1} , however, they appeared to be improperly installed, maintained and insufficient to manage sediment runoff (Figure 2.12).

Most of the sediment load were generated during significant rain events (12.7 mm). At W_w , sediment load was greater at W_1 , and decreased downstream towards W_3 . Average sediment load was 6.76 (SE=0.39), 3.56 (SE=0.34), and 2.01 (SE=0.31) kg/ha/day at W_1 , W_2 , and W_3 , respectively during the study period (Table 2.2). At W_E , different from W_w , more consistent pattern was observed; at W_E sediment load did not change from E_1 towards E_3 (Table 2.2), and average sediment load was 6.94 (SE=1.16), 6.59 (SE=0.97), and 7.23 (SE=1.29) kg/ha/day at E_1 , E_2 , and E_3 , respectively over the study period.

Figures 2.7 and 2.8 show the trend of sediment load under two significant rain events. Higher amount of flow per unit area usually caused higher amount of sediment load across monitoring sites of W_w and W_E . Similar to average sediment load, sediment load per unit area at W_w decreased from upstream to downstream during the events 2/21/14 and 4/7/14. Sediment load per unit area ranged from 10 to 13 kg/ha/day at W_w in 2/21/14, while it ranged from 27 to 41 kg/ha/day in 4/7/14. At W_E , sediment load per unit area showed same pattern with the total sediment load from upstream to downstream during the events 2/21/14 and 4/7/14. It ranged from 18 to 80 kg/ha/day, and from 76 to 128 kg/ha/day during the events 2/21/14 and 4/7/14, respectively.

Kara *et al.* (2014) developed relationships between streamflow and sediment load at each monitoring site using the LOADEST software during pre-harvest period. Using these relationships, we predicted sediment load using the streamflow data from the present study period to get an idea of how average sediment load changed following the upstream urban practices. At W_w , the average sediment load was predicted to be 0.10, 0.08, and 0.14 kg/ha/day at W_1 , W_2 , and W_3 , respectively (Table 2.2). The observed sediment load, on the contrary, was about 14 to 65 times higher than the predicted. At W_E , a similar scenario was examined; observed sediment load was about 20 to 73 times higher than the sediment yield predicted with sediment-discharge relationship of Kara *et al.* (2014).

Sediment concentrations were also scrutinized at each monitoring site. The two watersheds showed different patterns in sediment concentrations from upstream to downstream during the study period. At W_w , sediment concentration was greatest at W_1 , and lowest at W_3 . While sediment concentration decreased from upstream to downstream in W_w (Figure 2.13), it generally increased from upstream to downstream in W_E (Figure 2.14). Lowest sediment concentrations were observed at E_1 , and highest sediment concentrations were monitored at E_3 in W_E .

It appeared that upstream disturbances were completed in early spring 2015. Changes on trend of sediment concentrations in time were also analyzed in order to observe if there was any reduction in concentration following the completion of disturbances at the upstream section. The concentration observations (18 in total) were divided into 2 periods to compare the trend changes between two periods. Period 1 (2/21/14-2/2/15) included first 12 concentration data while Period 2 (7/20/14-5/20/15) included last 12 concentration data because LOADEST requires at least 12 samples to predict sediment yield. LOADEST regressions were developed for periods 1 and 2 at

each watershed. Sediment concentrations decreased through time in both W_W and W_E after the upstream disturbances was completed (Figure 2.15). As can be seen in each graph, the same amount of flow generates lower sediment concentrations during Period 2.

4. DISCUSSION

4.1. Water Yield

On W_W , flow per unit area was higher at W_1 and slightly decreased at W_2 indicating increased water consumption by trees. It has been shown that there is an inverse relationship between vegetation cover and flow (Bosh and Hewlett, 1982), and flow usually increases with decreasing vegetation cover. It was also suggested that increasing tree density with strong root system in riparian zones may result in decreases in stream water velocity by slowing water (Comfort, 2005), and resulting in higher decreases in water yield (Ziemer, 1986). At W_3 , flow per unit area was slightly higher than flow at W_2 , but similar to W_1 , although both W_{W2} and W_{W3} had the same land use; forest. This difference may be attributed to presence of a road located on the north boundary of W_3 . It is possible that surface runoff on W_3 may be higher due to the presence of the road, that may contribute more water to the stream. Roads have the capacity to contribute both surface runoff and intercept subsurface flow (Wemple, 1994). Several studies conducted in Southern US has indicated that forest roads can increase streamflow (Arthur *et al.*, 1998; Swank, *et al.*, 2001; Grace, 2005). Observation of the direct runoff may substantiate this consideration because we found that direct runoff per unit area at downstream stations (W_3 and E_3) on both watershed was higher than upstream monitoring stations.

On W_E , higher flow per unit area was also observed at downstream site (E_3), however, surprisingly, the forested middle section (W_{E2}) generated more flow than upstream section (W_{E1}). This is rather unexpected and difficult to explain. However, our only consideration for

this surprising observation might be the presence of two rills through W_{E2} . A high direct runoff at E_2 can also be attributed to the presence of these rill. Due to open understory, surface flow might have created these rills that further increase surface runoff and streamflow at this section. Overland flow starts as inter-rill flow, and inter-rill flow may turn into rill flow based on topography, surface cover and infiltration rates (Govers, 1991; Robichaud, 2000; Zartl *et al.*, 2001). Thus, rills may result in more erosive energy (Pietrazsek, 2006), and rapid surface incision (Gilley *et al.*, 1990). Development of these rills on W_{E2} might be because of the existence of open understory with no dense vegetation. Decreases on rock and vegetative cover usually increase the runoff velocity (Takken *et al.*, 1998). Another reason for the increasing water yield at downstream sites may be due to convergence of flow. A slight increase in flow per unit area at downstream site (E_3) can also be attributed to higher direct runoff caused by the presence of the forest road.

In the previous study, as mentioned before, SMZ of the W_w was partially harvested, and water yield from W_{W2} and W_{W3} had significantly increased following the harvest (Kara *et al.*, 2015). It was stated that W_{W3} produced more flow per unit area than upstream sections, and W_{W2} generated significantly higher flow than W_{W1} . Before the partial cut, they also reported that W_{W1} and W_{W3} had similar amount of flow per unit area while it was least from W_{W2} (Kara *et al.*, 2012). Our current data suggest that the forested middle section (W_{W2}) recovered, and pattern in flow returned to the pre-harvest conditions in the 6 years following the harvesting. Current data also shows that the roads may be still the main source of runoff to downstream section (W_{W3}).

It should be noted that there was no discharge and runoff data between 5/16/2014 and 5/28/2014 at E_1 due to sediment deposition where the transducer of E_1 was installed during this period. In addition, flow per unit area at W_w seemed to be higher than it was at W_E during the

period between 12/24/2014 and 5/10/2015. We speculate that the pond in W_{E1} may have held the flow before it reached the monitoring sites during this time period.

4.2. Sediment Yield

In each watershed, a substantial amount of sediment load was observed due to presence of upstream disturbances. On W_w , the highest sediment load over the study period was measured at the upstream site (W_1), and the sediment load decreased towards downstream sites (W_2) and W_3). Comparison of our data with the previous study shows the magnitude of the upstream urban disturbances. Kara *et al.* (2014) reported that sediment load was 0.40, 0.84 and 1.56 kg/ha/day at W_1 , W_2 and W_3 , respectively, following the partial cutting within SMZ while recent sediment load ranges from 2.01 to 6.76 kg/ha/day. However, recent data also suggest that pattern in sediment load has changed. Sediment load increased in downstream direction in the previous study (Kara *et al.*, 2014) while it is decreasing in downstream direction now. Decrease in sediment load at downstream sites are probably due to deposition between upstream site (W_1) and downstream site (W_3). More details on the instream erosion and deposition dynamics will be given in Chapter III. The change in pattern in sediment load, and decreasing sediment load in downstream direction may suggest that previously treated SMZ of W_w , and the clearcut area between W_{W3} and W_{E3} have recovered and do not generate additional sediment to the downstream sites. It should be noted that treated SMZ of W_w presents an uneven-aged structure with denser understory.

In W_E , average sediment load during the study ranged from 6.59 to 7.23 kg/ha/day. In the previous study, Kara *et al.* (2014) found that sediment load was higher at the downstream site (E_3) than the middle site (E_2) due to the existence of previous clearcutting. Recent data shows that sediment load did not significantly increase at E_3 suggesting also that vegetation in clearcut

area has recovered and does not contribute more sediment to the downstream site (E₃). Vegetative cover is very effective in reducing runoff and erosion (Freebairn *et al.*, 1986; McIvor *et al.*, 1995). The excessive amount of sediment load measured all monitoring sites on both watersheds points out that most of the sediment was originated from the disturbed area to the downstream sections.

Reductions in sediment concentrations when comparing Period 1 and 2 may be mostly related to intensity of urban disturbances because the first period of concentrations were higher than the second period of concentrations. It suggests that the system may have started to slowly recover after upstream disturbances subsided. Future balance between disturbances and BMPs may determine the future conditions of the disturbed system (Wright and Schoellhamer, 2004). Recovery of study watersheds may take a long time since the decreases in sediment concentration was very slow (Figure 2.15).

Wahl *et al.* (1997) monitored sediment load from an urbanized (11 ha) and forested watershed (37 ha) in South Carolina, and concluded that the urbanized stream generated 66 % more annual sediment load than the forested stream (4.9 versus 3.0 kg/day). In another study, Robert and Pierce (1974) found that compared to forested watershed, urbanization increased suspended sediment concentration by over two orders of magnitude (from 20 to 2400 mg/l) in Patuxent Basin, Maryland. Sediment yield from construction sites in urbanized areas ranges from 104 to 685 kg/ha/day depending on the size of the watershed (Wolman and Schick, 1967). Daniel *et al.* (1979) observed sediment load in 3 residential construction sites in southeastern Wisconsin, and concluded that when compared to an agricultural watershed generating less than a 2.74 kg/ha/day sediment yield, 3 of the construction sites produced average sediment load of 53 kg/ha/day.

Although we have much lower load compared to those studies, this is most probably because only a small acreage of the upstream section was construction site.

It has been commonly recommended that BMPs, especially SMZs, are very effective at protecting water quality from forestry and urban practices (AL Forestry Commission, 1999; Lecay, 2000; Ward and Jackson, 2000; Wynn *et al.*, 2004). Recent data suggested that capacity of incorrectly installed and maintained straw bales and silt fence located around the upstream disturbances in our study area were very poor and inefficient. Straw bales and silt fences are adopted for use in construction sites for erosion and sediment control, but, many applications of straw bales are not successful for erosion and sediment control due to nature of straw bales, problems with placement and installation (Fifeld, 1999). Wishowski *et al.* (1998) evaluated the trapping efficiency of silt fence, and stated that as the particle size of sediment reduces, efficiency of trapping decreases. Silt fences do not usually work effectively because of inappropriate design, installation and maintenance (USEPA, 2012).

Our current data points out the importance of the proper use of BMPs at the entire watershed scale. If there is no or poorly functioning BMPs in one location of the watershed where high sediment is generated, properly installed BMPs in other locations may not be effective when the overall reduction is considered. SMZs are usually excluded from harvesting and site preparation activities (Blinn and Kilgore, 2004). Thus, if SMZs are not effective at protecting water quality, it may be argued that retaining SMZs might be a loss in economic income for the landowners. However, since SMZs have some other benefits such as providing habitat and creating passages for wildlife and controlling flooding (Daniels *et al.*, 2012), SMZs should not be excluded from the BMPs in these watershed.

5. CONCLUSIONS

Even though BMPs are known to be very effective to protect water quality from urban, forestry and agricultural practices, an uncontrolled source of sediment within a watershed may reduce and eliminate the positive effects of these practices in watershed. Dynamics of BMPs are well-understood, but, less is known about the impacts of uncontrolled upstream disturbances on downstream water quality. Sediment yield during a 15-month period was observed on two small watersheds on Marry Olive Thomas Demonstration Forest near Auburn, Alabama. Effects of upstream urban practices on downstream sediment yield were evaluated.

Excess amount of sediment load was observed at downstream monitoring sites of the study watersheds due to intense urban upstream disturbances, even though properly installed BMPs were present downstream. Current data suggests that upstream urban disturbances where BMPs are not present or inadequate can negatively impact downstream water quality regardless of the presence of downstream BMPs. The data also highlight the significance of watershed management as a whole, and the importance of BMP applications in the entire watershed. Future monitoring is also necessary to determine the duration of the impact, and time to return the pre-disturbance period.

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Table 2.1: Average flow per unit area and water yield at each monitoring site

Monitoring Stations	Discharge per unit area (l/sec/ha)	Water yield (mm)
W ₁	0.094	371
W ₂	0.082	323
W ₃	0.096	379
E ₁	0.055	219
E ₂	0.069	271
E ₃	0.074	292

Table 2.2: Sediment yield over the study period at each monitoring site

Monitoring Stations	Observed Sediment Yield (kg/ha/day)	Predicted Sediment Yield (kg/ha/day)	Previous Studies' Sediment Yield (kg/ha/day)
W ₁	6.76	0.10	0.40
W ₂	3.56	0.08	0.84
W ₃	2.01	0.14	1.56
E ₁	6.94	0.34	0.76
E ₂	6.59	0.09	0.35
E ₃	7.23	0.15	0.46

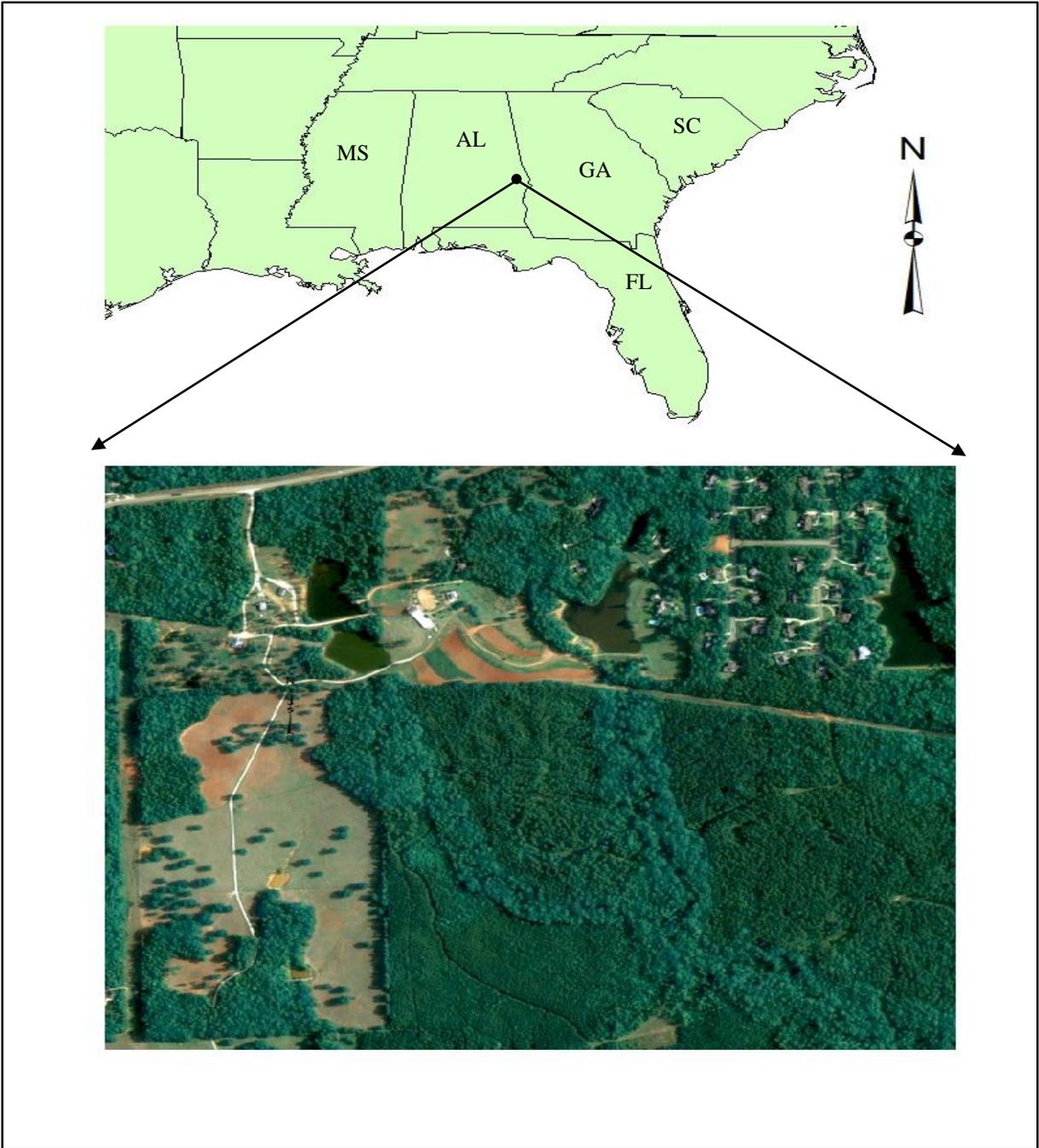


Figure 2.1: Location of Auburn and Mary Olive Thomas Demonstration Forest

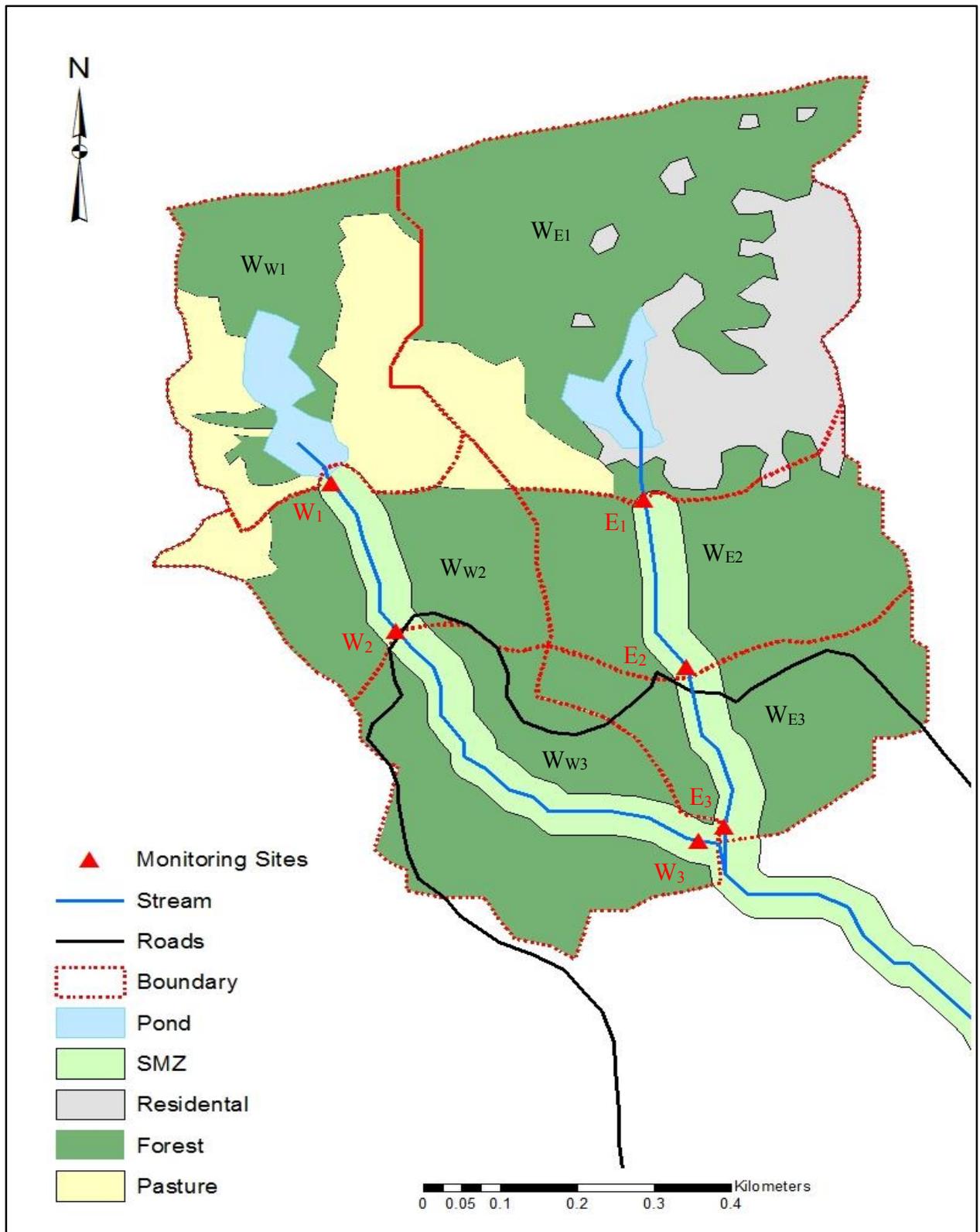


Figure 2.2: LU/LC distribution in study watersheds

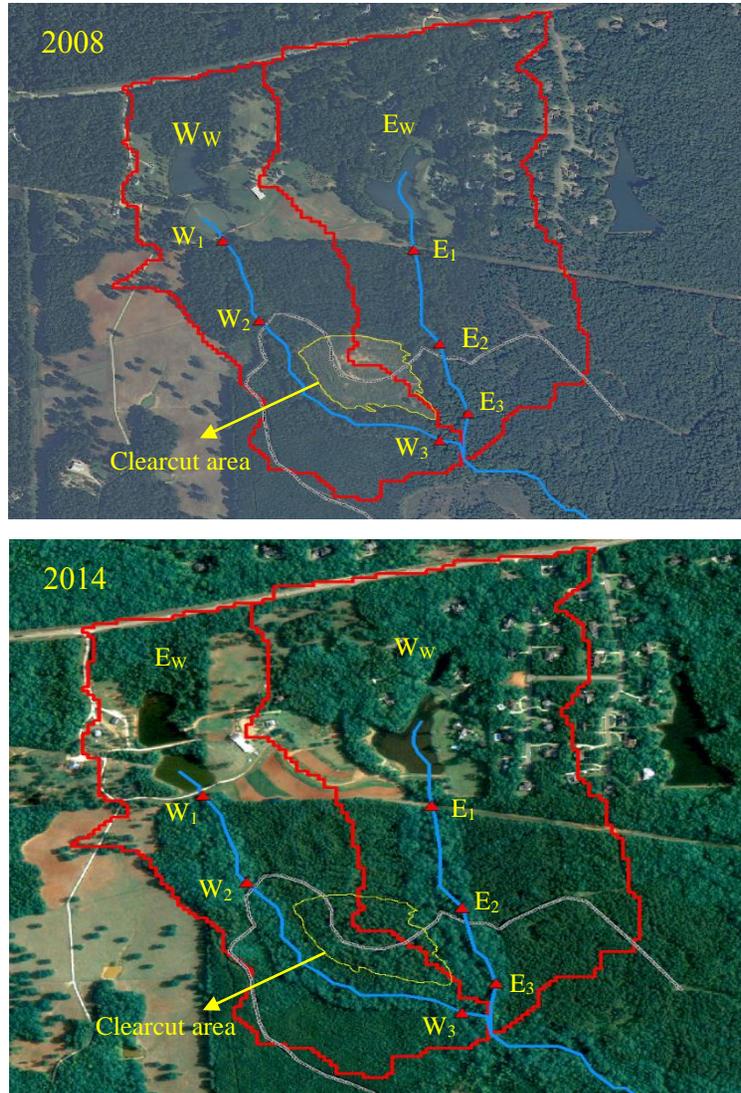


Figure 2.3: Clearcut area in 2008 and 2014

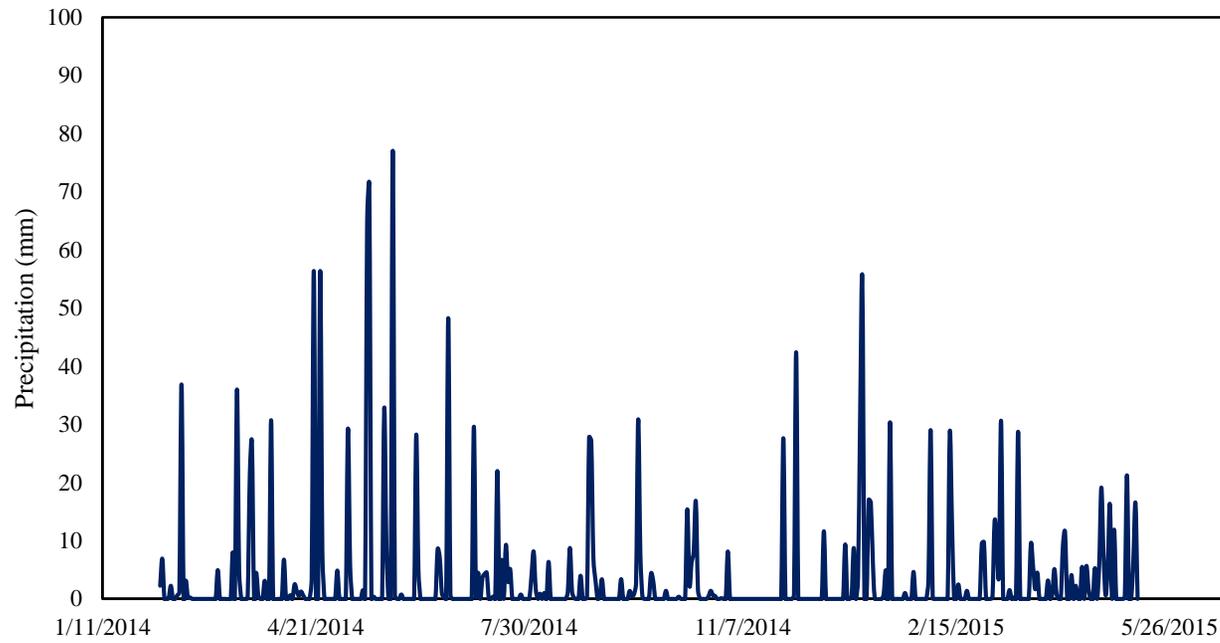


Figure 2.4: Rainfall distribution during the study period (<http://climate.ncsu.edu/>)

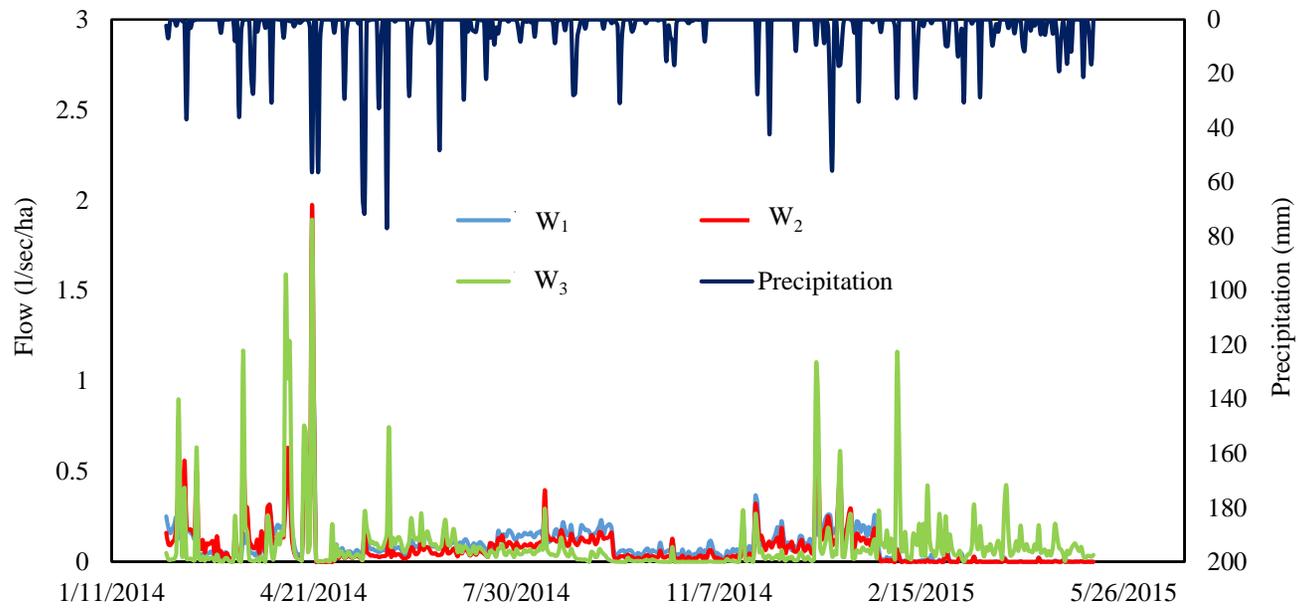


Figure 2.5: Discharge per unit area of West watershed, W_w

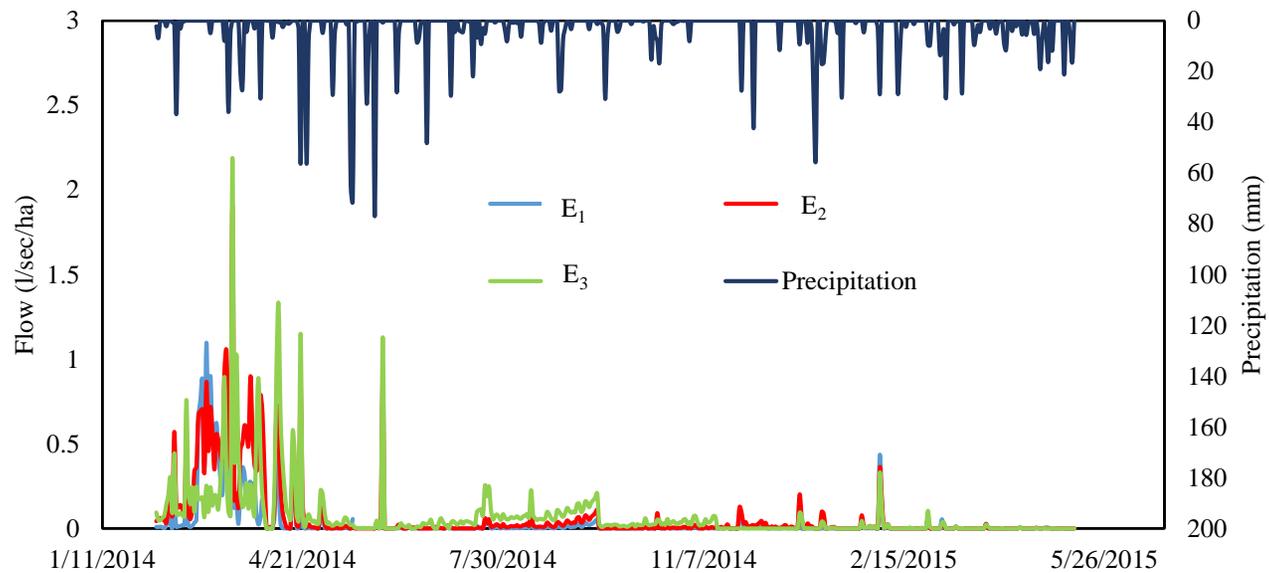


Figure 2.6: Discharge per unit area of East watershed, W_E

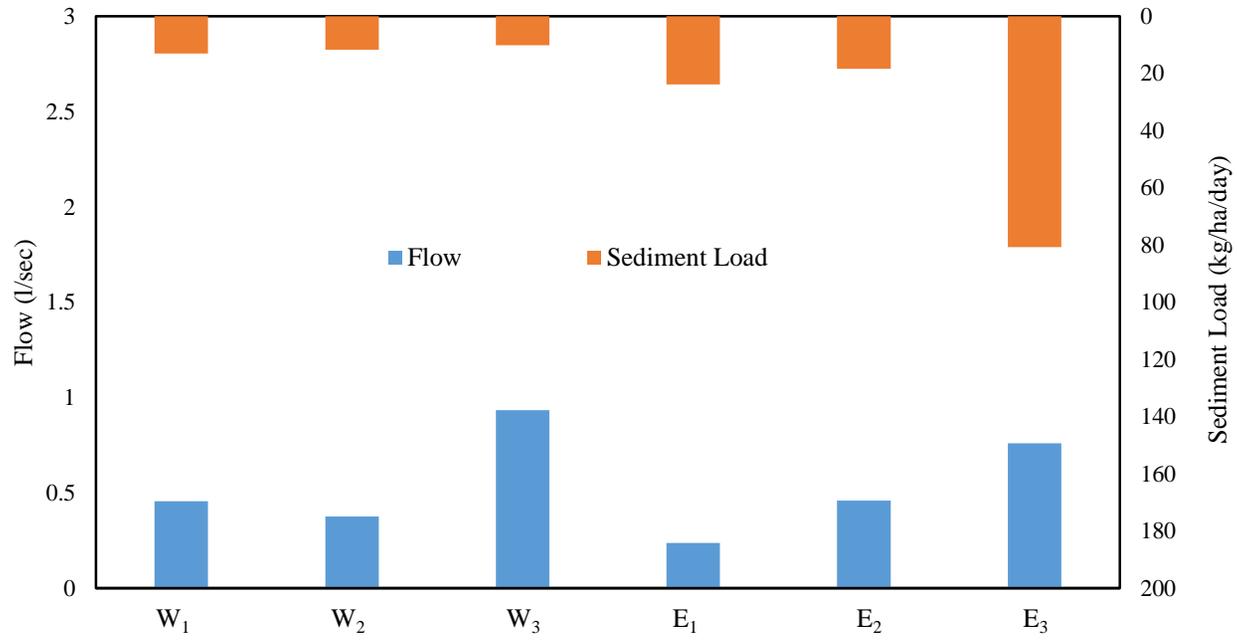


Figure 2.7: Discharge per unit area and sediment load in both W_W and W_E during the event of 2/21/2014

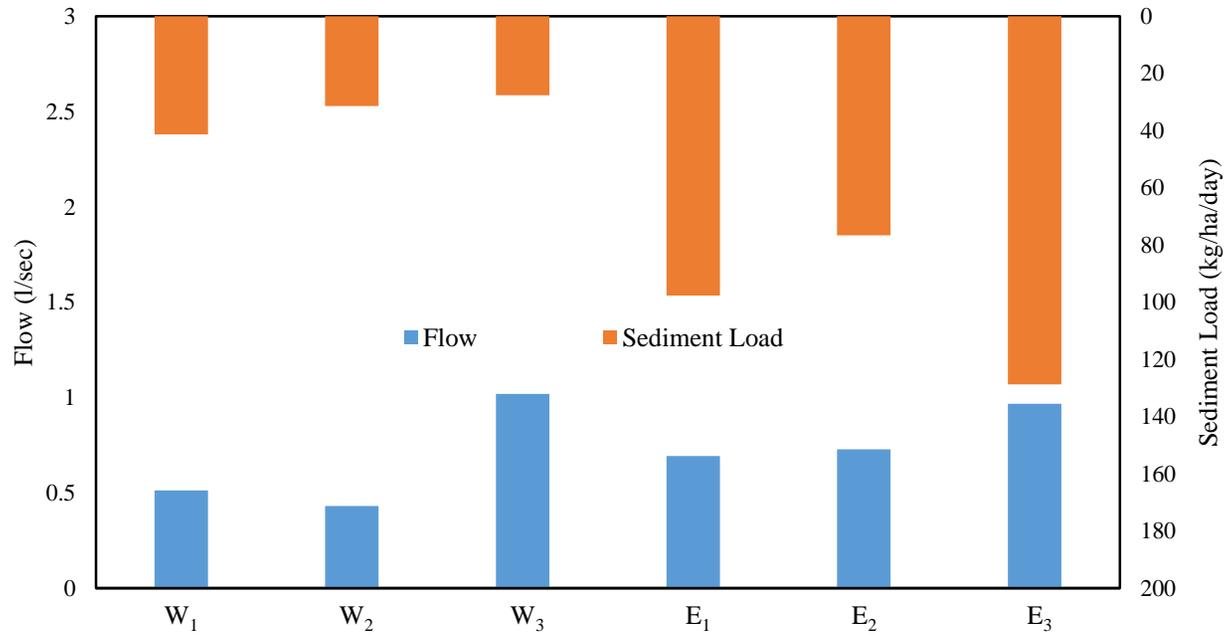


Figure 2.8: Discharge per unit area and sediment load in both W_W and W_E during the event of 4/7/2014

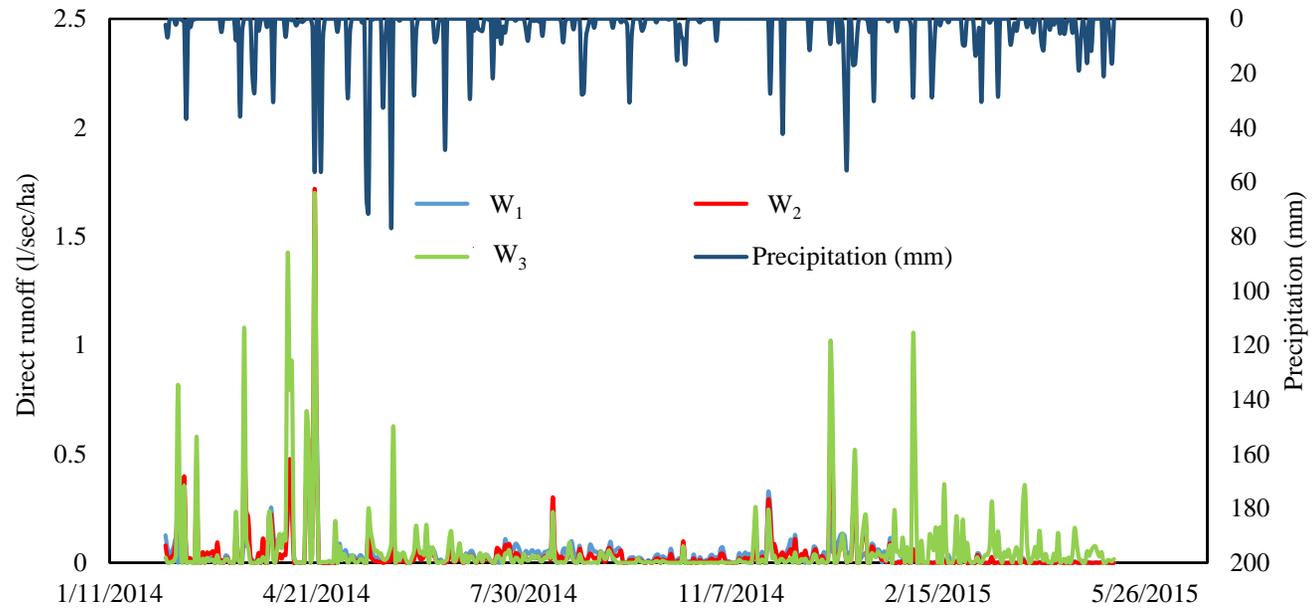


Figure 2.9: Direct runoff per unit area from each sub-watershed in W_w

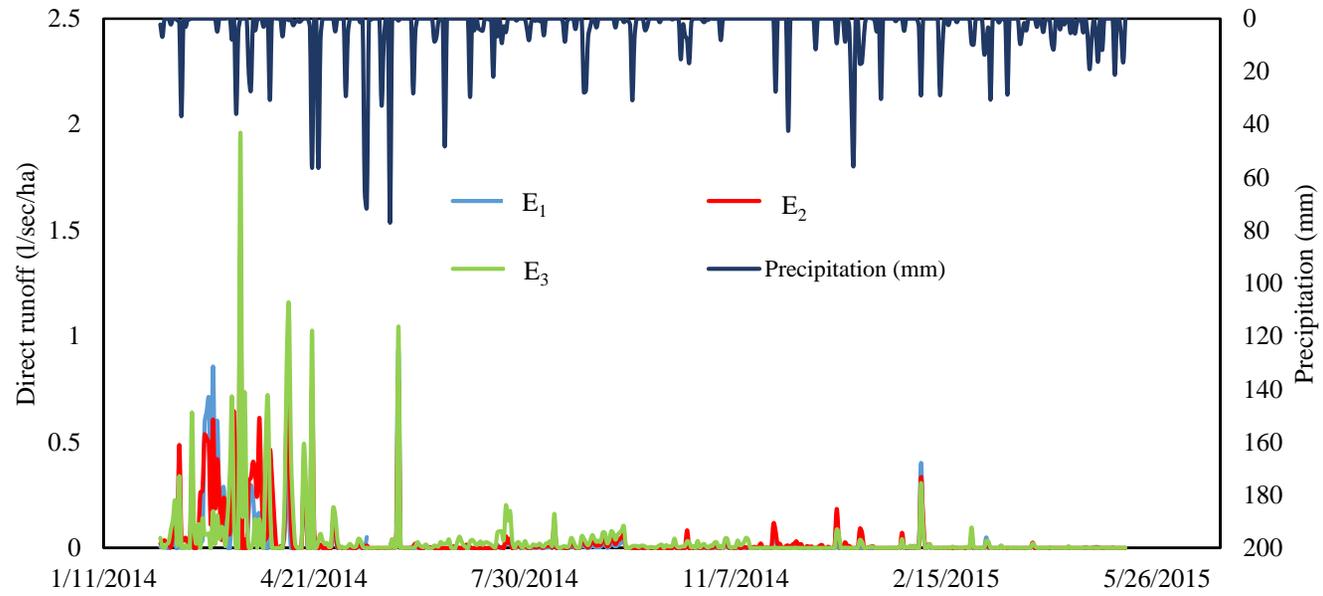


Figure 2.10: Direct runoff per unit area from each sub-watershed in W_E



Figure 2.11: Upstream disturbances; deepening the pond with an excavator



Figure 2.12: Observed silt fence (failed) and straw bale (washed out) that were installed to control sediment and runoff from the construction activities

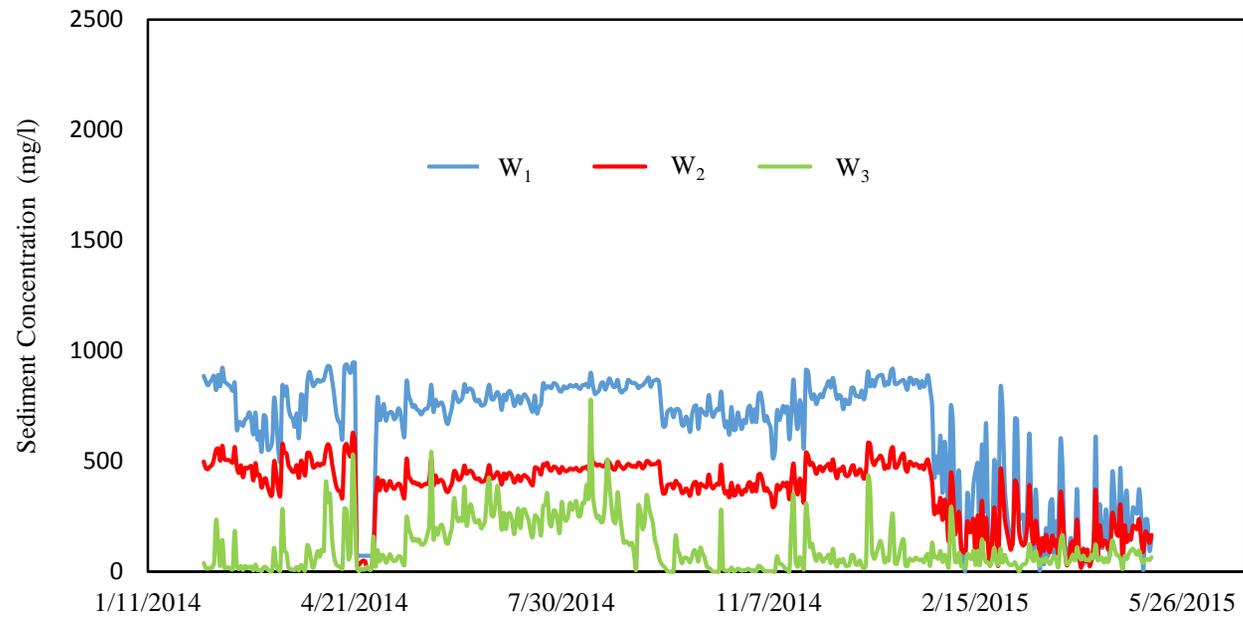


Figure 2.13: Sediment concentrations at the monitoring sites of W_w

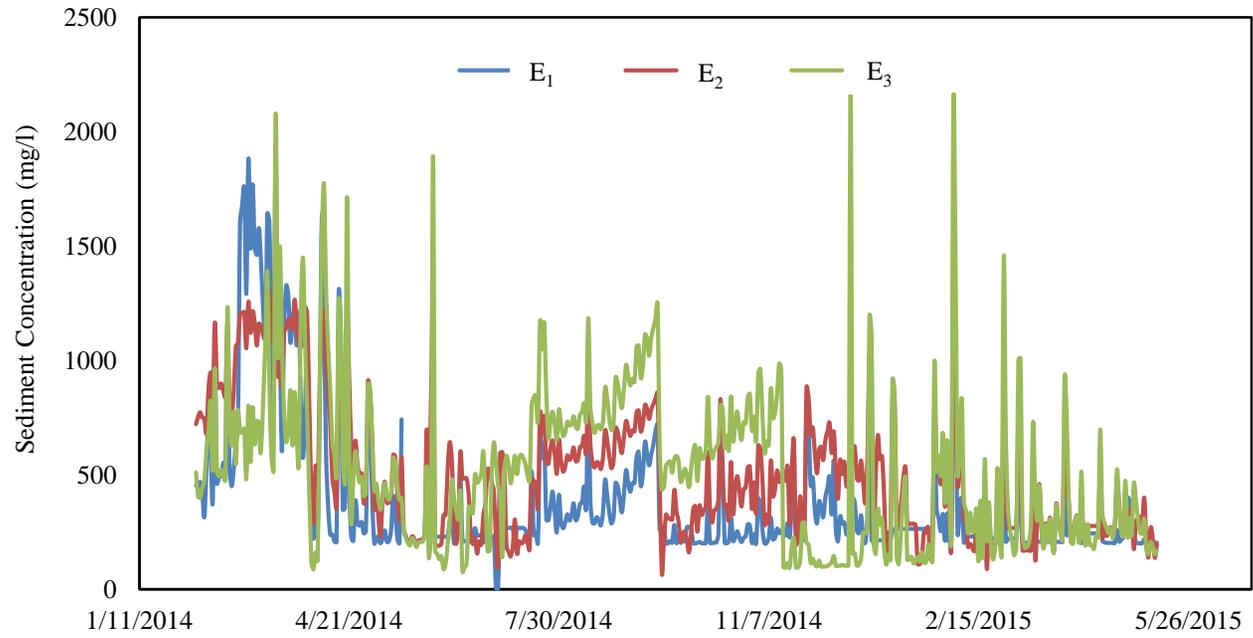


Figure 2.14: Sediment concentrations at the monitoring sites of W_E

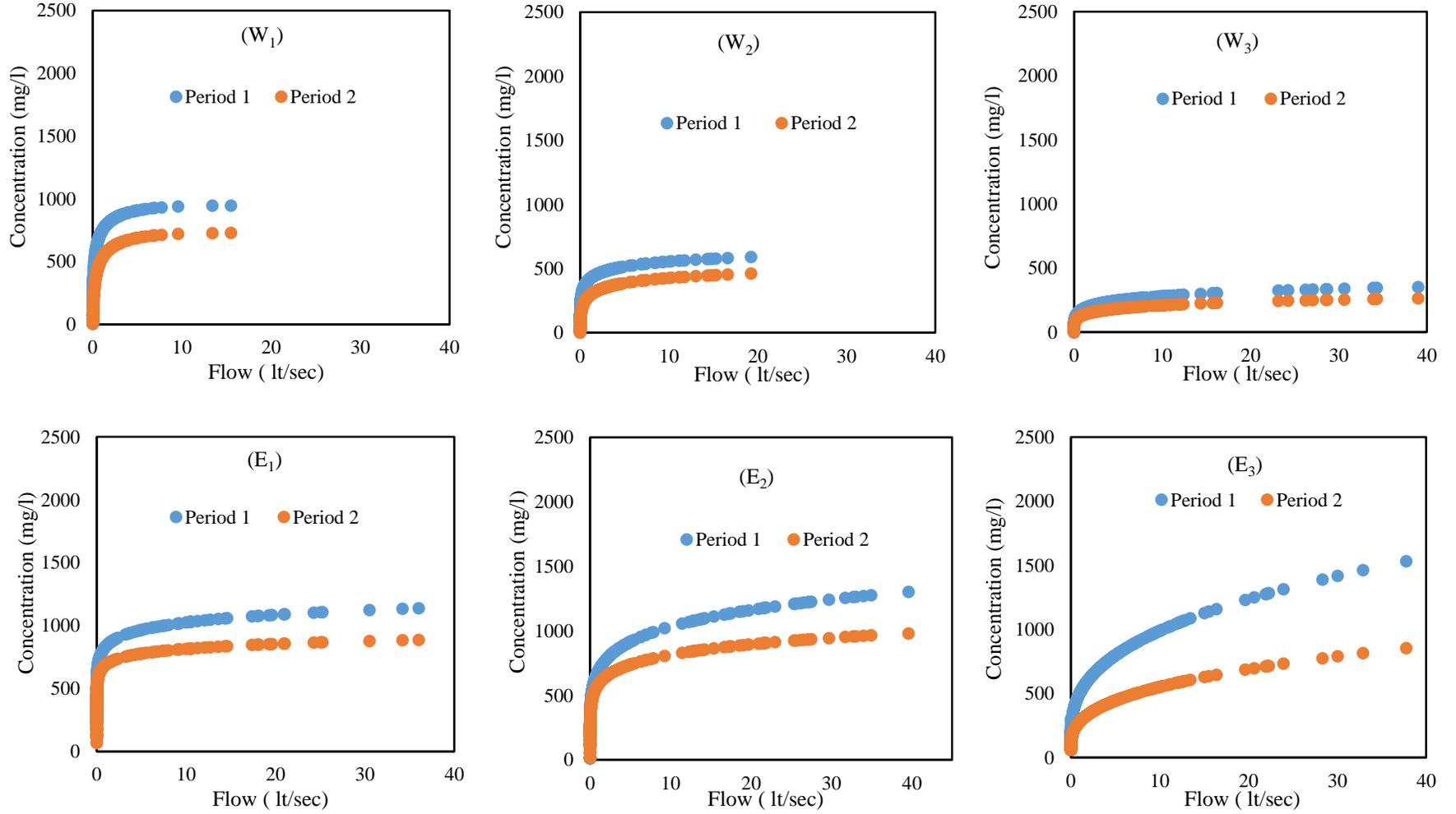


Figure 2.15: Changes in sediment concentration versus flow relationship through time at W₁, W₂, W₃, E₁, E₂, and E₃

CHAPTER III – CHANGES ON STREAM MORPHOLOGY PROTECTED BY BMPs
UNDER THE EFFECTS OF UPSTREAM URBAN PRACTICES

Abstract: Stream channels are usually more stable in forested watersheds. However, intensive land disturbances may disrupt the balance between flow and sediment supply, and result in variations in stream morphology even in the presence of well-designed best management practices (BMPs). This research evaluated the impacts of upstream urban practices on downstream morphology where streamside management zone (SMZs) were present in two small adjacent watersheds in Auburn, Alabama. Field surveys were conducted at 12 stream transects including measurements of channel cross sections at bankfull stage for one and half year. Both instream deposition and erosion were observed in both watersheds as a result of upstream disturbances, but they did not seem to change channel dimensions. Unlike the East stream channel dimensions, channel dimensions of West stream exhibited a very similar pattern with the reference streams within the same physiographic province. SMZs seemed to be inefficient to stabilize streambanks.

Key words: Disturbance, sediment, stream buffer zones, stream morphology, urban.

1. INTRODUCTION

Water has an important role on chemical and physical weathering. Water can decompose rocks, move soil particles, and change the shape of a hillside (Zaimes and Emanuel, 2006). This shows the importance of water in forming landforms as water is transported from watersheds to streams. Streams are natural systems that transport both water and sediment. The transportation of water and sediment are the important functions of streams.

Stream morphology is dynamic, and it can continuously change both in space and time. Streams work toward dynamic equilibrium which means that sediment delivered to the channel is eventually equal to sediment transported and discharged (Zaimes and Emanuel, 2006). A stable stream channel moves towards a state of equilibrium, and reacts physically to the streamflow and sediment coming from upstream (Lane, 1955). Streamflow and sediment supply to a stream are strongly affected by local geology, physiography and climate in its watershed (Montgomery, 1999). Land disturbances, soil types and topography have also impact on stream morphology and flow.

Disturbances on land can have serious effects in fluvial systems since they increase or decrease water and sediment discharge (Boix-Fayos *et al.*, 2007). Stressors may alter the balance between flow and sediment supply resulting in variations in stream morphology (Lane, 1955). Intensive agriculture, overgrazing and deforestation enlarge the stream channels and increase sediment yield (Kondolf *et al.*, 2002; Simon and Darby, 2002). Conversion of forested land to other land uses/land cover (LU/LC) such as pasture may cause accelerated erosion, channel aggradation and widening (Liebault *et al.*, 2005). In addition, deforestation may result in stream incision and widening (Wasson *et al.*, 1998).

Urbanization is another land disturbance that alters sediment dynamics and water discharge. Urbanization causes increasing magnitude and frequency of peak flows, channel degradation and channel widening (Trimble, 1997; Galster *et al.*, 2008), it also increases impervious cover and soil compaction (May *et al.*, 2002). Urbanization increases peak flows because water is transported very fast from impervious surfaces, and therefore, it impacts stream channels by changing channel dimensions, types of bed materials and riparian vegetation (Booth 1990; Avalio 2003; Othitis *et al.*, 2004). The relationship between urbanization and stream morphology is related to different amounts of impervious surfaces (Schueler, 1994). Excessive amount of storm water may be carried to stream channel by roads (Chin and Gregory, 2001), and road crossings may induce bank erosion and influence bed materials of the stream channels (Avolio, 2003).

As a result of external influences such as disturbances, streams may be subject to higher temperatures, sediment and chemical loading, and streambank erosion if best management practices (BMPs) are not used. As a type of BMP, streamside management zone (SMZ) is a vegetated strip of land along a stream or adjacent to a body of water such as lakes (Simpson, 2002). SMZ vegetation is an important management practice for efficiency, and also for maintenance of stream ecosystems (Tait *et al.*, 1994). SMZs control stream temperature (Moore *et al.*, 2005), and serve as a filter and stabilizer (Naiman and Decamps, 1997). Roots in SMZs have a significant role in stabilization of streambanks. Tensile strength of the roots are effective in withstanding shear stresses of streambank soil (Waldron, 1977). SMZ vegetation can also significantly affect stream morphology.

There is a strong relationship between streams and their riparian zones (Studinski *et al.*, 2012). Disturbances such as urban development and harvesting may reduce the benefits gained

by stream buffers (Davies and Nelson, 1994; Sponseller *et al.*, 2008). In this study, the impacts of upstream disturbances on downstream stream morphology where SMZs are present were observed in two small watersheds near Auburn, AL. The objectives of this research are 1) to analyze the alterations in stream morphology under the impacts of upstream urban disturbances, and 2) to observe the capacity of downstream SMZs to stabilize the stream channels structure in the presence of upstream urban practices. It was hypothesized that the responses of downstream morphology to upstream urban disturbances would be variable even in the presence of downstream BMPs.

2. METHODOLOGY

2.1. Study Site

The study was conducted on Mary Olive Thomas Demonstration Forest near Auburn, AL that was described in detail in Chapter 2 (Figure 2.1). Study area is comprised of 2 small adjacent watersheds, West and East Watersheds (W_W and E_W , respectively). Details of the sub-watersheds, monitoring stations used for obtaining stream stage measurements, and land use of the study watersheds were lay out for both watersheds in Chapter 2 (Figure 2.2). Intensive urban disturbances located on upstream sections of the study watersheds (W_{W1} and E_{W1}) can be seen in the figure 2.12 in Chapter 2.

2.2. Channel Transects

Field measurements of channel cross sections/transects were conducted in order to understand the impacts of urban disturbances on the stream morphology. Survey sites were determined for each stream in W_W and W_E . The sites which are fast water habitat units in straight sections of the reaches were preferred. Sites with bankfull stage indicators were selected. Six survey sites were determined for each watershed, and in total 12 sites were selected to make

cross section measurements. They were named from upstream to downstream as T_{W1} , T_{W2} , T_{W3} , T_{W4} , T_{W5} and T_{W6} in W_W , and T_{E1} , T_{E2} , T_{E3} , T_{E4} , T_{E5} and T_{E6} in W_E (Figure 3.1).

The standard topographic survey methods were followed during measurements of stream cross sections with permanent site marking stakes, a string line and level, a measuring tape, engineering survey flags and a depth rod (Harrelson *et al.*, 1994). A horizontal measuring line, which is a string, was first established across the channel between permanent end point markers, so that they do not wash out over time (Figure 3.2). The height of the string line on the end point markers was recorded so that future cross section surveys could be made at same elevation. The measuring tape through the string line was placed to read distances in which depth readings are recorded (Figure 3.3). Left bank facing the downstream was always the starting point for measurements in each survey site. Depth readings were taken by using rod at each intervals. Intervals were determined according to 5-10 % of the total width measured at all survey sites.

Cross section measurements were repeated through time following relatively big rain events larger than 2.54 cm (1") since we believe that possible changes in the stream channel may be attributed to strong rain events and upstream urban practices in this study. Stream cross sections were plotted on a grid paper using a scale of 1:20 to calculate the deposited and eroded areas for each cross section.

Deposited and eroded areas in a cross section were traced separately since a survey site may be exposed to both erosion and deposition simultaneously. For instance, T_{E5} was subject to strong changes during the period of 7/5/14-7/29/14. The survey site included both instream erosion and deposition (Figure 3.4). Deposited and eroded areas were used to estimate the amount of sediment transported or eroded between successive channel transects in downstream direction.

After the average of deposited and eroded areas were calculated for each cross section over time, they were multiplied with the distance between the two transects. The distances and slopes between $T_{W1}-T_{W2}$, $T_{W2}-T_{W3}$, $T_{W3}-T_{W4}$, $T_{W4}-T_{W5}$, $T_{W5}-T_{W6}$, and $T_{E1}-T_{E2}$, $T_{E2}-T_{E3}$, $T_{E3}-T_{E4}$, $T_{E4}-T_{E5}$, $T_{E5}-T_{E6}$ are indicated in table 3.1.

To calculate the total volume of deposited and eroded sediment between $T_{W1}-T_{W2}$, the total deposited and eroded area over time at T_{W1} and T_{W2} were first calculated, and then averaged. The average of eroded and deposited areas were then multiplied with the distance between T_{W1} and T_{W2} . This process was repeated between each cross section from upstream to downstream. The difference between the amount of deposited and eroded sediment between 2 cross sections, called 'net sediment deposition', was calculated, and it indicated the dominant process between the channel transects. It was accepted that when the net sediment deposition is less than 0 or more than 0, the dominant process between these cross sections is erosion or deposition, respectively.

2.3. Stream Classification

Streams are classified to understand the stream condition and response of the stream channels under the impacts of land disturbances. Rosgen Classification System is the most preferred and applied classification system in the USA (Juracek and Fitzpatrick, 2003). The Rosgen Classification System uses bankfull discharge dimensions for measurements (Rosgen, 1994). The system classifies streams based on channel morphology. Three more cross sections were determined and surveyed for the measurements of stream morphology dimensions from each stream channel. The following order was used for classification of streams:

- Determination of single or braided channel,

- Entrenchment ratio,
- Width to depth ratio,
- Sinuosity,
- Slope ranges, and
- Median size of bed material.

a) Single or braided channel determination

Braided streams have number of alluvial channels (Lane, 1957). Field visits indicated that both streams in the study area have a single channel.

b) Entrenchment Ratio

The ratio of width of the flood-prone area to the width of the bankfull channel is called entrenchment ratio (Rosgen, 1994 and 1996). It is used for determination of channel incision. Bankfull width and flood prone area width were first determined for the streams in W_W and W_E . Since erosion, sediment transport and bar building as a result of deposition are most efficient when discharges are near the bankfull, alluvial channels are mostly controlled by bankfull discharge (Harrelson *et al.*, 1994). Field indicators of bankfull stage is variable, thus identification of bankfull stage could be difficult in the field. Active floodplains are the main indicator of bankfull stage; but in the absence of active floodplains, there are some other indicators to identify bankfull stage such as; bank slope, vegetation, soils, point bars and undercuts (Harrelson *et al.*, 1994).

Bankfull indicators, point bars, undercuts and vegetation were marked with survey flags along the left and right bank of streams in both watersheds. A measuring tape was stretched perpendicular to the flow direction from the point of the left bank to the point of the right bank

where bankfull indicators were identified. The width between two points were recorded. All of the bankfull width measurements were taken over a stable riffle in both stream channels. The maximum bankfull depth was also measured with a rod at each location where bankfull width was recorded. The maximum bankfull depth is the distance between the tape and the deepest point on channel bottom. In order to calculate bankfull cross sectional area (A_{BKF}), the depth between the channel bottom and the tape was measured with a survey rod at equally spaced intervals across the stream channel. Then, the depth measurements were averaged to determine a mean bankfull depth for both streams. The following formula was used for the calculation of A_{BKF} (Harman, 2000).

$$A_{BKF} = \sum_{i=1}^n (W_i * h_i)$$

where W_i is the width of each increment, and h_i is the mean depth of each increment.

The flood-prone width was measured at the elevation of twice the maximum depth at bankfull (Rosgen, 1994 and 1996). The measuring tape was stretched across the channel until we intersected the next adjacent terrace or hillside on both side of the channel. This total distance gave the flood-prone width. Eventually, flood-prone width was divided by bankfull width to determine entrenchment ratio of streams.

c) **Width to Depth Ratio**

Bankfull width was divided by the mean bankfull depth to determine width to depth ratio of streams in the study area (Rosgen, 1994 and 1996).

d) Sinuosity

Sinuosity was calculated as channels length divided by a straight line valley length (Rosgen, 1994). A 10 m Digital Elevation Model (DEM) was used to identify stream lengths and straight line valley lengths on ArcGIS. Rivers with a sinuosity lower than 1.1 are classified as straight while those between 1.1 – 1.5 are sinuous, and ones with higher than 1.5 sinuosity are meandering (Leopold and Wolman, 1957).

e) Slope Ranges

The bed slope of streams were determined by using a 10 m DEM with ArcGIS. Bed slope of the streams was used as a surrogate for water surface slope in this study. Change in elevation between upstream and downstream was divided by the length of the stream to identify the slope of both stream channels.

f) Median Size of the Bed Material

The composition of streambed materials and banks is a key element for stream character. It affects channel stability, erosion rates and sediment supply sorted throughout the stream channel (Doll *et al.*, 2003). As a result of management practices or natural disturbances, particle size distribution may change over time. The impacts of upstream disturbances on bed materials of streams were documented by quantitative description of the bed material (Wolman, 1954).

The Wolman pebble count method analyzes in total 100 pebble samples along the stream reach (Wolman, 1954). This technique required 2 people: one observer was needed for reading with a ruler, and the other one was needed for taking note with a notebook. The steps below were followed during pebble count measurements:

- Both streams in W_W and W_E were first characterized to identify pools and riffles. The percentages of total length of the profile which are riffle and pool were computed. The percentages were used to decide on the number of pebbles collected at these features.
- The particles were sampled in a cross section perpendicular to the flow direction from left bankfull to right bankfull in transect.
- By averting the eyes, the first particle was collected while reaching down over the tip of the boot.
- Intermediate axis of each particle (Figure 3.5) collected at the bottom of the stream was measured, and noted to pebble count data sheet (Figure 3.6).

This process was repeated by taking one step ahead in direction of the right bank until the required number of measurements were recorded. In total 100 particles were collected in both streams. Then, the D_{50} of the bed material was determined for both streams. The D_{50} describes the median particle size, which means that 50 % of the sampled material is finer than the representative particle diameter (Bunte and Abt, 2001). For both streams, a cumulative frequency plot of the particle size distribution was created, and it provided the D_{50} for West and East stream.

Bank Height Ratio (BHR) of both streams was also calculated to estimate the degree of channel incision (Rosgen, 2001). The ratio of lowest bank height of the cross section to maximum bankfull depth gives the BHR of a stream (Rosgen, 2001) (Figure 3.7). If the BHR is between 1.0 – 1.05, 1.06 – 1.3, 1.3 – 1.5, or >1.5, than the stability rating is stable (low risk of degradation), moderately unstable, unstable (high risk of degradation), or highly unstable, respectively.

3. RESULTS

3.1. Channel Transects

Stream cross sections were surveyed after significant rain events 2.54 cm (>1") from 3/31/2014 to 7/26/2015. In total, 8 cross section measurements were taken at T_{E1}, T_{W4}, T_{W6}, T_{E1}, T_{E4} and T_{E6} during this period. The other sites, T_{W2}, T_{W3}, T_{W5}, T_{E2}, T_{E3}, and T_{E5} were surveyed 6 different times in total. These cross sections were added after T_{W1}, T_{W4}, T_{W6}, T_{E1}, T_{E4} and T_{E6} were installed since more survey sites were needed to obtain more precise results.

Both West and East stream channel exhibited a dynamic pattern. Instream erosion and deposition were observed in all survey sites in W_W (Figure 3.8) and W_E (Figure 3.9) over time. While some of the survey sites experienced either only instream deposition or erosion, most of the survey sites were simultaneously exposed to instream deposition and erosion after a significant rain event.

More sediment was transported in the West stream than the East stream along the stream cross sections (Table 3.2). Net instream erosion and deposition between the cross sections are shown in Figure 3.10. Compared to instream erosion, deposition was dominant among all of the survey sites in W_W. The rate of sediment deposition evidently exceeded the rate of instream erosion at West stream. While the total amount of eroded sediment along the stream channel of W_W was 8.6 tons/ha, the total amount of deposited sediment was 13.3 tons/ha over time. The net sediment deposition was +4.8 tons/ha in West stream channel (Table 3.2).

On E_W, the total amount of eroded sediment was 4.3 tons/ha while the amount of deposited sediment was 5.2 tons/ha along the channel transects. Although instream deposition was the dominant process between some of the survey sites, the downstream survey sites were exposed

to instream erosion rather than instream deposition. The net amount of deposited sediment was +0.9 tons/ha (Table 3.1).

3.2. Stream Classification

Bankfull area, bankfull width, bankfull mean depth, flood-prone width, width to depth ratio, entrenchment ratio, and BHR were measured/calculated in 3 cross sections in each stream channel, separately (Appendix B). Based on cross sectional measurements, West and East stream indicated distinct morphologic characteristics. Stream morphology characteristics for each stream can be seen in Table 3.3.

Both streams showed similar types of channel habitat units. The four types of stream channel habitat units, which are pool, glide, riffle and run, were present in both streams. Particle size distribution indicated that both streams had different channel bed material (Appendix C). Size frequency of both streams are indicated in Figure 3.11 and 3.12. D_{50} was 0.7 mm for the West stream while it was 11.5 mm for the East stream (Figure 3.11 and 3.12). This means 50 % of the sampled bed material is finer than 0.7 mm in the West stream channel while 50 % of the sampled bed material is finer than 11.5 mm in the East stream channel. Thus, the bed material of West stream and East stream is coarse sand and medium gravel, respectively. Eventually, streams types of West and East streams were determined as E5 and F5, respectively, based on Rosgen Stream Classification System (Rosgen, 1994) (Figure 3.13).

4. DISCUSSION

4.1. Channel transects

Both West and East stream channel were subject to significant instream deposition and erosion over time even though the stream channels were surrounded by forest. Stream channels

are normally stable in forested watersheds (Miner, 1968). The instability of the streams in this study was strongly associated with the upstream urban disturbances. Both streams were affected by the alterations in upland runoff and sediment supply. Increase or decrease in the amount of sediment delivered from upstream may result in deposition and/or erosion with degradation and/or aggradation of the stream bed with ensuing changes in channel morphology (Speiran, 1996). Urban practices within a watershed have a dramatic effect on water and sediment discharge (Boix-Fayos *et al.*, 2007), and cause higher peak flows by creating imbalance of forces in the channel (Schumm, 1977). Peak flows from construction sites may exceed by 2 to 6 times the peak flows from the same area prior to urban development (Carter, 1961), and this may strongly influence stream morphology.

While instream deposition was the dominant process in the West stream channel, deposition and erosion were almost in balance in the East stream channel across the cross sections. This might be due to the difference between stream water velocities in the two stream channels. East stream flow was always faster than West stream flow throughout the study period since its channel is narrower than West stream channel (Figure 3.13). In case of decreases in water velocity, streams may not be able to transport the material that it carries anymore. It will start to drop the particles to its bed and keep them depositing (Peterson *et al.*, 2015). Observed sediment concentrations at the upstream monitoring stations can also explain why west stream channel was subject to much more instream deposition. Average sediment concentrations observed at W_1 and E_1 were 647.2 (SE=12.1) mg/liter and 431.3 (SE=15.5) mg/liter, respectively. This indicated that more sediment entered, transported and deposited through the West stream channel than the East channel as a result of the upstream urban disturbances.

The significance of net instream deposition and erosion between channel transects varied along both stream channels. Slope variations through the stream channel may have played an important role in the increases/decreases on deposition and erosion in the stream channel. At W_w , the slope between T_{W3} and T_{W4} is very high compared to the slopes between other channel transects (Table 3.1), and the lowest deposition was observed between T_{W3} and T_{W4} . The slope and instream deposition between remaining transects did not change significantly at the West stream channel. Instream deposition along the East stream channel may have caused the decreases on sediment concentration in downstream direction. At E_w , the slope is lower between the upstream channel transects, and it increased through downstream. This may have contributed to instream deposition between upstream channel transects, and erosion between downstream channel transects.

This study included E and F Rosgen types streams based on the measurements of entrenchment ratio, and width depth ratios. The E type streams are less incised with low width/depth ratios and they exhibited high channel sinuosities (Rosgen, 1994). On the other hand, F type streams are characterized by very high width/depth ratios, and they can develop accelerated channel aggradation/degradation (Rosgen, 1994). Brantley *et al.* (2013) collected stream morphology data at 21 reference streams in Piedmont Alabama, and then they classified those reference streams based Rosgen Classification System. The drainage areas of the reference streams ranged from 0.2 to 242.1 km². Their study included Rosgen stream types B, C and E. When our morphologic data (bankfull cross sectional area, bankfull width, bankfull mean depth and slope) were compared with the morphologic data of these reference streams, we found very similar values that substantiates our field measurements.

Rosgen type E streams exhibit very stable channels because they are surrounded by well-developed floodplains with vegetation (Ward *et al.*, 2008). Although West stream was categorized as type E, local downcuttings as a result of upstream disturbances may have caused in stream erosion and deposition along the stream channel. Conversely, type F streams are incised streams (Ward *et al.*, 2008). East stream was classified as type F, and it developed instream channel deposition and erosion during the study. There was no type F stream among the 21 Alabama piedmont reference streams studied by Brantley *et al.* (2013). The difference in the morphology of the East stream may be associated with the legacy effects of sediment from historical land use disturbances. Streams of Southeastern Piedmont have been exposed to excess sediment during and end of the 1800's by poor agricultural practices (Richter and Markewitz, 2001). Then, cotton farming was abandoned and agricultural fields were converted to pasture and forests (Trimble, 1974). These disturbances were followed by channelization. These changes resulted in channel incision and accelerated bank erosion (Ruhulman and Nutter, 1999). East stream may still be in the process of transporting legacy sediments as a result of land use disturbances while west stream may have reached a stable equilibrium condition.

Bed material of the two streams was also different from each other. Bed material of the West and East streams were coarse sand and medium gravel, respectively. Brantley *et al.* (2013) found that most of the 21 reference streams in Alabama Piedmont had gravel bed material. They also observed some reference streams with a sandy bed material. The difference of bed material between West and East stream may be attributed to water speed. East stream with higher velocities may have carried finer bed material to downstream, and coarser material may have remained in the stream bed. In addition, disturbances may also have been more severe in W_w as it was observed that W_1 had approximately 50 % higher sediment concentration than E_1 . Another

consideration may be the sequence of the disturbance. Construction usually brings sandy texture sediment to the streams. Thus, since upstream disturbances were first started in W_E , sandy sediment in East stream may have been washed out over time, and coarser sediment may have remained in the streambed of the East stream. On the other hand, sandy sediment in West streambed may not have been carried out yet due to the later start of upstream disturbances in W_w .

Although streambank erosion is not the main focus of this study, it is a very significant contributor to stream instability. BHR ratios of the both streams indicated that streambank of the West and East stream are moderately unstable and highly unstable, respectively. Composition of riparian vegetation determines the susceptibility of streambank to erosion. Vegetation can be very effective in providing bank stability and roughness on the floodplain in small drainage areas (Keller and Swanson, 1979). Riparian vegetation contributes hydraulic flow resistance and erosional resistance, which are key in maintaining channel stability (Buttle, 1995). SMZs serve as a filter and stabilizer for streambank (Naiman and Decamps, 1997). Even though both streams are covered by forest and SMZs with woody species along the streams, their streambank are under the risk of erosional processes.

5. CONCLUSION

Use of BMPs in mitigating the impacts of human induced disturbances on stream morphology have been studied for many years. As a result of external influences, streams may be subject to higher temperatures, sediment loading, and streambank erosion if BMPs are not present. However, the capacity of BMPs in maintaining stream morphology under the unrestrained disturbances may be questionable. Effects of intensive upstream disturbances on the morphology of two small streams were observed for 18 months in Marry Olive Thomas

Demonstration Forest near Auburn, Alabama. Excessive amount of sediment as a result of upstream urban disturbances entered stream channels, and disrupted the equilibrium between flow and sediment transported along the stream channels. Upstream disturbances did not initially change the channel dimensions of both streams, but both stream channels were exposed to instream erosion and deposition. Current data suggested that intensive upstream disturbances may negatively influence downstream morphology where BMPs are in place. It also suggested that uncontrolled disturbances may negate the benefits gained by SMZs. This study emphasizes the necessity of watershed management in a holistic fashion in order to mitigate the impacts of disturbances more efficiently. Future studies are also required to evaluate the recovery of the system following the completion of the disturbances.

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Table 3.1: Distances and slopes between channel transects

Reaches	Slope (%)	Distance (m)	Reaches	Slope (%)	Distance (m)
T _{W1} -T _{W2}	1.4	85	T _{E1} -T _{E2}	2.4	85
T _{W2} -T _{W3}	2.1	100	T _{E2} -T _{E3}	1.2	85
T _{W3} -T _{W4}	2.7	85	T _{E3} -T _{E4}	1.1	65
T _{W4} -T _{W5}	1.3	200	T _{E4} -T _{E5}	2.2	100
T _{W5} -T _{W6}	1.4	240	T _{E5} -T _{E6}	1.6	100

Table 3.2: Amount of instream deposition and erosion over the period (3/31/2014-7/26/2015) at W_E and W_E

Reaches	Erosion(tons)	Deposition(tons)	Net(tons)	Reaches	Erosion(tons)	Deposition(tons)	Net(tons)
T _{W1} -T _{W2}	41.4	96.2	+54.8	T _{E1} -T _{E2}	39.8	49.8	+10.0
T _{W2} -T _{W3}	63.2	96.8	+33.6	T _{E2} -T _{E3}	32.2	49.4	+17.2
T _{W3} -T _{W4}	54.0	59.5	+5.5	T _{E3} -T _{E4}	17.4	43.4	+26.0
T _{W4} -T _{W5}	83.1	136.6	+53.6	T _{E4} -T _{E5}	58.6	50.8	-7.8
T _{W5} -T _{W6}	85.8	120.7	+34.9	T _{E5} -T _{E6}	66.8	65.5	-1.3
Total (tons)	327.4	509.9	182.4	Total	214.7	258.9	44.1
Total (tons/ha)*	8.6	13.3	+4.8	Total (ha)	4.3	5.2	+0.9

**Total is divided by watershed drainage area.*

Table 3.3: Stream morphology characteristics for West and East stream

	West	East		West	East
Bankfull Area (m ²)	0.7 - 1.5	0.6 - 1.02	Drainage Area (km ²)	0.3	0.5
Bankfull Width (m)	2.7 - 3.7	2.7 - 4.3	Sinuosity	1.6	1.3
Bankfull Mean Depth (m)	0.3 - 0.6	0.2 - 0.4	Water surface slope (m/m)	0.019	0.014
Width to Mean Depth Ratio (m/m)	9.0 - 11.5	12.0 - 17.1	BHR	1- 1.2	1.9 - 4
Flood-prone Width (m)	13.4 - 24.4	4.3 - 5.2	Rosgen Stream Classification	E5	F4
Entrenchment Ratio (m/m)	3.6 - 7.4	1.2 - 1.6			

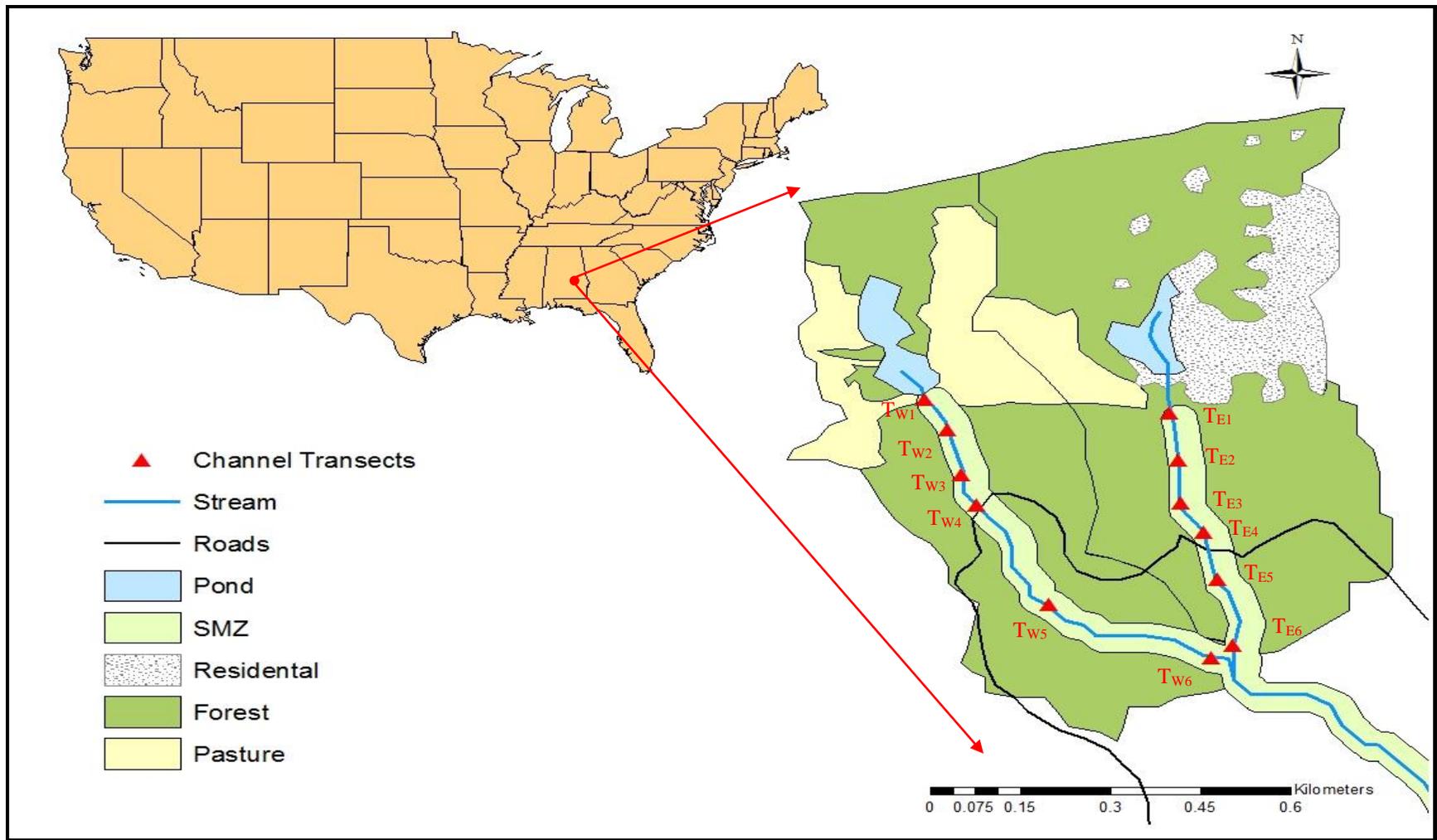


Figure 3.1: Channel transects surveyed in W_w and W_e



Figure 3.2: Permanent start and end points in a cross section



Figure 3.3: Measuring tape on the string line

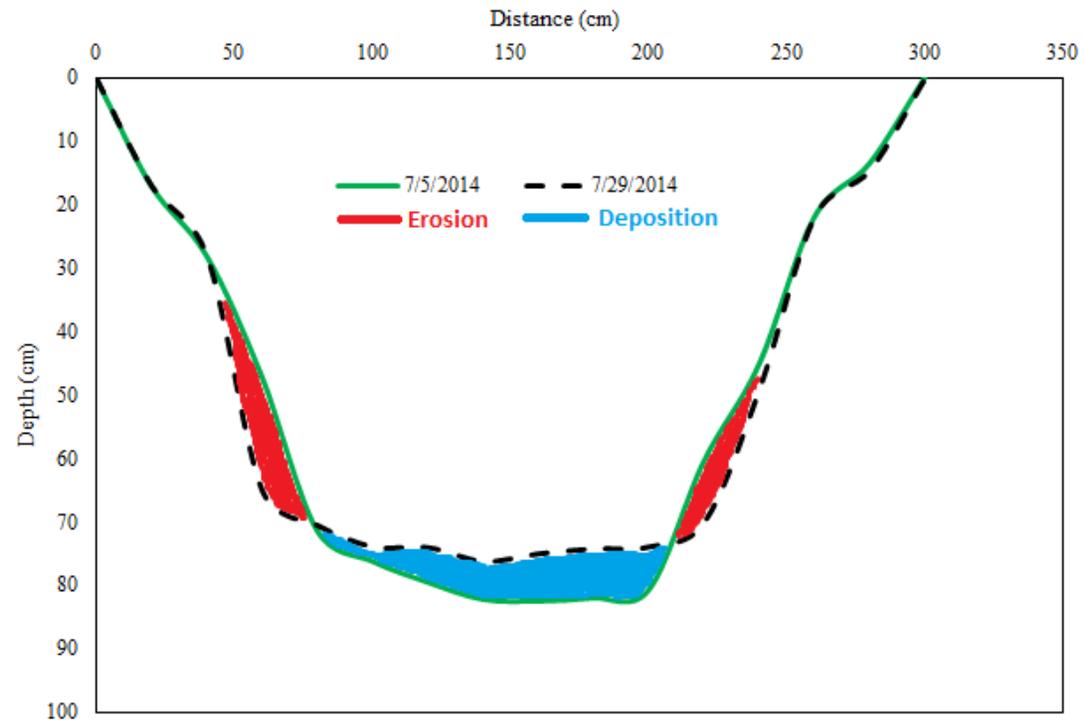


Figure 3.4: Changes at T_{E5} from 7/5/2014 to 7/29/2014

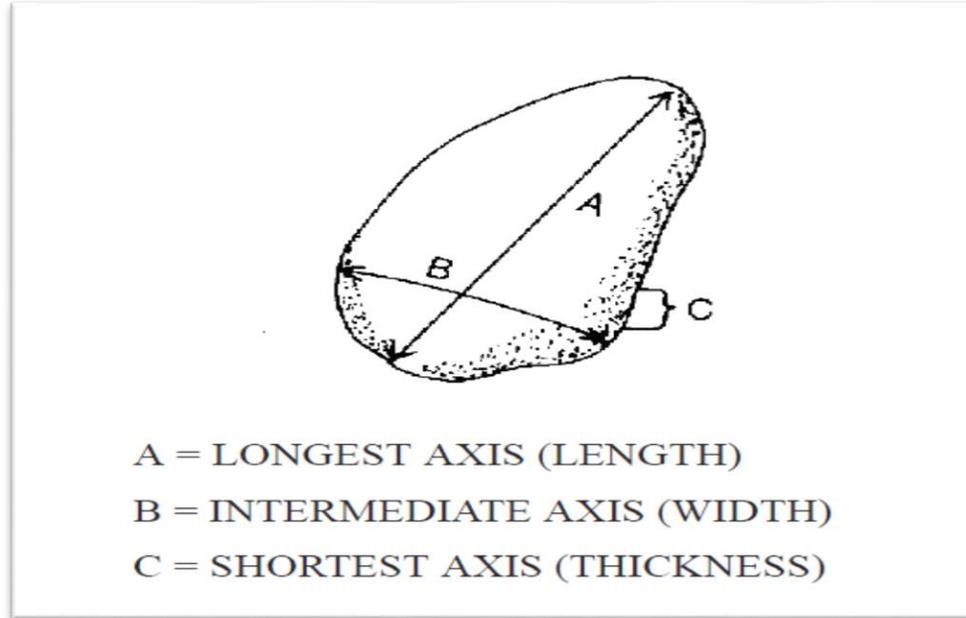


Figure 3.5: The axes of a pebble (Harrelson *et al.*, 1994)



Figure 3.6: Pebble Count

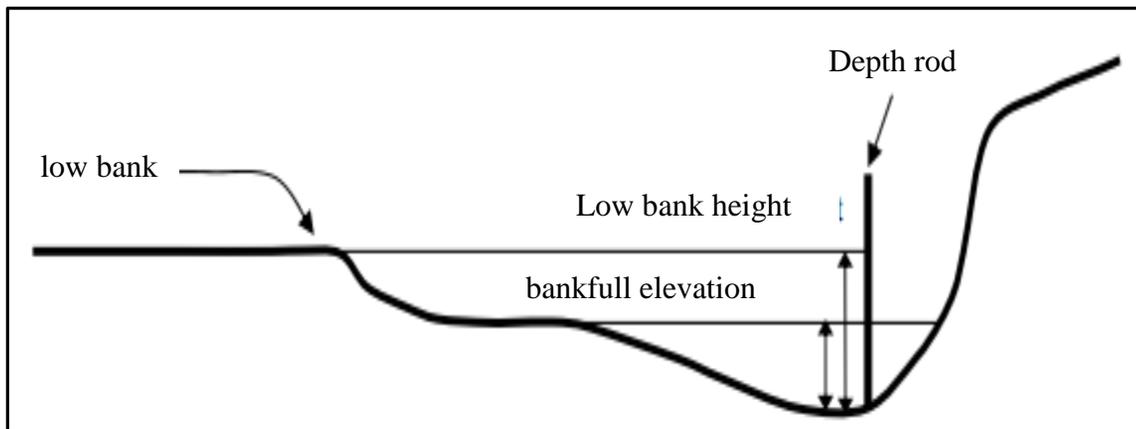


Figure 3.7: Measuring low bank height (Kline *et al.*, 2004)

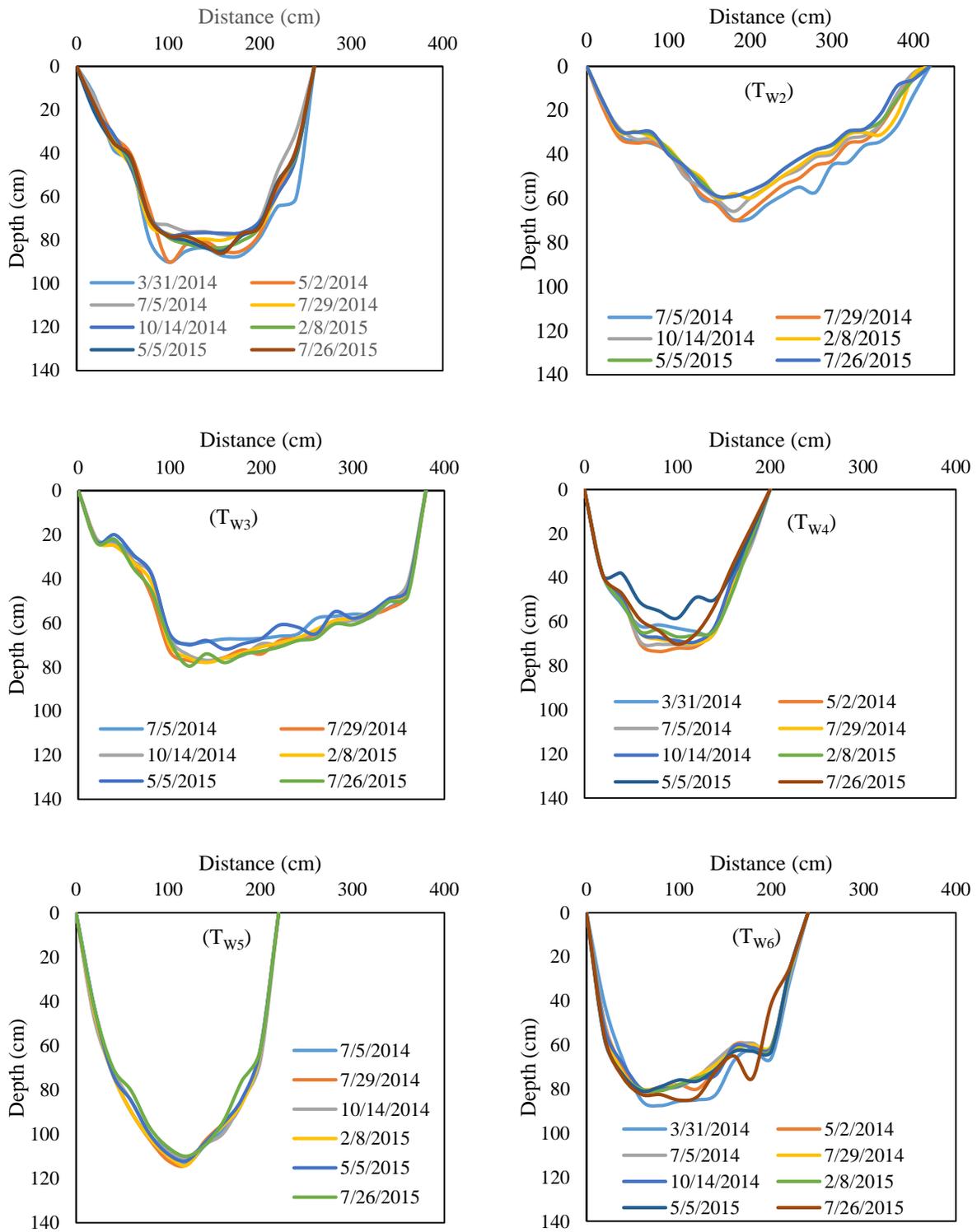


Figure 3.8: Changes in stream channel transects over time at T_{W1} , T_{W2} , T_{W3} , T_{W4} , T_{W5} , and T_{W6}

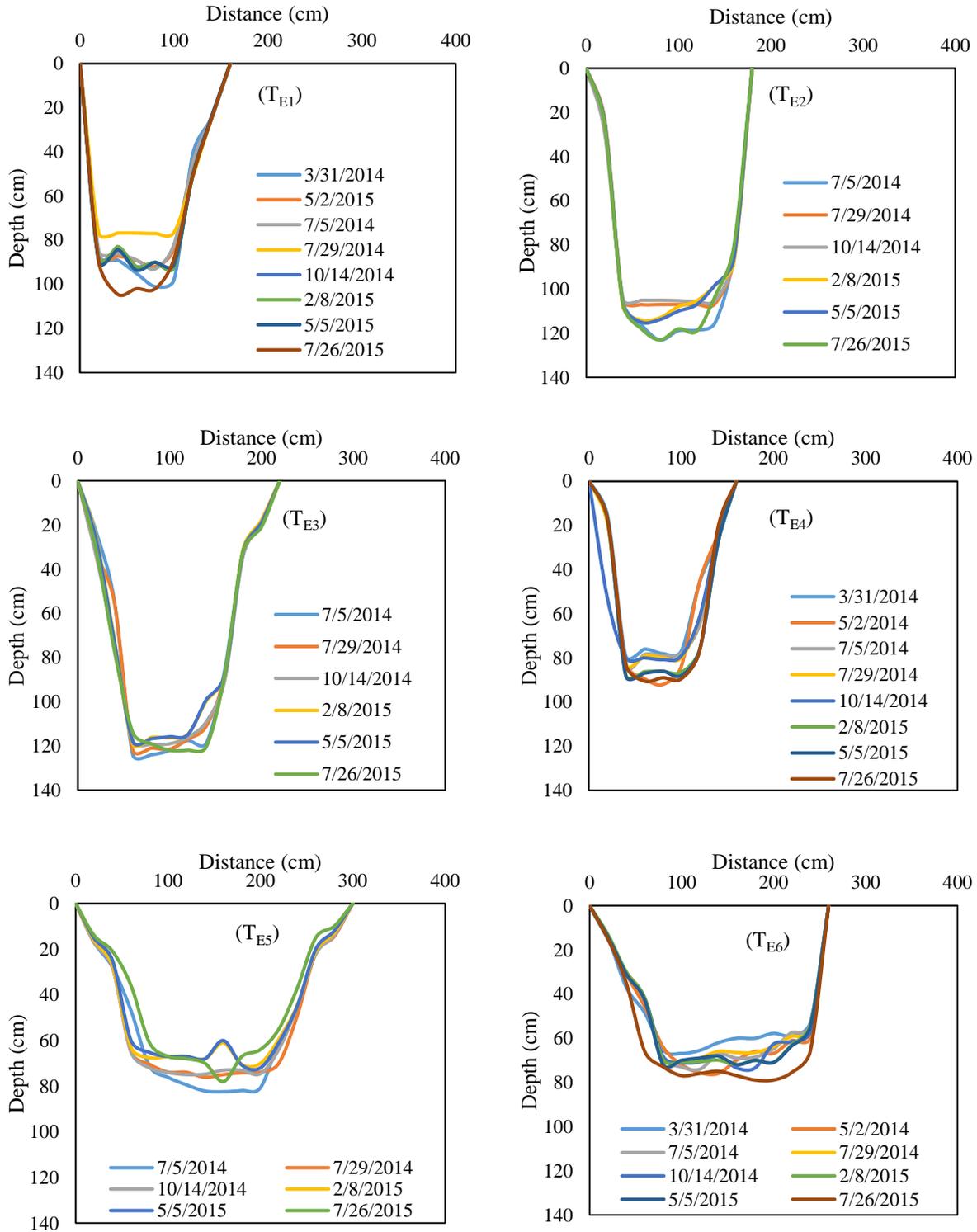


Figure 3.9: Changes in stream channel transects over time at T_{E1} , T_{E2} , T_{E3} , T_{E4} , T_{E5} , and T_{E6}

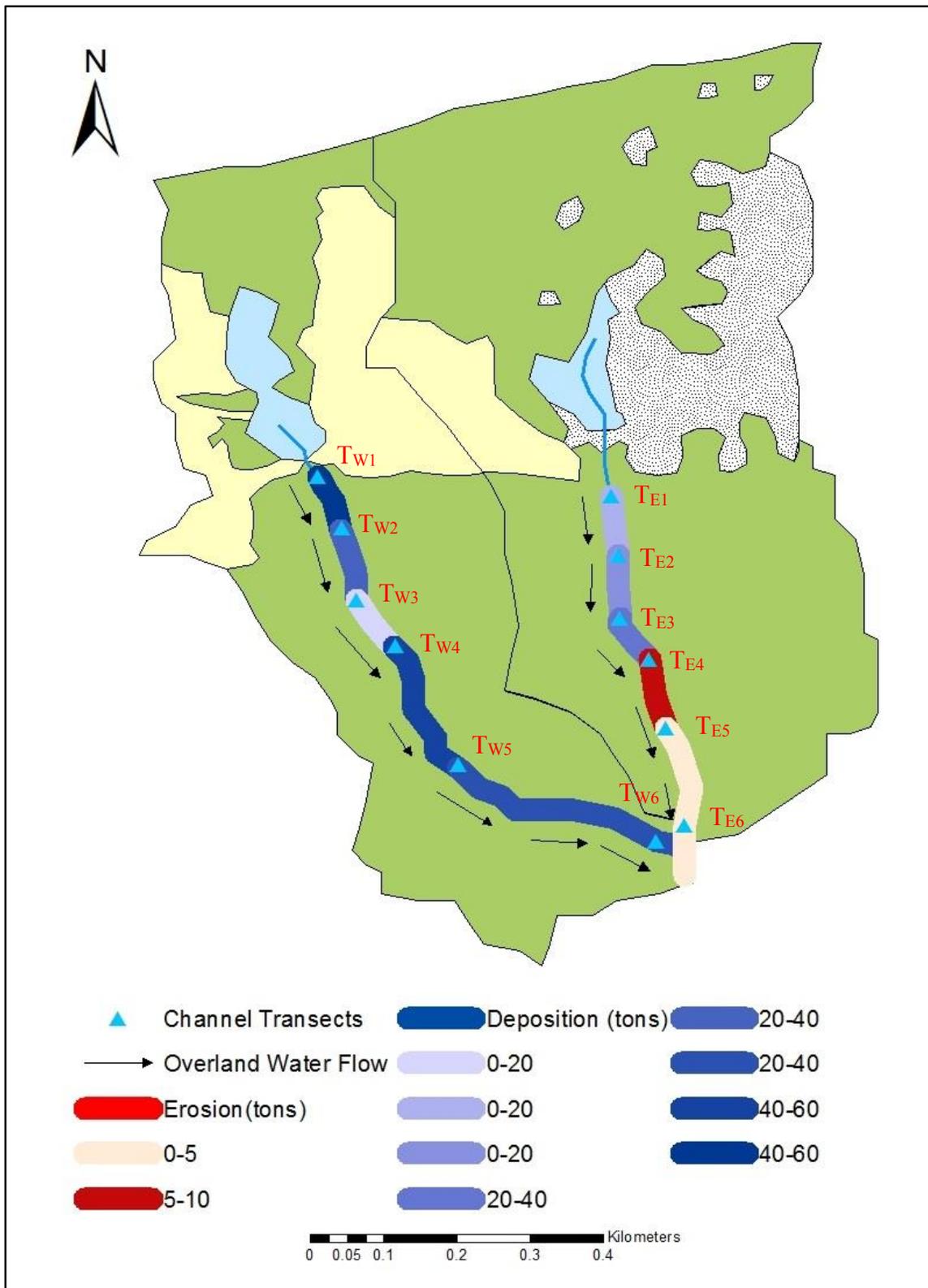


Figure 3.10: The net instream deposition and erosion between channel transect

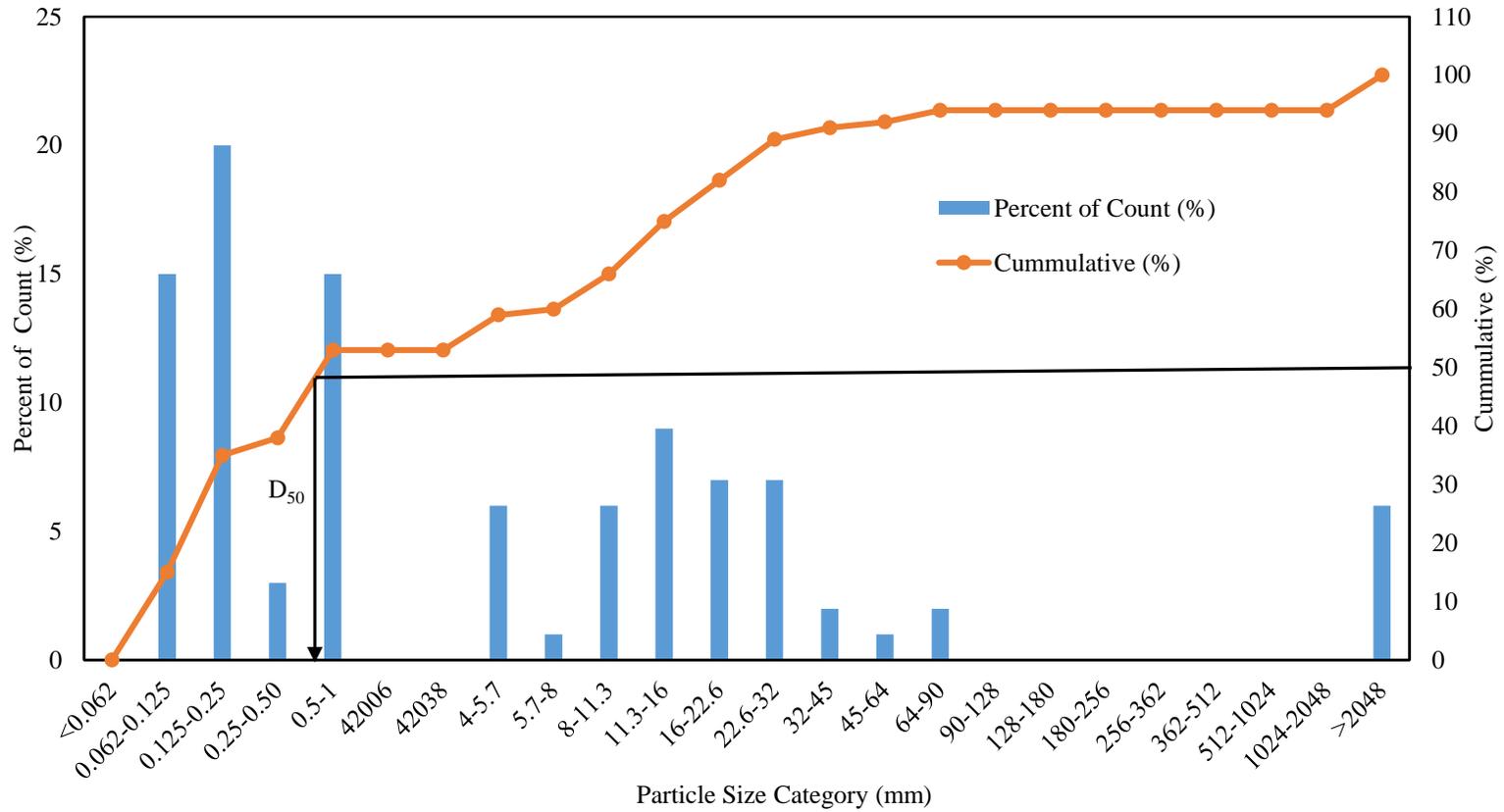


Figure 3.11: Pebble count size frequency and particle size distribution in West stream

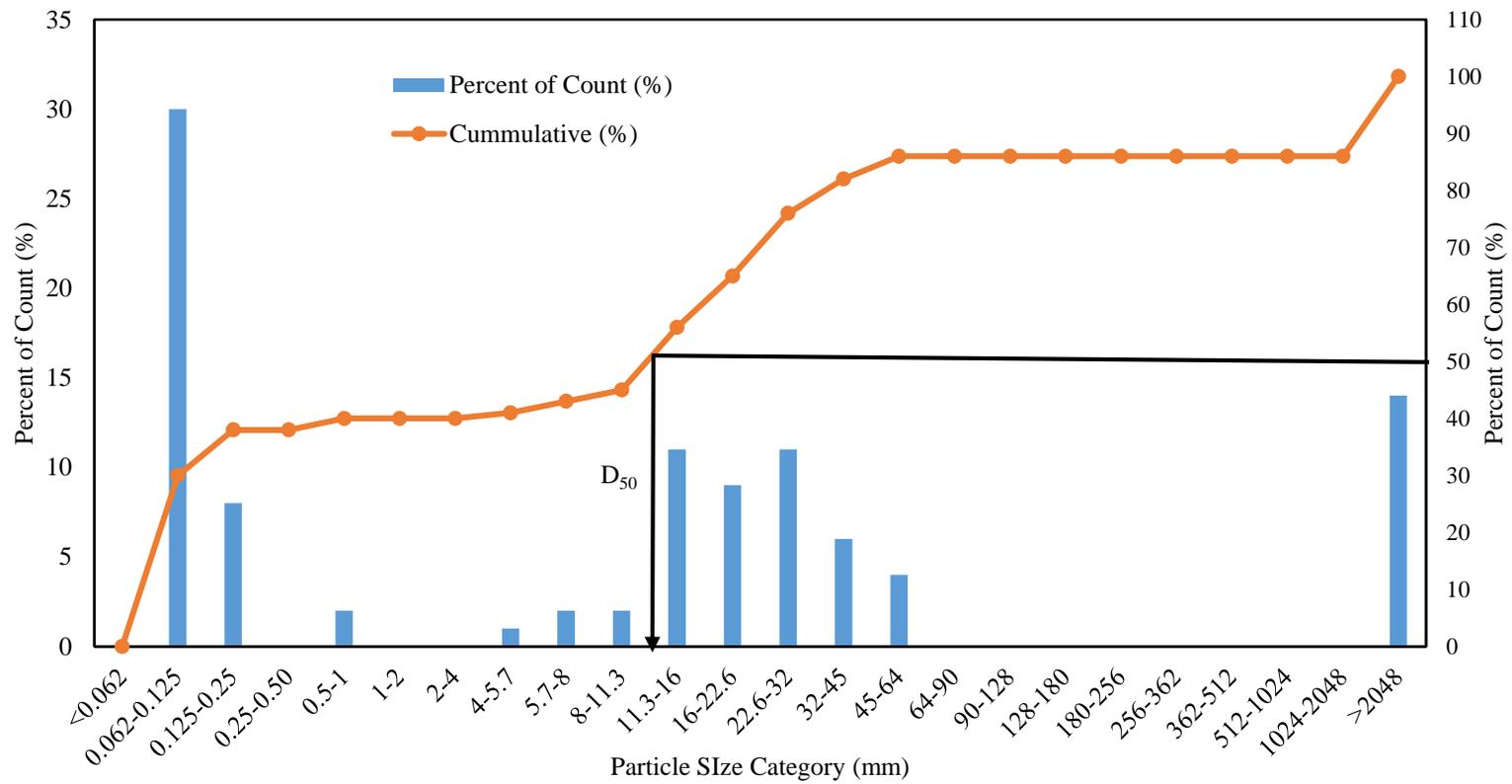


Figure 3.12: Pebble count size frequency and particle size distribution in East stream



Figure 3.13: West (a) and East (b) streams are classified as stream type E and F, respectively

CHAPTER IV – CONCLUSIONS

1. Project Summary

Forests are unique for water supply, and its quality and quantity (Bates *et al.*, 2008). Volume and peak rate of storm water are relatively low in forest streams as a result of the interception by trees, evaporation and infiltration. Consequently, soil erosion rates are usually low too in forested watersheds (Dunne and Leopold, 1978). The ability of trees to decrease runoff is significant especially in areas where forestry, agricultural or urban practices are present since overland flow that contributes to soil erosion can be minimized or slowed down by trees.

Although high quality of water is usually provided by forested watersheds; disturbances on land use are known to negatively affect water quality if BMPs such as SMZs are not well prescribed. However, any upstream urban or agricultural activities where BMPs are not present or are not efficiently functioning can influence downstream water quality regardless of the presence of well-designed downstream BMPs. This study was initiated on Marry Olive Thomas Demonstration Forest, Auburn, AL, USA. Sediment and water yield were monitored on discrete sections of two small adjacent watersheds during 16 months.

Impacts of upstream disturbances on downstream sediment yield and morphology were observed in the presence of BMPs. Monitoring the sediment yield at several locations in the study watersheds showed us how upstream disturbances affected downstream sediment yield. The influence of forested land and SMZs in mitigating sediment movement from disturbed area

were also monitored. This study also allowed evaluating the recovery of previously treated SMZs and clearcutting.

Flow per unit area did not significantly change from upstream to downstream in both watersheds during the study. Slight increases were observed at the very downstream monitoring stations of each watershed. The highest direct runoff was also observed at the most downstream monitoring stations of both watersheds. This seems to be due to presence of a forest road that goes through W_{W3} and W_{E3} . Forest roads can generally increase water yield (Grace, 2005). Increased water yield and runoff observed in the previous study at the downstream monitoring stations of W_W following partial harvesting within the SMZ almost reverted to pre-harvest conditions. This indicated that previously treated SMZ has been recovered.

Upstream disturbances resulted in excessive amount of sediment load at all of the monitoring stations. Decreasing sediment load from upstream to downstream in W_W showed that previously treated SMZs and clearcutting have been recovered, and do not produce additional sediment. Sediment load did not significantly changed through downstream in W_E , and it also suggested the recovery of the previous clearcutting. However, current sediment load was significantly higher than the sediment load observed in the previous study even 5 years after the partial harvesting of SMZ in W_W . Excessive amount of sediment load measured through monitoring stations in W_W and W_E was clearly indicated that sediment load was carried from upstream disturbed areas to downstream. In addition, reductions in sediment concentrations following completion of disturbances showed that the system may have been started to recover.

Changes at each channel transects over time were observed following the upstream disturbances. Intensive upstream disturbances altered the balance between flow and sediment

entering and leaving the streams by causing instream erosion and deposition along the both stream channels. Both streams were under the risk of streambank degradation even though they are covered by forest and SMZs. West and East stream channels had different channel dimensions. It seemed that this was not due to the current disturbance effect. West stream channel indicated very similar morphologic characteristics with the reference streams located in the same physiographic province. The different morphology of incised stream (East) may have been caused by the legacy effect of sediment from previous land use disturbances in the Southeast.

This study aimed to extend the current understanding of the disturbance impact on sediment yield and stream morphology in the presence of BMPs. Our findings expose the significance of BMP implementation on the entire watershed. If there is an uncontrolled sediment movement due to no or poor BMPs in a specific location of the watershed, well-designed BMPs in another location may not be effective. It seemed that upstream urban practices reduced the benefits gained by SMZs downstream. This study suggested that capacity of SMZs in protecting water quality may be decreased under the impacts of upstream disturbances. However, since SMZs have some other benefits such as providing habitat and passages for wildlife and controlling flooding (Daniels *et al.*, 2012), this study does not conclude that the use of SMZs are worthless. Above all, results highlighted the importance of managing watersheds in a holistic fashion in order to mitigate the impacts of disturbances more efficiently and economically.

2. Limitations

This study documented the linkage between upstream disturbances and downstream sediment yield in the presence of BMPs. Sediment concentrations were determined by direct measurements using manual sampling under significant rain events. Although most of the rain

events were caught (thanks to proximity of the study watershed to Auburn), few rain events that occurred particularly overnight were missed. Continuous water samples could have been taken during rain events, even during events occurred at nights, using the automatic water samplers.

Stream cross section surveys at several locations indicated the impacts of upstream disturbances on stream morphology. Even though the number of survey sites was sufficient to observe the changes in stream morphology for both streams, additional cross section surveys could have been used. This would provide more precise observation for stream morphology. However, because of the time limitation we were not able to have more survey sites for cross section observations.

3. Future Recommendations

Although this study suggests that upstream disturbances influenced downstream sediment yield and morphology, future monitoring is needed to observe the long term impacts of the upstream practices. Additional measurements for sediment yield and morphology are required to gain better understanding of the recovery of study streams following the completion of upstream urban disturbances. Inferences gained from this research will inform future discussions about the implications of BMPs within entire watershed when uncontrolled sediment is an issue.

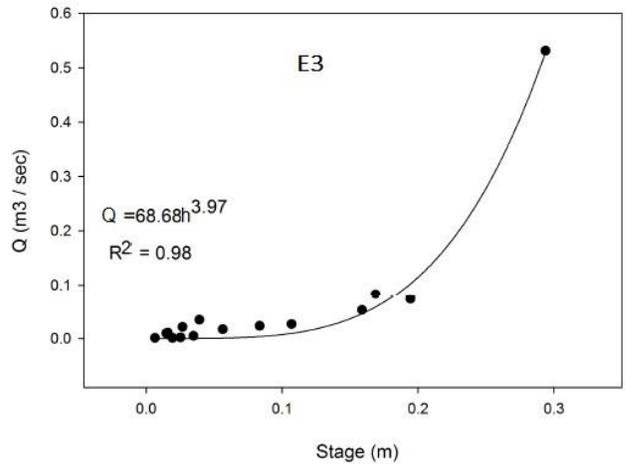
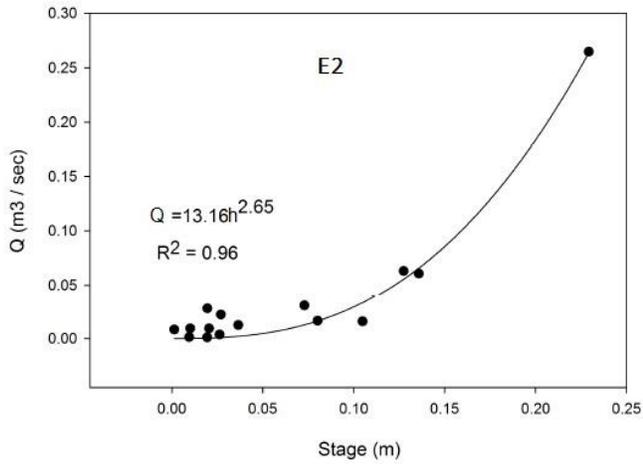
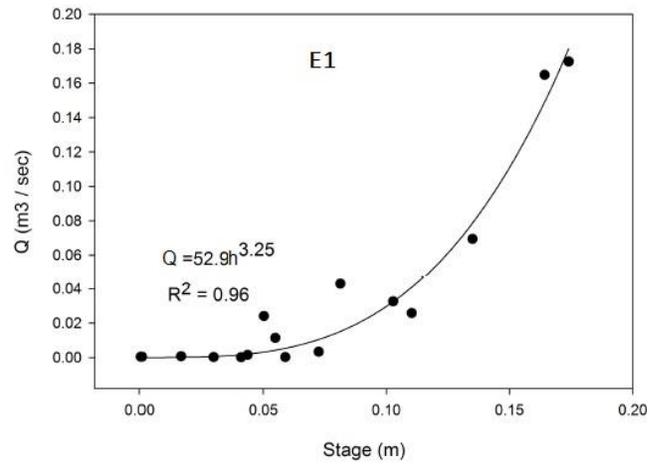
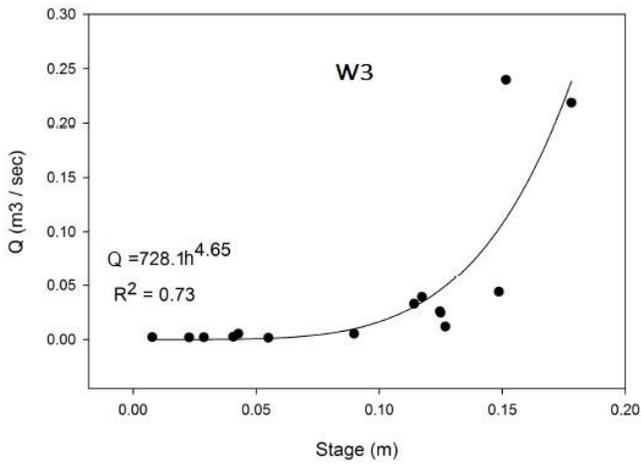
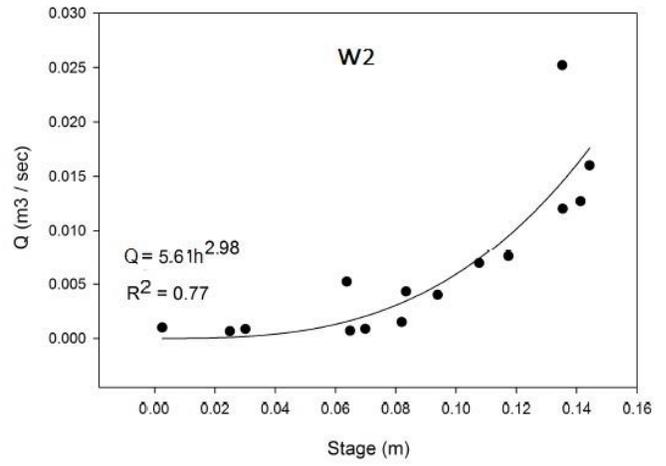
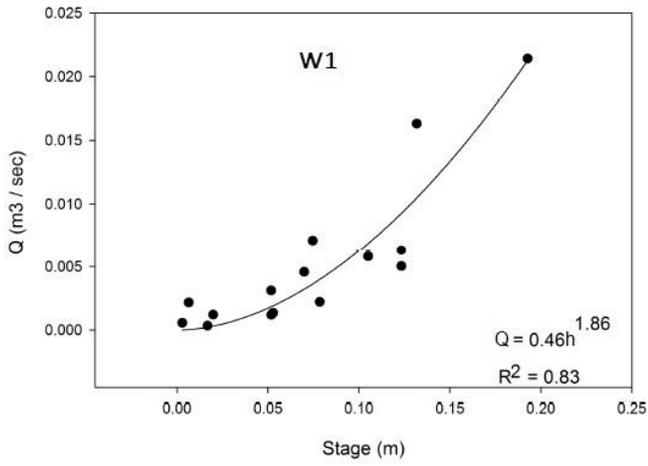
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APPENDIX A

RATING CURVES BETWEEN WATER STAGE AND DISCHARGES



APPENDIX B
CROSS – SECTION SURVEY

Appendix B. 1: Cross – section survey for the site 1 in West stream channel

Survey Crew	IC, FK	Date	8/20/2015
Site	1	$d_{bkf}=A_{bkf}/W_{bkf}$	0.8
A_{bkf} (sq ft)	7.23	$W/d=W_{bkf}/d_{bkf}$	11.5
W_{bkf} (ft)	9.2	$BHR=LBH/d_{mbkf}$	1.2
d_{mbkf} (ft)	1.15	$ER=W_{fpa}/W_{bkf}$	6.5
LBH (ft)	1.4		
W_{fpa} (ft)	59.4		

Station (ft)	Bkf Depth (ft)	Width (ft)	Bkf Area (sq ft)	Note
0	0	1	0.38	
1	0.75	1	0.93	
2	1.1	1	1.13	
3	1.15	1	1.1	
4	1.05	1	1.04	
5	1.03	1	1.02	
6	1	1	0.88	
7	0.75	1	0.58	
8	0.4	1	0.2	
9	0			
		Total	7.23	

Appendix B. 2: Cross – section survey for the site 2 in West stream channel

Survey Crew	IC, FK		
Site	2	Date	8/20/2015
A_{bkf} (sq ft)	16.3	$d_{bkf}=A_{bkf}/W_{bkf}$	1.3
W_{bkf} (ft)	12.1	$W/d=W_{bkf}/d_{bkf}$	9
d_{mbkf} (ft)	2.5	$BHR=LBH/d_{mbkf}$	1
LBH (ft)	2.6	$ER=W_{fpa}/W_{bkf}$	3.6
W_{fpa} (ft)	44		

Station (ft)	Bkf Depth (ft)	Width (ft)	Bkf Area (sq ft)	Note
0	0	1	0.2	
1	0.4	1	0.65	
2	0.9	1	1.01	
3	1.12	1	1.34	
4	1.55	1	1.78	
5	2	1	2.03	
6	2.05	1	2.28	
7	2.5	1	2.4	
8	2.3	1	2.3	
9	2.3	1	1.5	
10	0.7	1	0.58	
11	0.45	1	0.23	
0	0			
		Total	16.3	

Appendix B. 3: Cross – section survey for the site 3 in West stream channel

Survey Crew	IC, FK	Date	8/20/2015
Site	3		
A_{bkf} (sq ft)	12.1	$d_{bkf}=A_{bkf}/W_{bkf}$	1.1
W_{bkf} (ft)	10.8	$W/d=W_{bkf}/d_{bkf}$	9.6
d_{mbkf} (ft)	2.3	$BHR=LBH/d_{mbkf}$	1.3
LBH (ft)	3.1	$ER=W_{fpa}/W_{bkf}$	7.4
W_{fpa} (ft)	80		

Station (ft)	Bkf Depth (ft)	Width (ft)	Bkf Area (sq ft)	Note
0	0	1	0.3	
1	0.6	1	0.68	
2	0.75	1	0.88	
3	1	1	1.5	
4	2	1	2.15	
5	2.3	1	2.2	
6	2.1	1	1.88	
7	1.65	1	1.23	
8	0.9	1	0.7	
9	0.5	1	0.4	
10	0.3	1	0.15	
0	0			
		Total	12.1	

Appendix B. 4: Cross – section survey for the site 1 in East stream channel

Survey Crew	IC, FK	Date	8/20/2015
Site	1	$d_{bkf}=A_{bkf}/W_{bkf}$	0.7
A_{bkf} (sq ft)	6	$W/d=W_{bkf}/d_{bkf}$	13.3
W_{bkf} (ft)	8.9	$BHR=LBH/d_{mbkf}$	2.8
d_{mbkf} (ft)	0.85	$ER=W_{fpa}/W_{bkf}$	1.6
LBH (ft)	2.35		
W_{fpa} (ft)	14.2		

Station (ft)	Bkf Depth (ft)	Width (ft)	Bkf Area (sq ft)	Note
0	0	1	0.18	
1	0.35	1	0.43	
2	0.5	1	0.58	
3	0.65	1	0.73	
4	0.8	1	0.83	
5	0.85	1	0.83	
6	0.8	1	0.8	
7	0.8	1	0.78	
8	0.75	1	0.6	
9	0.45	1	0.23	
0	0			
		Total	6	

Appendix B. 5: Cross – section survey for the site 2 in East stream channel

Survey Crew	IC, FK		
Site	2	Date	8/20/2015
A_{bkf} (sq ft)	9.7	$d_{bkf}=A_{bkf}/W_{bkf}$	0.9
W_{bkf} (ft)	10.8	$W/d=W_{bkf}/d_{bkf}$	12
d_{mbkf} (ft)	1.5	$BHR=LBH/d_{mbkf}$	1.6
LBH (ft)	2.45	$ER=W_{fpa}/W_{bkf}$	1.4
W_{fpa} (ft)	15.4		

Station (ft)	Bkf Depth (ft)	Width (ft)	Bkf Area (sq ft)	Note
0	0	1	0.15	
1	0.3	1	0.6	
2	0.9	1	1.08	
3	1.26	1	1.38	
4	1.5	1	1.5	
5	1.5	1	1.48	
6	1.45	1	1.27	
7	1.08	1	0.99	
8	0.9	1	0.6	
9	0.29	1	0.4	
10	0.5	1	0.25	
0	0			
		Total	9.7	

Appendix B. 6: Cross – section survey for the site 3 in East stream channel

Survey Crew	IC, FK		
Site	3	Date	8/20/2015
A_{bkf} (sq ft)	10.5	$d_{bkf}=A_{bkf}/W_{bkf}$	0.8
W_{bkf} (ft)	13.8	$W/d=W_{bkf}/d_{bkf}$	17.1
d_{mbkf} (ft)	1.45	$BHR=LBH/d_{mbkf}$	4.3
LBH (ft)	6.2	$ER=W_{fpa}/W_{bkf}$	1.2
W_{fpa} (ft)	17.1		

Station (ft)	Bkf Depth (ft)	Width (ft)	Bkf Area (sq ft)	Note
0	0	1	0.05	
1	0.1	1	0.15	
2	0.2	1	0.25	
3	0.3	1	0.35	
4	0.4	1	0.45	
5	0.5	1	0.7	
6	0.9	1	1.03	
7	1.15	1	1.2	
8	1.25	1	1.3	
9	1.35	1	1.4	
10	1.45	1	1.3	
11	1.14	1	1.12	
12	1.1	1	0.88	
13	0.65	1	0.33	
0	0			
		Total	10.5	

APPENDIX C
PEBBLE COUNT DATA

Appendix C. 1: Pebble count for West stream

Description	Material	Size (mm)	Riffle	Pool	%	%	Cum %	d16	d35	d50	d84
Silt/Clay	silt/clay	<0.062	0	0	0.0%	0.0%	0.0%	0.1	0.2	0.7	21.6
Sand	very fine sand	0.062	4	11	4.0%	11.0%	15.0%				
	fine sand	0.125	12	8	12.0%	8.0%	35.0%				
	medium sand	0.25	0	3	0.0%	3.0%	38.0%				
	coarse sand	0.50	3	12	3.0%	12.0%	53.0%				
	very coarse sand	1.0	0	0	0.0%	0.0%	53.0%				
Gravel	very fine gravel	2.0	0	0	0.0%	0.0%	53.0%				
	fine gravel	4.0	4	2	4.0%	2.0%	59.0%				
	fine gravel	5.7	1	0	1.0%	0.0%	60.0%				
	medium gravel	8.0	5	1	5.0%	1.0%	66.0%				
	medium gravel	11.3	7	2	7.0%	2.0%	75.0%				
	coarse gravel	16.0	4	3	4.0%	3.0%	82.0%				
	coarse gravel	22.6	4	3	4.0%	3.0%	89.0%				
	very coarse gravel	32	1	1	1.0%	1.0%	91.0%				
	very coarse gravel	45	0	1	0.0%	1.0%	92.0%				
Cobble	small cobble	64	1	1	1.0%	1.0%	94.0%				
	medium cobble	90	0	0	0.0%	0.0%	94.0%				
	large cobble	128	0	0	0.0%	0.0%	94.0%				
	very large cobble	180	0	0	0.0%	0.0%	94.0%				
Boulder	small boulder	256	0	0	0.0%	0.0%	94.0%				
	small boulder	362	0	0	0.0%	0.0%	94.0%				
	medium boulder	512	0	0	0.0%	0.0%	94.0%				
	large boulder	1024	0	0	0.0%	0.0%	94.0%				
	very large boulder	2048	0	0	0.0%	0.0%	94.0%				
Bedrock	bedrock	>2048	4	2	4.0%	2.0%	100.0%	0.09	0.18	0.67	21.57
	TOTAL / %of whole		50	50	50.0%	50.0%	100.0%				

Appendix C. 2: Pebble count for East stream

Description	Material	Size (mm)	Riffle	Pool	%	%	Cum %	d16	d35	d50	d84
Silt/Clay	silt/clay	<0.062	0	0	0.0%	0.0%	0.0%	0.1	0.2	11.5	46.5
Sand	very fine sand	0.062	3	27	3.0%	27.0%	30.0%				
	fine sand	0.125	5	3	5.0%	3.0%	38.0%				
	medium sand	0.25	0	0	0.0%	0.0%	38.0%				
	coarse sand	0.50	2	0	2.0%	0.0%	40.0%				
	very coarse sand	1.0	0	0	0.0%	0.0%	40.0%				
Gravel	very fine gravel	2.0	0	0	0.0%	0.0%	40.0%				
	fine gravel	4.0	0	1	0.0%	1.0%	41.0%				
	fine gravel	5.7	1	1	1.0%	1.0%	43.0%				
	medium gravel	8.0	2	0	2.0%	0.0%	45.0%				
	medium gravel	11.3	6	5	6.0%	5.0%	56.0%				
	coarse gravel	16.0	7	2	7.0%	2.0%	65.0%				
	coarse gravel	22.6	8	3	8.0%	3.0%	76.0%				
	very coarse gravel	32	4	2	4.0%	2.0%	82.0%				
very coarse gravel	45	4	0	4.0%	0.0%	86.0%					
Cobble	small cobble	64	0	0	0.0%	0.0%	86.0%				
	medium cobble	90	0	0	0.0%	0.0%	86.0%				
	large cobble	128	0	0	0.0%	0.0%	86.0%				
	very large cobble	180	0	0	0.0%	0.0%	86.0%				
Boulder	small boulder	256	0	0	0.0%	0.0%	86.0%				
	small boulder	362	0	0	0.0%	0.0%	86.0%				
	medium boulder	512	0	0	0.0%	0.0%	86.0%				
	large boulder	1024	0	0	0.0%	0.0%	86.0%				
	very large boulder	2048	0	0	0.0%	0.0%	86.0%				
Bedrock	bedrock	>2048	8	6	8.0%	6.0%	100.0%	0.07	0.15	11.47	46.5
	TOTAL / %of whole		50	50	50.0%	50.0%	100.0%				