Sedimentation in a Tupelo-Baldcypress Forested Wetland 12 Years Following Harvest Disturbance

by

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Sediment accumulation and loss were measured in a water tupelo (*Nyssa aquatica*) – baldcypress (*Taxodium distichum*) forested wetland in years 2, 7, 10 and 12 following harvesting disturbance. A 3 X 3 Latin Square design was replicated three times and compared to a pseudo-replicated reference stand (REF). Disturbance treatments were chainsaw felling of trees with (1) helicopter removal of logs (HELI), (2) helicopter removal of logs followed by simulated skidder removal (SKID), and (3) helicopter removal of logs followed by glyphosphate application (GLYPH). Measurements of sediment accretion show little difference in the treatments and reference in the first two years following harvest. After two years the harvest treatments accumulate more sediment than the reference. Of the harvest treatments, the GLYPH plots accumulate the greatest quantities of sediment. The difference in sediment accumulation between the treatments and reference begin to fade in the 12^{th} year of recovery. Results show that skidder and helicopter harvests differed very little in the amount of sediment trapped, while Glyphosphate application increased the wetland's ability to trap sediment beginning after the second year. In addition, sediment trapping in associated with herbaceous cover that slows flood waters and allows soil particles to precipitate from flood water.

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Introduction

Wetland management continues to be a complex issue in natural resource management. Over the past four-hundred years, wetlands in the United States of America have been drained, filled, and converted to other uses. Wetlands were seen as impediments to development. Wetland destruction was even encouraged by the US government, as federal and state laws funded and promoted wetland conversion to agricultural and other economic uses (Shepard et al., 1993).

Public awareness about the importance of the role of wetlands in the environment is growing (Shepard et al., 1993, and Brown and Sullivan, 1988). There is concern about the future of wetlands and the current health of the remaining wetlands. Protecting wetlands entails understanding the functions wetlands and, moreover, being able to decide which areas are indeed wetlands so that they may be managed appropriately. Unfortunately, defining the term *wetland* has been a terribly difficult issue in natural resource management (Tolliver, 1993, and Dubensky et al. 1993). It is easy to understand why areas such as the Florida Everglades are wetlands, as they are inundated by water all year round, but other areas may not experience flooding frequently – if ever, or they may only be wet seasonally, yet these areas, too, may be jurisdictional wetlands.

Learning to understand the functions of wetlands is a great challenge. Wetlands are known to perform an array of benefits to the environment including improved water quality and wildlife habitat, but some may also contribute negatively, such as methane production (Walbrige, 1993 and Stepniewski and Glinski, 1988), although there are far greater sources of air pollution (largely anthropogenic).

It is understandable why some wetlands have been set aside in parks and reserves. These areas are essential to environmental health. But more importantly, these areas were incredibly difficult to inhabit. Often the land was unsuitable for building structures or growing cash crops. Today, if land is unsuitable for construction, agriculture, or mining it's best use is often considered a park, reserve, or wildlife sanctuary.

Nonetheless, areas such as wetlands may provide recreation and silviculture, thereby helping the local economy (Walbridge, 1993). Finding equilibrium between protection and exploitation is essential. Though there are no regulations specific to wetlands, there are regulations applicable to *waters*. In 1899 the US Army Corps of Engineers (USACE) issued the Rivers and Harbors Act that protected wetlands with a direct impact on navigable waters (Dubensky et al., 1993). Over 50 years later, Congress passed the Federal Water Pollution Control Act (Dubensky et al., 1993). The act was amended in 1972 to include protection of physical, biological, and chemical integrity of *waters of the United States*, becoming known as the Clean Water Act (PL-92-500, 86 stat. 816, 33 USC 1251) (Dubensky et al., 1993). To the act was added the section 404 provision (53 F.Reg. 20764) that established a permit program to control the discharge of dredge or fill material into navigable waters (Dubensky et al., 1993). In addition, the act also gave the US Environmental Protection Agency (EPA) authority in conjunction with the USACE to develop the section 404 (b)(1) guidelines to implement the section 404 program, though authority to issue permits remained with the USACE (Dubensky et al., 1993).

In the Clean Water Act, the term *waters of the United States* led a simple discharge permit for navigable waters into a serious responsibility for the USACE (Dubensky et al., 1993). *Navigable waters* were considered a subset of the *waters of the*

United States, thereby expanding the permitting authority of the USACE (Dubensky et al, 1993). The USACE was originally reluctant to expand its permitting authority to include wetlands into the definition of waters of the United States. However in 1975, Federal District Court (NRDC vs. Callaway, 392 F. Supp. 685 [DDC 1975]) required that the USACE amend its regulations and expand coverage to wetlands (Dubensky et al., 1993). *Waters* now included all surface waters and tributaries, adjacent wetlands, and isolated waters or wetlands – the use or degradation of which could affect interstate or foreign commerce (Dubensky et al., 1993). By 1977 the USACE had acquired control over all waters of the United States – including wetlands (Dubensky et al., 1993).

There are many wetland definitions and terms some of which may be confusing and even contradictory (Mitcsh and Gosselink, 1993). Furthermore, since substantial economic stakes may be at hand, politics may play an influence on drawing a definition. Wetlands are delineated from surrounding landscape by three distinguishing features; (1) the presence of water, either at the surface or at the root zone; (2) unique soil conditions that differ from surrounding uplands; and (3) presence of hydrophytes and the lack of flood intolerant species (Mitsch and Gosselink, 1993). All three of the criteria are usually required for the area to be classed as a wetland, though in disturbed sites two may be adequate (Cubbage and Flather, 1993).

Water must be present for at least part of the year, though depth and duration of flooding varies grossly between individual wetlands. Since water levels may fluctuate seasonally and yearly, wetland boundaries may not be accurately drawn by the presence of water at one time (Mitsch and Gosselink, 1993). Furthermore, since wetlands may often be margins between uplands and deep water, they may be influenced by both systems. This position in the ecotone may suggest that wetlands are not a separate entity, but rather an extension of the terrestrial and/or aquatic system (Mitsch and Gosselink, 1993). Wetland species may be difficult to use in delineation since organisms range from those that have adaptations to live in either wet or dry conditions to those adapted solely for aquatic life (Mitsch and Gosselink, 1993).

The legal definition of wetlands is found in the USACE regulations for dredge and fill permit systems required by Section 404 of the 1977 Clean Water Act Amendments (Mitsch and Gosselink, 1993). The latest version of the definition is stated as:

> The term "wetland" means those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted to life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas. (33 CFR323.3(b); 1984)

There are approximately 104 million acres of wetlands in the lower 48 states, of which an estimated 50-60 million acres are under active forest management (Dubensky et al., 1993). In Section 404(f)(1), "normal farming, silviculture, and ranching activities such as plowing, seeding, cultivating, minor drainage, harvesting products of food, fiber, and forest products" are specifically exempt from the permitting system (Dubensky et al., 1993). The EPA closely scrutinizes "normal silvicultural activities," therefore exemption does not free silviculture of review and regulation (Dubensky et al., 1993). Silvicultural activities may be exempt from the Section 404 permit requirement provided the following five criterion are met; 1) the silvicultural activities are “normal”, 2) are conducted according to Best Management Practices (BMPs), 3) are part of an ongoing activity, 4) do not modify hydrology, and 5) do not add toxins, such as engine oil, to the system, (Mitsch and Gosselink, 1993; Brown and Binkley, 1993).

“Normal” silvicultural activities will directly and indirectly impact a given area. Varying harvest techniques such as pull boats, skidding and helicopter logging may produce different effects on the site. Channels, created by pull boat harvests in the 19th and early 20th century, remain evident in the latter part of the 20th century (Aust, 1989; Mader, 1990). Skidder traffic may alter physical characteristics of the soil such as increased soil density and strength, decreased pore space, and decreased gas exchange (Aust, 1989). The loss of vegetation, especially trees, reduces the amount of evapotranspiration and may result in a higher water table (Brooks et al., 1997). A high water table may discourage the growth of flood intolerant vegetation. Furthermore, the change in vegetation may alter the roughness of the site and thereby change the wetland’s ability to trap and contain sediment carried in floodwaters.

The long-term effects of harvest in forested wetlands are not well understood and initial differences related to harvesting methods may change over time. This study was designed to (1) evaluate the differences in sediment trapping in different harvesting methods and (2) to examine how sedimentation rates change over a 12-year period following harvest disturbance.

Literature Review

Role of Wetlands in Sedimentation

One of the most widely recognized functions of wetlands is the improvement of water quality through sediment removal (Johnston, 1991, Hupp and Bazemore, 1993). Though some amount of suspended sediment in surface waters is considered normal (Johnston, 1991), excess levels can cause changes in water temperature and dissolved oxygen which are detrimental to some aquatic organisms (Binkley and Brown, 1993). Furthermore, sediments carried to waterways may take with them contaminants such as heavy metals and agricultural chemicals such as nitrates, phosphates, and pesticides (Johnston, 1991). These pollutants can lead to chemical contamination, changes in pH, and algal blooms in waterways. Not only are there negative environmental effects associated with chemical contamination, but there are economic effects as well since pollutants increase the costs of water treatment.

Concern has risen over the loss and degradation of wetlands. In an effort to preserve the amount of original wetland resources, their ability to improve water quality by sediment and nutrient retention has been touted by wetland managers. However, it is plausible that this situation could be to the detriment of wetlands as they are exploited as disposal sites for anthropogenic wastes (Johnston, 1991). Processes such as denitrification are probably indefinitely sustainable without detriment to the wetland, however, processes like excessive phosphorus loading are clearly not (Johnston, 1991). For this reason care should be taken not to exploit wetlands as panaceas for pollution in lieu of sustainable management activities upstream.

Though the sediment trapping ability of wetlands is commonly acknowledged, very little is actually understood about the quantitative amounts of sediment tapped by forested wetlands (Hupp and Bazemore, 1993). Though sedimentation studies have been preformed, the majority of them were conducted in salient systems such as tidal marshes rather than forested fresh-water systems (Hupp and Bazemore, 1993, Hatton et al., 1983). In addition, previous studies may be subject to error since sediment quantities were based on water budget data which is very difficult to obtain and is commonly subject to substantial error (Hupp and Bazemore, 1993). Furthermore, comprehensive studies vary greatly in their hydrologic methods, assumptions, and interpretations posing considerable limitations to understanding the role of forested wetlands in sediment removal, storage, and transport (Hupp and Bazemore, 1993).

Sediment deposition in flood plains and forested wetlands increases with floods, excessive channelization of upstream waterways, and long hydroperiods (Hupp and Bazemore, 1993). Rivers in the Gulf Coastal Plain, where this study was conducted, generally contain high concentrations of suspended sediment because of highly erodible uplands and channel instability incurred by anthropogenic activities (Hupp and Bazemore, 1993). According to Kleiss (1996), studies of vertical accumulation in freshwater wetlands had been limited since more research efforts have been focused on coastal marshes due to submergence and subsequent land loss during the past century. A literature review by Johnston (1991) found only 14 published reports of sediment accumulation in freshwater wetlands were found in the entire country. Reported sedimentation accumulation rates range between 0.07 and 2.62 cm/year, varying on wetland characteristics and measurement methods (Mitch et al., 1979; Hupp and Bazemore, 1993; Johnston, et al., 1984; Kleiss, 1996).

According to Hupp and Bazemore (1993), Mitsch et al. (1979) estimated an average sediment deposition rate of less than 0.2 cm/year by using sediment traps in a cypress-tupelo forested wetland in southern Illinois. In a Wisconsin stream, Johnston et al. (1984) reported a deposition rate of 2.62 cm/year for natural levee areas based on ^{137}Cs dating, however, no overall wetland deposition rate was estimated. Cooper et al. (1987) also employed ^{137}Cs techniques and estimated an average deposition rate of less than 0.26 cm/year in a coastal plain swamp in North Carolina (Hupp and Bazemore, 1993). In a forested wetland on the Cache River in Arkansas, Hupp and Morris (1990) reported mean deposition rates of 0.34 cm/year in slough areas and 0.07 cm/year for elevated portions or 'islands'. Depending on flood stage, The potential for sediment deposition in the low areas is greater than in higher areas because the low areas are flooded more frequently and for a greater amount of time (Hupp and Bazemore, 1993, Lugo et al, 1980).

Kleiss (1996) employed three techniques (clay pads, ^{137}Cs, and a dendrogeomorphic method) to estimate vertical accretion in a bottomland hardwood wetland also along the Cache River in eastern Arkansas. In the natural levee site, the average accumulation was the greatest with 0.86, 0.41, and 0.27 cm/year for the clay pad, ^{137}Cs, and dendrogeomorphic methods, respectively (Kleiss, 1996). Within the cypress/tupelo swamp the average accretion was 1.55, 0.21, 0.26 cm/year for the respective methods (Kleiss, 1996). The slightly elevated areas ("ridges" or "islands") yielded average accretion rates of 0.27, 0.13, 0.08 cm/year for the respective methods (Kleiss, 1996). Finally, in the abandoned channels average accretion based on the respective methods was 0.80, 0.18, 0.23 cm/year (Kleiss, 1996).

Standard fluvial geomorphology states that the higher elevation of natural levees are a result of the retardation of stream velocity when the water leaves the channel, allowing the largest particles to be deposited adjacent to the bank (Kleiss, 1996). This characteristic can be seen in Johnston et al.'s (1984) study that found the sediment deposition on the natural levee of the Wisconsin wetland was higher in areas associated with a stream bank, rather than a back water area. In Kleiss's (1996) study, the greater accumulation rates where found in the cypress/tupelo swamp rather than the natural levee, conflicting with theory and other studies (Johnston et al., 1984). Kleiss (1996) attributed the discrepancy with the small size of particles suspended in water entering the Cache River wetland and associated these particles with agricultural run-off. In the geologic sense, the levee was formed in the natural way, by means of coarser materials. Currently coarse materials are still deposited onto the levee, however, the flood waters that reach beyond the levee are still heavily laden with fine materials (Kleiss, 1996). Swamps retain water and allow time for particle deposition.

Temporal Variations in Sedimentation

Sediment deposition in wetlands is beneficial to downstream water quality by reducing turbidity and absorbing contaminants held on solid particles. Some reworking of deposited sediments may occur due to fluvial activity in the flood plain, however, sedimentation is a relatively irreversible process (Johnston, 1991). This situation may be explained in part by the protection of vegetation and leaf litter that armors the ground surface against raindrop impact (Kleiss, 1996).

The season in which a flood event occurs can also be a factor in sediment deposition. In the first flood event of the year, the flood plain is covered in herbaceous growth and leaf litter. This cover provides protection from the erosive forces of raindrops and the wetland will be able to accumulate new sediment while losing little through erosion (Kleiss, 1996). Later on in the season, the litter layer has washed away and much of the herbaceous growth has senesced and submerged. Without the protection provided by this cover the raindrop impact loosens and dislodges soil particles, resulting in greater erosion than earlier in the season (Kleiss, 1996).

Phillips (1989) suggests that wetlands are capable of acting as sediment sinks while at the same time witnessing losses to lateral erosion in flood plains. In geologic time (millennia) sediment stored in wetlands, as their role is considered in fluvial systems, are considered to be in transport, rather than wetlands being considered as permanent sinks for sediment storage (Phillips, 1989, Mitsch and Gosselink, 1993). Kleiss (1996) suggests that the greater sediment accumulation rates found in the cypress/tupelo swamp when compared to the natural levee of the Arkansas wetland are a temporary phenomenon in the geologic sense. However, it is still widely accepted that wetlands serve as sinks for sediment, nutrients, and pollutants.

Organic Soil Accumulation

Wetlands will also accumulate organic material when biomass production exceeds the decomposition rate. Most organic soils occur in wetlands because the lack of oxygen slows the rates of decay (Johnston, 1991). Unlike mineral sediment production, which may rely on inputs of soil material from outside the wetland, organic soil production relies primarily on production and decomposition of material *in situ* (Johnston, 1991).

There are three primary processes that affect the formation of peat. First, is the loss of organic matter by animals, microorganisms, and leaching (Johnston, 1991). The second is the loss of the physical structure of the organic material (Johnston, 1991). Finally, is a change in chemical state by the production of new types of molecules by microorganisms and chemical reactions (Johnston, 1991).

The rate of organic matter accumulation is affected by the nature of the plant material, climate, fire, geologic factors, flooding, and anthropogenic disturbance (Johnston, 1991). In Georgia's Okefenokee Swamp, organic soil accretion estimated by C^{14} analysis was 0.06, 0.05, and 0.06 cm/year for a Marsh, Shrub, and Cypress site respectively (Flebb, 1982, Johnston, 1991). The thickness accretion generally decreases with depth of the peat profile because the organic material continues to decompose and becomes compacted over time (Johnston, 1991).

Accumulation of Fluvial Materials

In order to understand the effects of time on patterns of sediment dispersal, the general features that influence sedimentation must be analyzed. Several sources in literature recognize the landscape features that lead to sediment precipitation in fluvial systems (Johnston, 1991, Phillips, 1989). These features include characteristics that lead to low stream power, such as high amounts of shrub/scrub vegetation, mild stream inclination, and sluggish meandering shallow channels.

The ability of water to transport and carry sediments depends on the velocity of the water and the size of the particles in transport (Johnston, 1991). According to Johnston (1991), erosion occurs at velocities above a threshold for a given particle size, transport occurs at velocities above the critical erosion velocity, but above the threshold velocity for particle precipitation. There is a geomorphic threshold where rising water velocity inhibits deposition, however, the exact nature of this process is poorly understood (Hupp and Bazemore, 1993). Stream power, or water velocity, can be calculated using the Manning Equation (Brooks et al., 1997):

$$V = 1.49/n * R^{2/3}S^{1/2}$$

Where V = Velocity

R = Hydraulic radius (Cross-sectional area divided by the wetted perimeter)

S = Stream gradient

n = roughness coefficient

Velocity is positively related to the slope inclination of the channel as expressed by the stream gradient. Stream power is also affected by the depth and width of the channel, expressed by the hydraulic radius. The type substrate lining the channel including vegetation within the channel can also affect the water velocity as expressed by Manning's roughness coefficient (Brooks et al., 1997).

In wetlands, the hydraulic radius and stream gradient decreases while channel roughness increases resulting a decrease in stream velocity (Johnston, 1991). These conditions allow a stream's sediment load to precipitate into the surrounding wetland.

Within the life span of human beings, slope inclination of a particular channel may not change very much, though in geologic time it may undergo several series of change. This type of occurrence can be seen in the Grand Canyon where over the course of eons the Colorado River had carved a channel through the desert. Conversely, in alluvial flood plains and river deltas, sediment deposition within the channels leads to filling of old channels and the changes in channel pattern in the alluvial environment (Kleiss, 1996, Hatton et al., 1983). Though the erosion and deposition activities in these two examples may not change the overall stream inclination in geologic time, the shorter-term changes in slope lead to changes in channel location, which will have an effect on how sediment is deposited (and removed) from the environment.

Changes in land use activities may also have a considerable effect on patterns of sedimentation. In Kleiss's research (1996) one of the sample plots was located in an ox-bow attributed to an old river channel. Analysis was preformed on the sediment accumulated in the channel since the river changed course. It was found that sediments are filling the ox-bow at a rate of about 1 cm/year (Kleiss, 1996). The two-meter deep feature has been evident for at least 12,000 years (Kleiss, 1996). Therefore, it is clear that this feature has not been accumulating this amount of sediment through the entire course of its existence. This increase in sediment deposition may suggest increased sediment loads through the past century. However, Hupp and Bazemore (1993) also suggest that an increase in sediment accumulation levels witnessed over recent decades may be attributable to the effects – or lack thereof – of compaction, since the sediments had little time to settle.

Another example where measured sedimentation rates do not exactly match historical records can be seen in Aust's (1989) study of a cypress-tupelo forested wetland in southern Alabama. In a reference plot, buried logs were consistently encountered at 50-60 cm of depth (Aust, 1989). Analysis by ^{137}Cs indicated that the annual sedimentation rate for the previous 35 years was approximately 1.4 cm/year (Aust, 1989). The past harvest was conducted was 70 years before the study, indicating that the amount of accumulated sediment should be approximately 98 cm. If the buried logs were indeed remnants of the prior harvest, the discrepancy between their location in the soil profile and the "predicted" depth may be explained by soil compaction as previous described by Hupp and Bazemore (1993).

Sedimentation is correlated with elevation, hydroperiod, topography, and the amount of ponding (Hupp and Bazemore, 1993). Local elevation may be one of the most important variables affecting patterns in spatial sediment deposition (Hupp and Bazemore, 1993). Substantial differences in water velocities exist over short distances across inundated forested wetlands (Hupp and Bazemore, 1993). This variation in water velocity may account for substantial variation in sediment accumulation in forested bottomlands (Hupp and Bazemore, 1993). In addition, upstream channelization can reduce the upstream sedimentation rates through shortening of the hydroperiod, and thereby increasing downstream sediment loads (Hupp and Bazemore, 1993).

Vegetation plays a role in sediment deposition since herbaceous growth within the channel will slow the water and can trap suspended sediments. The type of vegetation and its particular growth habits will affect how much sediment is deposited (Aust, 1989). Grasses and shrub/scrub vegetation will provide the highest amounts of sediment trapping, due to the high amount of plant surface area exposed to the water in the stream channel. Tree boles may not trap as much sediment as shrub/scrub plant matter, however, forested wetlands rather than marshes alone, are associated with sediment removal.

The pattern and placement of vegetation is affected by the elements of topography such as levees (Hatton et al., 1983). Slight changes in the elevation will influence the types of plants that occupy ranges of micro-topography (Mitsch and Gosselink, 1993). There may also be relationships between certain plant communities and distance from the channel. Thus, an analysis of vegetative composition should also include a spatial element, however, it may be hard to draw definitive conclusions from these studies since there are complications associated with parallels between the hydraulic gradients and regional sedimentation gradients (Hatton et al., 1983).

Vegetative systems are not static and undergo changes in plant structure and composition through the process of succession, until a dynamic equilibrium is reached. Therefore, a wetland's ability to remove sediments from water will change as the pattern and composition of the plant community evolves over time.

Systems in the early stages of succession where grasses and shrubs dominate will be able to trap more sediment than a forested wetland where the majority of photosynthetic material lies well above the water table. Therefore, it is conceivable that a

50-acre marsh will trap more sediment than a 50-acre bottomland hardwood stand, given all other conditions were the same. Forested wetlands are, of course, not completely void of grasses and shrub/scrub vegetation as marshes are not completely void of trees. However, the role of the dominant vegetation type is important.

Moreover, the importance of vegetation dynamics is a concern for managers. A current marsh may be a forested wetland 50 years in the future. Furthermore, a forested wetland today, can be a marsh next year provided the appropriate natural events or management actions. Therefore, management activities must be preformed in order to maintain any given desired vegetative structure. This is particularity important if wetlands are being managed for the sole purpose of water quality improvement via sediment removal.

The physical characteristics of the channel are also important in sediment removal. Those features that lend themselves to the slowing of water velocity within the channel will increase sediment deposition. Again, the slope inclination of the channel is directly related to water velocity. In addition, the depth and width of the channel is important. Wide shallow channels expose more surface area to the water, allowing it to slow down, while narrow deep channels concentrate the water and increase velocity. In addition, the substrate lining the channel is also and important feature. Rough channel bottoms will lend themselves to water slowing more so than will smooth channel bottoms. This is most evident in urban environments where concrete lined stream channels quickly remove water from sites, reducing the hydroperiod, and thus decreasing sediment precipitation.

Silviculture and Sedimentation

Wetlands of the United States are an important ecological and economic resource. Timber production is considered a major function of forested wetlands for industrial and non-industrial private lands owners (Dubensky et al., 1993). Forested wetlands are the most exstensive wetlands in the United States and they have served as a source for high quality timber products including cypress oak and gum (Shepard, et al., 1993).

Forest management practices are a concern for forest managers because they may alter the quality of water that drains from the forest, thus affecting the quality of water in downstream environments. The most typical concern associated with the quality of water draining areas of forestry activity is erosion and subsequent sedimentation that can lead to changes in downstream channel conditions (Binkley and Brown, 1993).

Erosion is a particular concern in upland areas where colluvial activity is more apparent. Most silviculture-induced erosion in uplands is associated with forest roads, skid trail and logging decks (Yoho, 1980). Since water quality degradation by silviculture is considered non-point source pollution, it is subject to an array of regulations form the federal, state and local levels (Binkley and Brown, 1993; Tolliver, 1993; Cubbage and Flather, 1993). Many states have developed their own Best Management Practices (BMPs) that are designed to promote sustainable harvests through soil protection. State BMPs may be mandatory, such as in Maryland's Chesapeake Bay program, while other states, like Virginia, may have a voluntary approach (Hawks, et al., 1993, Floyd and MacLeod, 1993). Hawks et al. (1993) makes comparisons between voluntary and regulatory forest water quality protection programs.

Removing suspended sediments in water provides one of the primary functions of wetlands. Since silvicultural activities may expose the soil surface to the erosive forces of raindrops, it is logical that such activities may lead to rates of sediment loss in excess of what is considered acceptable levels. Silvicultural activities alter the vegetative composition of the area, thus affecting the wetlands ability to remove sediment by means of contact with plant material. In addition, the equipment used during the harvest operation can lead to changes in the mirco-topography by compaction as well as movement of the soil. Changes in the micro-topography may affect the "local" water velocity and vegetative composition within the wetland.

During the 1987-1988 flood season, sediment accumulation data was collected in cypress-tupelo forest in the Mobile-Tensaw River Delta of southern Alabama (Aust, 1989; Aust et al., 1991). Four silvicultural treatments were applied to the wetland. The first treatment was harvest by helicopter, the second treatment simulated skidder harvest, the third treatment was glyphosphate application following helicopter harvest, and the final treatment was a control (Aust, 1989; Aust et al., 1991).

In one flood season, different patterns in sedimentation were found in the four treatments (Aust, 1989; Aust et al., 1991). The helicopter logging operation resulted in the highest amount of sediment trapping, 2.2 mm, the skidder harvest had a lower average amount of trapped sediment (1.2 mm), however, this difference was not significantly less than the helicopter method (Aust, 1989; Aust et al., 1991). The control plots yielded an average one-year sediment accumulation of 1.1 mm, followed by the glyphosphate treatment that resulted in an average accretion of 0.7 mm of sediment (Aust, 1989; Aust et al., 1991).

The sediment accretion patterns attributable to different silvicultural regimes were expected (Aust, 1989; Aust et al., 1993). The helicopter harvest sites had high amounts of slash, stumps, and herbaceous growth (Aust et al., 1991). The skidded sites were similar to the helicopter sites in the amount of stumps and herbaceous growth however the slash was a more broken due to trafficking by the skidder (Aust et al., 1991). In addition, there was evidence that some of ruts created by skidding were beginning to collapse, resulting in what appeared to be erosion at some of the sediment rods used for measurement (Aust et al, 1991). Therefore, it is presumable that the amount of sediment trapped by the skidded treatment is actually greater than reported (Aust et al., 1991). In the glyphosphate application all the plant matter was killed, thereby exposing the soil surface to raindrop impact and thus increasing the potential for erosion. Moreover, the lack of vegetative protection resulted in the lowest sediment accumulation of the four treatments (Aust et al., 1991).

The results of this study were surprising because clearcutting by either method increased or maintained the yearly level of sediment accumulation when compared to the reference plots, indicating an actual improvement over pre-harvest conditions (Aust et al., 1991).

Seven years after the initial harvest treatments were applied measurements were again recorded at the study site. After the seventh year since harvest, the helicopter treatment accumulated an average of 0.116 cm/year and the skidder treatment received 0.092 cm/year (Aust et al., 1997). The difference in sedimentation between the two treatments was not found to be statistically significant (Aust et al., 1997). The reference plots accumulated an average of 0.054 cm/year, while the glyphosphate application resulted in the highest average accumulation of 0.149 cm/year (Aust et al., 1997).

The results of this four-year study are quite different from the previous study that analyzed a single flood season (Aust et al., 1991; Aust et al., 1997). Aust et al. (1997) found that there was a direct correlation between sediment accretion and ground flora vegetation after seven years. As discussed previously, the amount and type of vegetation present as it contributes to roughness and subsequently slows water and traps sediment.

It was hypothesized that the future vegetative growth in the helicopter treatment would surpass the skidder harvest due to soil damage incurred by skidder traffic that would reduce hydraulic conductivity and result in poor soil aeration (Aust and Lea, 1991; Aust et al., 1997). The results seven years after disturbance were not in accordance with the hypothesis since no significant difference was recorded in average groundflora biomass between the two harvest treatments (Aust et al., 1997). This situation may be an indication that soil damage incurred by skidder traffic may not be an insurmountable problem in forested wetlands as it is in other areas such as uplands. This may be in part due to the wetland's annual inputs of sediments and nutrients, combined with shrinking and swelling of the wetland soil (Aust et al., 1997). Wetlands are also considered very productive sites, leading to an abundance of organism activity. Rooting behavior in wetland plants reduced the effects of compaction; furthermore, burrowing activities of organisms such as worms and crayfish also help to reduce soil impacts from heavy machinery over time.

While the amount of ground flora measured between the two harvest operations was not found to be different, there were some differences detected in the composition of vegetative species. *Fraxinus profunda, Salix nigra, and Taxodium distichum,* reacted similar to helicopter and skidding harvests. However, in the skidder harvest, *Nyssa aquatica* was found in greater abundance and presumably at the expense of *Fraxinus caroliniana* (Aust et al., 1997). This abundance of *N. aquatica* was explained by the skidder impacts to the soil in the first few seasons following harvesting (Aust et al., 1997). Skidder traffic would result in soil conditions more favorable to flood tolerant species such as water tupelo, while the less tolerant species, such as Carolina ash, are less competitive.

The application of glyphosphate had an interesting effect on the floodplain vegetative composition. Essentially, the area was set back into early stages of succession as a freshwater marsh, dominated by herbaceous material with a few scattered *S. nigra* and *N. aquatica*. In the first few years following the harvest sediment trapping in this treatment was lower than the other two treatments and control. Yet, when enough time had passed to allow reseeding, dense marsh vegetation dominated, actually improving the wetlands ability to trap sediments when compared to the other treatments.

There are approximately 104 million acres of wetlands in the lower 48 contiguous United States, of these; about 50-60 million acres are in active forest management (Dubensky et al., 1993). Though there are environmental changes that occur as a result of harvesting, the forest industry continues to promote the concept that forest management practices are compatible with wetland functions and values (Dubensky et al., 1993). In fact, periodic harvesting of riparian forests may actually enhance nutrient retention functions associated with plant uptake (Walbridge, 1993). Furthermore, silviculture may provide economic incentives for landowners to maintain forested wetlands in forest cover (Dunbensky et al., 1993).

Methods and Materials

Study Objectives

The objective of this study is to examine the sedimentation rates of a tupelo-cypress wetland twelve years after timber harvest disturbance.

The specific objectives were:

1) Compare effects of the three harvest regimes on sediment deposition in intervals following disturbance. Specifically the second (1988), seventh (1993), tenth (1996), and twelfth (1998) years after harvest.
2) Compare effects of each harvest treatment on sedimentation over a twelve-year span.

The null hypothesis used to test these objectives:

1) Sedimentation rates over a twelve-year period after harvesting are not affected by harvest method.

Study Site

Location and Site Description

This study was initiated in 1986, and researched by two Ph.D. students within the N.C. State Hardwood Cooperative (Aust, 1989; Mader, 1990). Aust (1989) examined physical and chemical changes to soil following harvest while Mader (1990) researched change in vegetation.

The research site was located in the Mobile-Tensaw River Delta in southwestern Alabama straddling Baldwin and Mobile Counties (FIGURES 1 and 2). The site is a fluvial deltaic plain formed by the confluence of the 11.2 million hectare watershed of the Alabama and Tombigbee Rivers. The delta is located at 31° North latitude and 88° East longitude. The delta is bounded on the south by Mobile Bay while the Southern Pine Hills Escarpment borders the eastern and western extremities. The major channels belong to the Mobile, Middle, and Tensaw Rivers and are interconnected by several streams and bayous. In addition, other abandoned channels and dead lakes are a common feature in the delta.

FIGURE 1. Location of study site and selected channels of the Mobile-Tensaw River Delta. Adapted from Smith (1988).

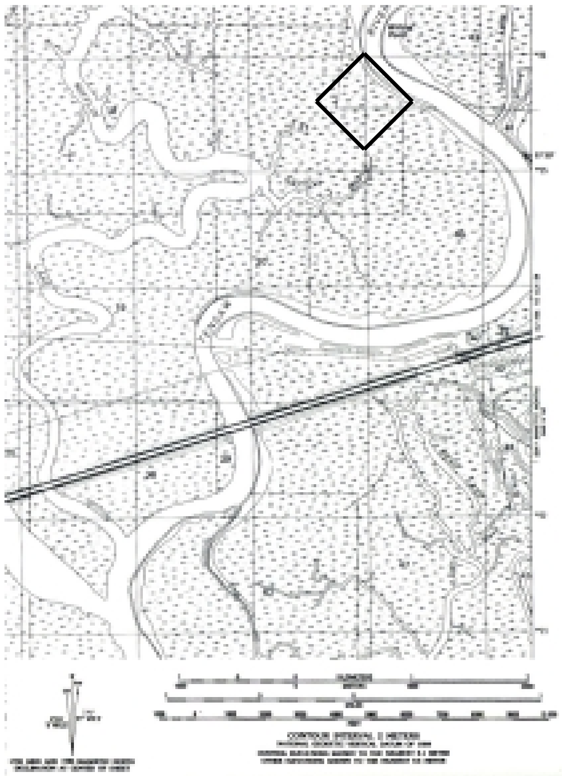


FIGURE 2. Small scale map of the Mobile-Tensaw River Delta in southwest Alabama. Area noted by box is the approximate location of the study site.

The study site is located at the delta's approximate center, on the western bank of the Tensaw River. The soil, topography, vegetation, hydrology and past use of the study site are typical of the region. The soil is a hydric fluvaquent, mixed, thermic clay (FIGURE 3). The topography of the site consisted of a higher elevation berm and lower elevation pond. The pond was relatively flat with less than a 15 cm change in elevation across the site. The berm was not included in the study. Overbank flooding of the channel typically occurs in winter and spring months and tides also regulate the level of groundwater within the study site.

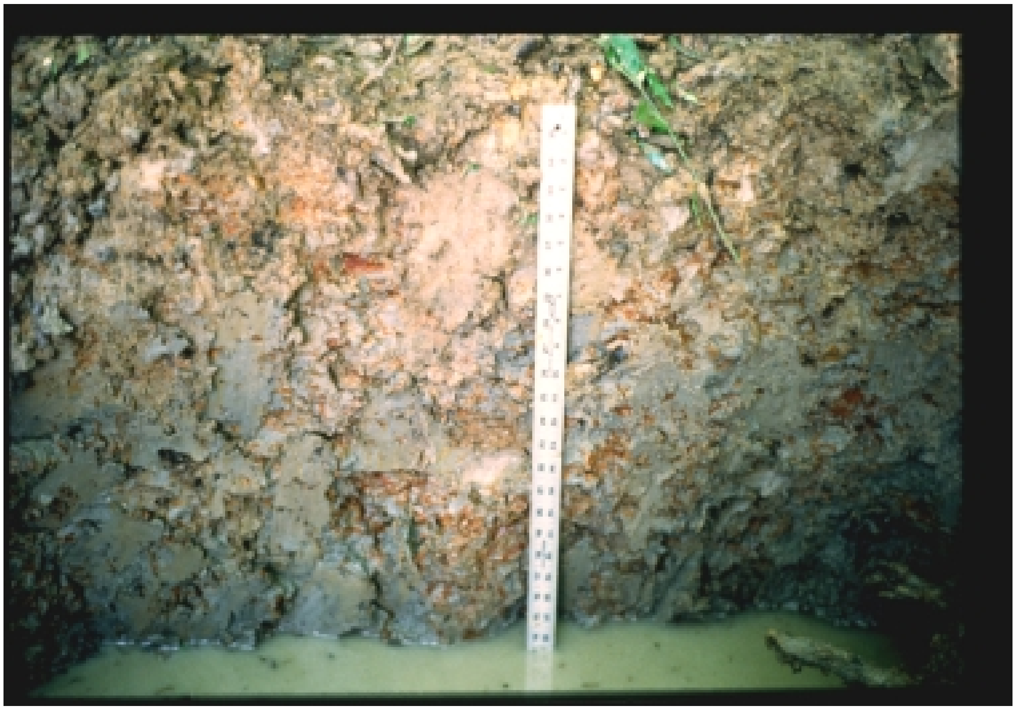


FIGURE 3. Soil pit during an extreme drought located within the study site.

Climate and Vegetation

The Mobile-Tensaw River delta potentially has a nine-month growing season and a climate considered warm-temperate to semi-tropical. However, the beginning of the growing season in wetter springs is associated more with hydrology than frost-free days. The rainfall averages 160 cm per year and is - except for the drier autumn - uniformly distributed throughout all seasons. In addition, the area is subject to rainfall from hurricanes and tropical storms.

Plant composition and productivity is influenced by hydroperiod, salinity and soil gradients. Water levels within channels and the forest stands is also affected by diurnal tides that may fluctuate an average of 0.5 m. Salinity is high on the southern fringe of the delta. In the southern reaches of the delta, inundation is frequent and organic soils support salt marshes. Marshes grade to shrub-scrublands and finally to forested wetlands as one travels northward from the southern reach of the delta. Approximately 8000 hectares of open water, 4000 hectares of marsh, 28,000 hectares of forested swamp, and 34,000 hectares of bottomland hardwoods comprise the river delta.

Water tupelo and bald cypress are common forest cover types found in the delta. Minor changes in topography influence the species composition within the forested portions of the delta, primarily due to effects on hydroperiod. In a typical cross-section of a forested delta, higher elevation berms areas are adjacent to channels and the lower downward sloping areas are found inland. Higher locations, such as berms and levees tend to support vegetation that is less flood-tolerant than the lower areas like sloughs, depressions, and ponds. Species less tolerant to flooding include *Quercus lyrata*, *Quercus nigra*, *Ulmus americana*, and *Nyssa silvatica* var. *bicolor*. Species more tolerant of flooding include *N. aquatica*, and *Taxodium distichum*.

The forest of the study site was a two-age stand with the majority of trees being about 70 years old with a few 130-year-old stems present at the beginning of the study in 1985. The younger trees regenerated after a pull-boat harvest around 1915-1920 and the older trees are probably residuals from a float-logging operation in the 1860's. At the beginning of the study *N. aquatica* comprised over 85% of the standing stock with *T. distichum* and *Fraxinus caroliniana* being the only other common species. Basal area exceeded 325 ft^2/ac (75 m^2/ha) and total volume was greater than 95 cords/acre (976 m^3/ha) (Aust, 1989).

FIGURE 4. Tupelo-baldcypress wetland found in the Mobile-Tensaw River Delta in southwest Alabama.

History

Anthropogenic use of the delta region's forest has a long history. Prior to European settlement, Native American tribes, primarily Touachis, Apalachees, Chickasaws, and Creeks, cleared fields and built mounds (Aust, 1989; Mader, 1990). In the 1700s during colonization, European settlers were able to clear some higher berm areas, but timber harvests were limited by crude float logging technology. Pull-boat cable logging systems were introduced in Louisiana by William Baptist in the 1890's and this technological advancement allowed for accelerated year-round timber harvest (Aust, 1989; Mader, 1990).

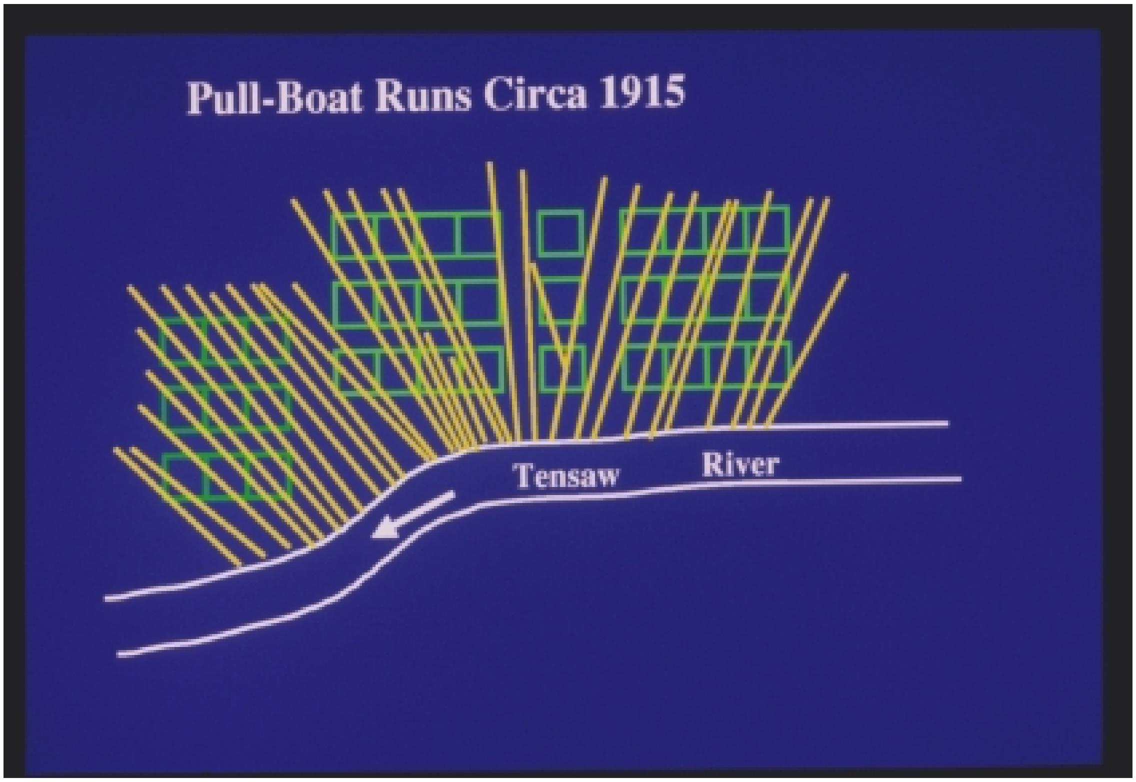


FIGURE 5. Location of pull-boat runs within the study site that were visible in 1986.

Pull-boat harvests created "runs" as stems were pulled along the ground by a cable next to the riverbank. Runs were typically laid out in a fantail pattern as seen in FIGURE 5 (Mader, 1990). The pull-boat operation usually entailed the use of approximately one dozen men who would perform all necessary tasks including felling, rafting and placement of logs on booms (Mader, 1990). This system was more efficient that previous logging operations where it was not uncommon for entire settlements to participate in logging activities, though it is intuitive that this type of harvest caused considerable damage to residual stands (Mader, 1990). Between 1895 and 1950 nearly all stands within the delta were harvested by pull-boats. Many of the harvests were complete clear-cuts and others were similar to high-grade selection. Since 1950, pull-boat operations have been replaced by skidder, cable and helicopter harvest systems.

Past logging activities are still present at the study site. Old pull-boat channels are numerous, however, the old channels (FIGURE 6) no longer open to the main river channel except when the river reaches flood stage.

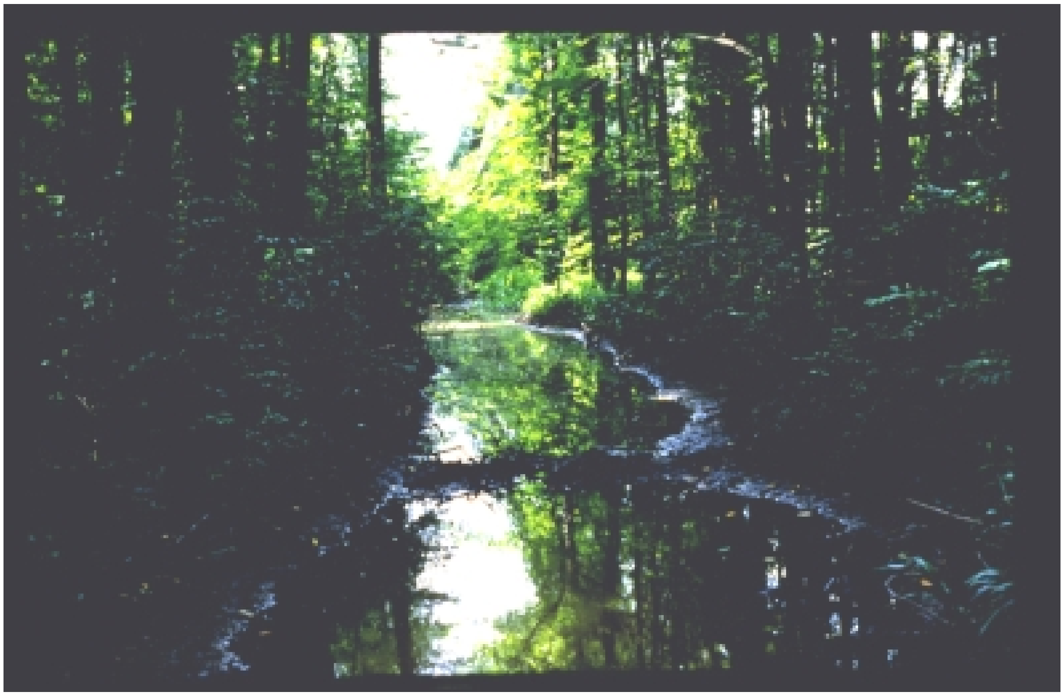


FIGURE 6. An example of a large pull boat run found within the study site in the Mobile-Tensaw River Delta.

Experimental Design

The study was designed to compare three harvesting treatments and a fourth reference plot. The treatments were:

(1) Helicopter removal of logs (HELI) using a Bell 205™ helicopter with a 2000 Kg lift capacity.
(2) Glyphosphate application (Rodeo ™) after helicopter removal of logs (GLYPH). Backpack sprayers were used to apply 1.75 percent solutions to regrowth in July 1987 and June 1988.
(3) Simulated rubber–tire skidder removal of logs (SKID) using a Franklin 105™ skidder with 86-cm-wide tires after helicopter removal of logs.
(4) Non-harvested 70-year-old (in 1986) tupelo-baldcypress reference stand (REF).

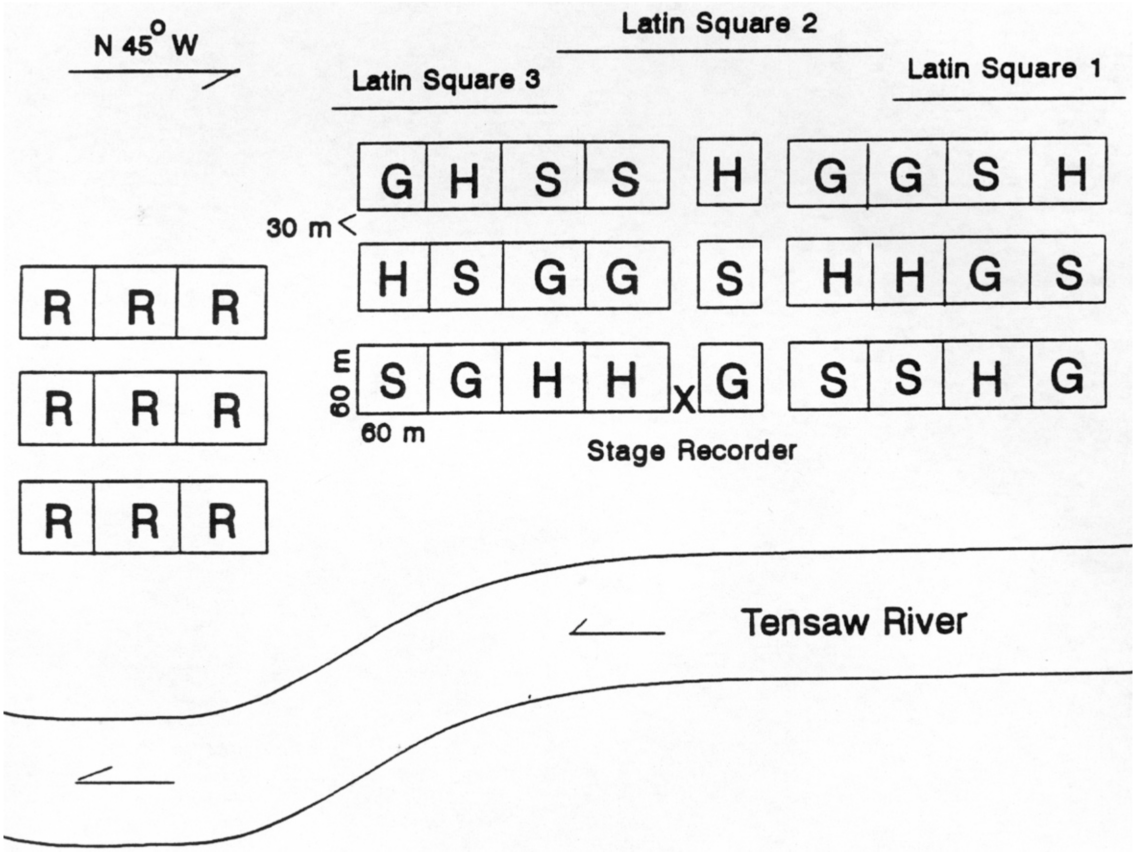


FIGURE 7. Latin Square study design. G = Glyphosphate, H = Helicopter, S = Skidder, and R = Reference [Adapted from Aust (1989)] and aerial photo showing Latin square layout in February 1997 (courtesy terraserver.com).

To apply the treatments, the entire area was fell by chainsaw and all logs were removed by helicopter (FIGURE 8). Glyphosphate was applied to treatment two using backpack sprayers during the first two growing seasons following the harvest (1987, 1988) (FIGURE 9). Skidding was simulated in treatment three using a Franklin 305 ground skidder with 86 cm-wide tires (FIGURE 10). The skidder trafficked approximately 54% of the ground. Efforts were made to duplicate the many one-pass skid trails that would be used in a ground-based log removal operation typical to the area (Aust, 1989).

FIGURE 8. View of helicopter harvesting operation in the study site located in the Mobile-Tensaw River Delta.

FIGURE 9. Application of Glyphosphate following harvesting in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta.

FIGURE 10. Simulated Skidder harvesting in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta.

The study was designed as a 3 X 3 Latin Square with each square being replicated three times (TABLE 1) (Steel and Torrie, 1980; Gomez and Gomez, 1984). This had an advantage for evaluation of trends both along and distant from the river while also increasing the power of the design by replication. The three harvesting treatments were represented in the three squares and sampling was conducted in the years; 1987, 1988, 1993, 1996, and 1998. The reference square, a single 3 X 3 Latin Square, was established in a roughly 80 year-old tupelo-cypress stand adjacent to the treatment plots. Since it was not feasible to log around a reference area, it was located 30 meters away from the study area (FIGURE 11).

FIGURE 11. Aerial view of the Latin Square layout of the experiment in the Mobile-Tensaw River Delta.

Statistical Analysis

Analysis follows standard ANOVA procedures for a Latin Square Design in Minitab version 13 at an alpha level of 0.05. Sediment accretion values in treatment plots and REF area are compared for significant differences in each year of sampling (1988, 1993, 1996, 1998) by t-tests and median separation tests using the Mann-Whitney test (*NOTE: in Minitab version 13 the Mann-Whitney test is actually the Wilcoxon Rank Sum test. These tests yield identical p-values and only differ in their test statistics* (Hollander and Wolfe, 1999; MINITAB Manual, 2000). In the 1993 data, Pearson's Correlation Coefficient is calculated in Minitab between the average sediment accretion and average overstory wet biomass for 27 treatment plots. In the 1998 data, mean and median separation tests are conducted on blocks, columns, rows and treatments. Pearson's Correlation Coefficient is also calculated between the average depth to rust (measured on the sediment rods in 1998) and total average sediment accretion for the 27 treatment plots. Following the correlation test, mean and median separation tests are conducted between rust measurements in the treatments and REF area. Pearson's Correlation Coefficient is also calculated between average Basal Area (ft^2/acre) and total sediment accretion. Similar to the rust data, mean and median separation procedures are conducted on the treatments and basal area values. In the temporal analyses, each treatment is compared for significant differences between intervals over the twelve-year span using paired t-tests and its nonparametric counterpart the Wilcoxon Signed Rank procedure (Hollander and Wolfe, 1999, Ott, 1993).

TABLE 1. Generalized ANOVA for a 3 X 3 Latin Square replicated three times as used by this study.

Source	**d.f.**
Squares	s-1
Rows in Squares	s(r-1)
Rows	r-1
Rows * Squares	(r-1)(s-1)
Columns in Squares	s(c-1)
Columns	c-1
Columns * Squares	(c-1)(s-1)
Treatments in squares	s(t-1)
Treatments	t-1
Treatments * Squares	(t-1)(s-1)
Time in Squares	s(p-1)
Time of Observation	(p-1)
Time * Treatment	(p-1)(r-1)
Time * Squares	(p-1)(s-1)
Time * Tmt * Squares	(p-1)(r-1)(s-1)
Experimental Error	s(r-1)(r-2)(p-1)
Observational Error	r^2(n-1)(p-1)s
Total (corrected)	sr^2np-1

Measurements

Sedimentation (erosion) rods were systematically placed at nine locations in each plot in the initial study in 1986. The rods were made of steel reinforcement rod with a metal washer welded to the rod so that is may be placed flush with the soil surface (FIGURE 12). Roughly 60 cm of the bar was placed below the soil, leaving 30 cm aboveground (Aust, 1989). Sediment accumulation and erosion could be measured by any changes in the soil surface relative to the washer (Aust, 1989, Brooks et al, 1991, Szabo, 1998). After field measurements, the rods were repositioned with the washers again flush with the soil surface (Szabo, 1998).

Sediment data was collected in 1988, 1993, 1996, and 1998. Due to logistical problems, sediment data for the first year following disturbance (1987) was only available for the reference plots (Aust, 1989).

FIGURE 12. Sediment rod used in experiment in a tupelo-baldcypress forested wetland in the Mobile-Tensaw River Delta of southwestern Alabama.

Results and Discussion

Sedimentation Rates Year-by-Year
Two years after disturbance – 1988

Following two years of recovery from harvest disturbance, no significant difference was detected in the sediment accretion in blocks, columns, rows, and harvest treatments within the experiment (TABLE 2).

Selected published studies report yearly sediment accumulation at a range between 0.08 and 2.4 cm/year (TABLE 3). Mean annual sediment accretion values in 1988 fall within this range while median rates fall at the lower end of the range, particularly the SKID and GLYPH treatments whose median rate is zero cm (TABLE 2).

Harvesting removes the forest canopy and exposes the ground to the erosive forces of raindrops. During this time the potential for erosion exists. Both the parametric and nonparametric tests for location shift failed to detect a significant difference in the amount of sediment trapped by the treatment plots and reference area between 1987 and 1988. This suggests that the erosion hazard that is often a concern to harvest areas in mountainous regions is not apparent to the same degree in the forested wetland. This is emphasized by the forested wetland's low slope, high amounts of slash remaining after harvest, and rapid re-growth of vegetation. Wetlands are associated with sediment capture and the results from this analysis illustrate that factors other than the presence of forest vegetation influence sediment capture in wetland areas.

TABLE 2. Effects of timber harvest treatments (TMT) on sediment accretion in cm on measurement periods recorded in 1988, 1993, 1996, and 1998 in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta in southwest Alabama.

	TREATMENT			
Measurement Period	**HELI** (cm/period)	**SKID** (cm/period)	**GLYPH** (cm/period)	**REF** (cm/period)
1987-1988				
MEAN	2.2 a[1]	1.3 a	0.7 a	1.1 a
(STD.DEV.)	(5.1)	(4.9)	(2.7)	(1.2)
MEDIAN	1.0 a[2]	0.0 a	0.0 a	1.0 a
(95%CI on true median)	(-1.0)	(+1.0)	(+1.0)	(-1.0)
1988-1993				
MEAN	8.2 b	6.7 b	10.7 c	3.8 a
(STD.DEV)	(3.8)	(4.5)	(6.0)	(3.3)
MEDIAN	8.2 c	6.5 b	9.0 d	3.5 a
(95%CI on true median)	(+/-0.7)	(+/-1.3)	(+/-1.0)	(+/-0.4)
1993-1996				
MEAN	7.8 b	6.8 b	10.3 c	4.1 a
(STD.DEV)	(4.9)	(5.9)	(6.1)	(2.9)
MEDIAN	7.6 b	6.0 b	8.9 c	3.5 a
(95% CI on true median)	(+/-1.0)	(+/-2.3)	(+/-1.3)	(+/-0.6)
1996-1998				
MEAN	2.0 a	1.9 a	2.9 b	1.2 a
(STD.DEV.)	(3.1)	(2.9)	(2.1)	(1.0)
MEDIAN	1.8 b	1.5 b	3.0 c	1.2 a
(Margin of Error)	(+/-0.3)	(+/-0.4)	(+/-0.6)	(+/-0.3)
TOTAL (1987-1998)				
MEAN	20.3 b	16.3 b	24.7 c	10.2 a
(STD. DEV.)	(10.1)	(12.5)	(12.8)	(6.43)
MEDIAN	18.3 c	15.1 b	19.5 c	9.5 a
(95%CI on true median)	(+/-2.0)	(+/-3.4)	(+/-3.1)	(+/-1.0)

[1] For each row containing mean values, different letters are significantly different according to a t-test at an alpha level of 0.05.
[2] For each row containing median values, different letters are significantly different according to the Mann-Whitney at an alpha level of 0.05.

TABLE 3. Sediment accretion for various wetlands reported by several authors in the United States.

Location	Accretion (cm/year)	Method	Author
White Clay Lake NE Wisconsin	2.4 (streamside) 1.3 (alluvial zone) 0.5 (non-alluvial zone)	^{137}Cs and soil morphology	Johnston et al., 1984
Marsh, Long Island, NY	0.47 and 0.63	^{210}Pb	Armentano and Woodwell, 1975
Salt Marsh, Louisiana	1.35 (within 7 m of tidal stream) 0.75 (45 m from stream)	^{137}Cs	DeLaune et al., 1978
Riparian Forest, Georgia	0.22	depth to argillic horizon	Lowrance et al., 1986
Iowa Wetland	1.7	^{137}Cs	Eckblad et al., 1977
Alluvial Cypress Swamp Illinois	0.8	sediment traps	Mitsch et al., 1979
Bottom Land Hardwood Wetland, Cache River, Eastern Arkansas	0.86 (natural levees) 0.41 (natural levees) 0.27 (natural levees) 1.606 (swamp) 0.21 (swamp) 0.257 (swamp) 0.267 (ridge) 0.13 (ridge) 0.08 (ridge) 0.6625 (abandoned channels) 0.1775 (abandoned channels) 0.18 (abandoned channels)	clay pads ^{137}Cs dendrogeomorphic method clay pads ^{137}Cs dendrogeomorphic method clay pads ^{137}Cs dendrogeomorphic mthod clay pads ^{137}Cs dendrogeomorphic method	Kleiss, 1996
Riparian Zone, Virginia	0.07-0.57	dendrogeomorphic analysis	Hupp et al., 1993
Bottomland Hardwood Swamp, West Tennessee	0.24-0.28	dendrogeomorphic analysis	Hupp and Bazemore, 1993

The rapid re-growth of vegetation in both the HELI and SKID treatment can be observed in FIGURES 13 through 16. Trafficking by the skidder increased variation in micro-topography in the treatment plot that resulted in greater abundance and growth of several sedge species in the SKID treatment seen in FIGURE 15 (Mader, 1990). The ground flora in SKID and HELI treatments consist of herbaceous matter with a few scattered woody stems. Physical characteristics of vegetation influence a wetland's ability to trap sediments suspended in floodwaters. Therefore, the similarity in sediment accretion experienced in all three treatments may partially be explained through the relatively similar physical make-up of herbaceous matter in all three harvest treatments.

FIGURE 13. HELI harvest plot at the end of the first growing season following harvesting disturbance in a tupelo-baldcypress wetland in southwest Alabama.

FIGURE 14. The same HELI harvest plot in the middle of the second growing season following harvesting disturbance in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

FIGURE 15. SKID plot during the first growing season following harvest disturbance in a tupelo-baldcypress swamp in the Mobile-Tensaw River Delta of southwest Alabama. The SKID plot's increased variation in micro-topography increased the abundance and growth of several sedge species (Mader, 1990).

FIGURE 16. SKID treatment two years after harvesting disturbance in a wetland along the Mobile-Tensaw River delta of southwest Alabama.

FIGURE 17 displays the understory in the REF plot. The physical vegetative composition is quite different from the harvest treatments, the notable differences being the presence of mature trees and lack of ground flora. Though the physical similarity of vegetation may explain the statistical similarity of the three harvest treatments, it does not explain why the harvest treatments are also statistically similar to the REF plot. In fact, this situation suggests that recently harvested areas in the Mobile-Tensaw River Delta trap similar amounts of sediment as a mature cypress-tupelo forest.

FIGURE 17. Understory of a tupelo-baldcypress (REF) wetland in the Mobile-tensaw River Delta of southwest Alabama.

In the REF plot, the closed canopy formed by mature trees intercepts light and reduces the amount of solar energy that reaches the forest floor. Since the overstory exploits great quantities of incoming solar energy, there is little light available for understory vegetation. As a result, the understory appears sparse in herbaceous cover when compared to the other treatments.

Harvesting exposes the ground to the erosive forces of rainfall (FIGURE 18). Herbaceous vegetation quickly colonizes the newly harvested plot, however, the colonization does not occur overnight. There is a time gap between harvested sites evolving from barren ground to complete herbaceous cover. During the time when ground is exposed it may be flooded by erosive waters.   So while the thick herbaceous cover seen in the harvest treatments two years after harvesting may suggest great sediment trapping capability, the amount of sediments trapped by the growth in the early years may only compensate for prior erosion losses.

FIGURE 18. Appearance of a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama shortly after timber harvesting. (HELI plot)

Histograms in FIGURE 19 display the frequency of individual recorded sediment accretion measurements for all three treatments and the REF plot between the first and second year following harvest. Note that there are some negative values of sediment accretion recorded in the SKID and GLYPH treatment. These negative values suggest that erosion might have been witnessed in the net sediment accumulation values measured in the harvest treatments, though possible sediment losses fail to contribute significantly in mean and median accumulation after two years of harvest recovery.

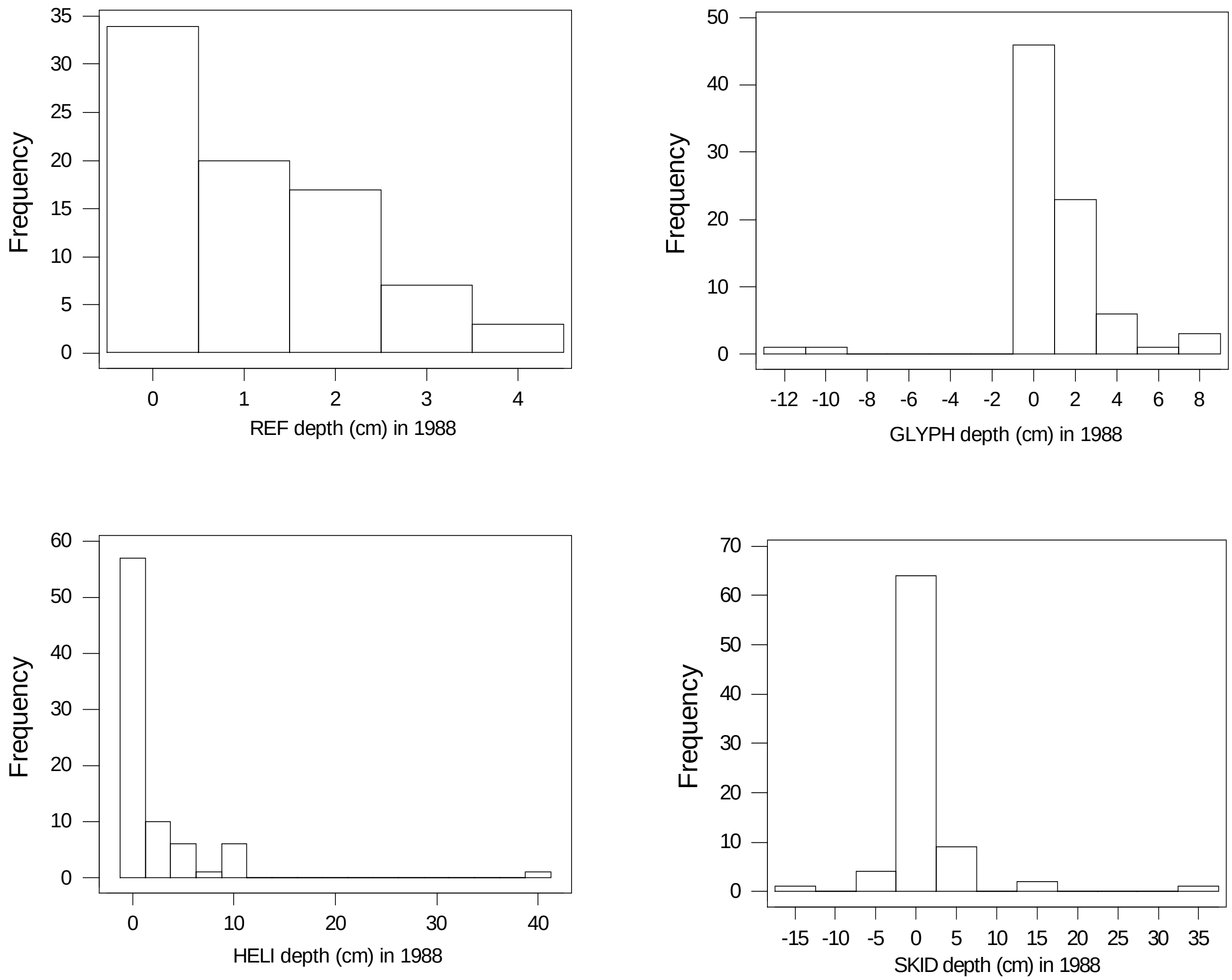


FIGURE 19. Histograms of sediment accretion between 1987 and 1988 in three harvest treatments and REF in a tupelo-baldcypress forested wetland in the Tensaw-Mobile River Delta of southwest Alabama.

Seven years after disturbance – 1993

Seven years of recovery following harvesting disturbance the ANOVA procedure revealed only treatment differences within the Latin Square experiment. The parametric analysis finds the GLYPH treatment with a mean of 10.7 cm, accumulating a significantly higher amount of sediment than the HELI and SKID treatments with means of 8.2 and 6.7 cm, respectively (TABLE 2). In addition, the REF area accumulated significantly less sediment, mean of 3.8 cm, during the five-year period. The nonparametric analysis yielded results similar to those obtained from the parametric test, though it detected a significant difference between the median sediment accumulation in the HELI (8.2 cm) and SKID (6.5 cm) treatments (TABLE 2).

TABLE 4 contains the annual sediment accretion values based on measurement made during a given year and the time since the previous measurement. These values, ranging from 0.7 to 2.1 cm/year based on 1993 measurements fall within the published range of 0.08 to 2.4 cm/year of wetlands (TABLE 3).

TABLE 4. The effects of three harvesting treatments on annual sediment accretion at intervals following harvesting disturbance in a tupelo-baldcypress wetland in the Tensaw-Mobile River Delta of southwest Alabama.

	TREATMENT			
Annual Sediment Accretion[1]	**HELI** (cm/year)	**SKID** (cm/year)	**GLYPH** (cm/year)	**REF** (cm/year)
1987-1988				
MEAN	2.3 a[2]	1.4 a	0.8 a	1.1 a
(STD.DEV.)	(5.1)	(5.3)	(3.0)	(1.2)
MEDIAN	1.0 a[3]	0.5 a	0.5 a	1.0 a
(95% CI on true median)	(+/-0.5)	(+/-0.5)	(+0.5)	(+/-0.5)
1988-1993				
MEAN	1.6 b	1.3 b	2.1 c	0.8 a
(STD.DEV.)	(0.8)	(0.9)	(1.2)	(0.7)
MEDIAN	1.6 c	1.3 b	1.9 d	0.7 a
(95% CI on true median)	(+/-0.2)	(+/-0.4)	(+/-0.3)	(+/-0.1)
1993-1996				
MEAN	2.6 b	2.2 b	3.4 c	1.4 a
(STD.DEV.)	(1.6)	(2.0)	(2.0)	(1.0)
MEDIAN	2.5 b	2.2 b	3.1 c	1.3 a
(95% CI on true median)	(+/-0.3)	(+/-0.5)	(+/-0.5)	(+/-0.1)
1996-1998				
MEAN	0.8 a	0.8 a	1.4 b	0.6 a
(STD.DEV.)	(1.2)	(0.9)	(1.0)	(1.1)
MEDIAN	0.9 b	0.8 b	1.5 c	0.6 a
(95% CI on true median)	(+/-0.2)	(+/-0.2)	(+/-0.3)	(+/-0.1)

[1] Annual sediment accretion was calculated by taking the measurement recorded in the given period and dividing it by the number of years between the previous readings. For example measurements taken in 1993 were divided by five, since there was a five year gap between the last measurements in1988.
[2] for each row, values with different letters are significantly different according to t-tests at an alpha level of 0.05.
[3] for each row, values with different letters are significantly different based on Mann-Whitney tests at an alpha level of 0.05.

FIGURE 20 graphically displays the significantly greater sediment accretion in the three harvest treatments compared to the reference site. The reference plot is a closed canopy forested wetland. The thick canopy filters the sunlight penetrating to the forest floor surface, thus reducing herbaceous ground cover (Zaebst et al., 1994). Since harvesting reduces tree cover, sunlight is allowed to reach ground level and promote growth of herbaceous cover. Thick herbaceous cover slows water flow and allows suspended sediments to precipitate from over-bank flow.

Furthermore, FIGURE 20 illustrates possible violation of the Gaussian distribution assumption required for the parametric statistical procedures. The SKID and HELI data appear Gaussian-like, however, the distribution of the GLYPH and particularly the REF area data points is quite skewed, suggesting that interpretation of the non-parametric results will better protect the set alpha level of 0.5%. Again, however, the only disagreement in statistical procedures was a significant difference detected between the HELI and SKID treatments in the nonparametric analysis.

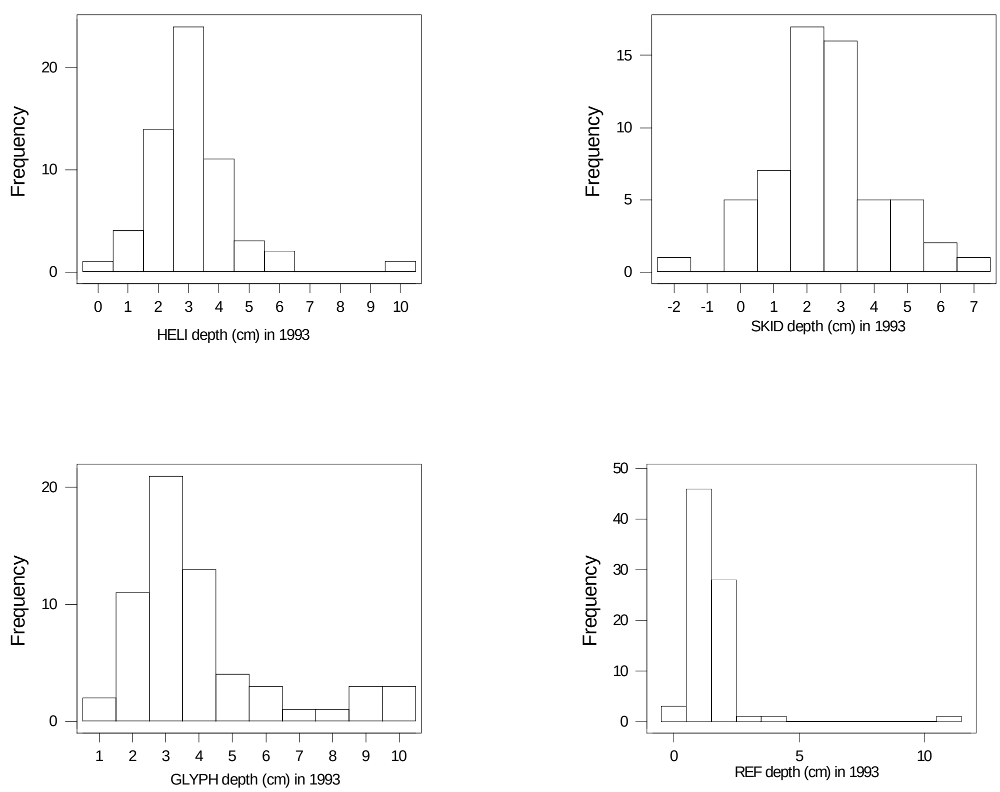


FIGURE 20. Histograms of sediment accumulation and loss in cm between the years 1988 and 1993 for three harvest treatments and reference plot in a tupelo-baldcypress wetland in southwest Alabama.

The nonparametric statistical procedure finds the HELI treatment with a median of 8.2 cm significantly greater than the SKID treatment with a median sediment accretion value of 6.7 cm (TABLE 2). The p-value yielded by the nonparametric test is 0.02. Interestingly, the p-value resulting from the parametric test is 0.05, so while it fails to reject the null hypothesis of no treatment difference, it is very close to the alpha level cut-off. In FIGURE 20 some negative values of sediment accretion can be seen. These negative values, none of which are seen in the HELI treatment, are responsible for lower mean and median sediment accretion values reported in SKID treatment. Negative values suggest sediment loss and/or erosion. Trafficking by the skidder during harvesting could have produced ruts and compacted soil leading to erosion. Though erosion appears to be occurring in the SKID treatment between the second and seventh year after harvesting it does not appear to be a detriment to the plot's ability to capture sediment, as its mean and median sediment accretion values remain significantly higher than the REF area.

The application of the herbicide glyphosphate after harvest results in the death of woody stems that remain on site. The appearance of the GLYPH sites resembles a freshwater marsh. Without the application of glyphosphate, root systems of harvested trees remain alive and woody stems in the form of stump spouts are quickly re-established (FIGURE 21). Since stump sprouts do not need to develop a new root system, they are able to grow into mature trees rapidly (FIGURE 22). The trees compete with ground flora for light and soil nutrients. Without the sprouts, however, ground flora dominates, though it is likely a matter of time before trees are established via seed origin.

FIGURE 21. One of the many stump spouts present in the HELI and SKID treatments in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

FIGURE 22. Stems in the SKID treatment 7 years following harvesting disturbance in a tupelo-baldcypress wetland the Mobile-Tensaw River Delta in southwest Alabama.

FIGURE 23 displays a strong negative relationship (Pearson's correlation coefficient of –0.704) between average overstory biomass and average sediment accretion. Zaebst et al. (1994) found a strong positive relationship between average ground flora biomass and sediment accretion after seven years of recovery following harvest disturbance (TABLE 5). Since overstory biomass consists of tree components, large amount of overstory biomass suggest a closed canopy system whereby the ground flora is shaded-out of dominance. Therefore, the negative correlation found between average overstory biomass and sediment accretion is complimentary of the positive relationship found by Zaebst et al. (1994).

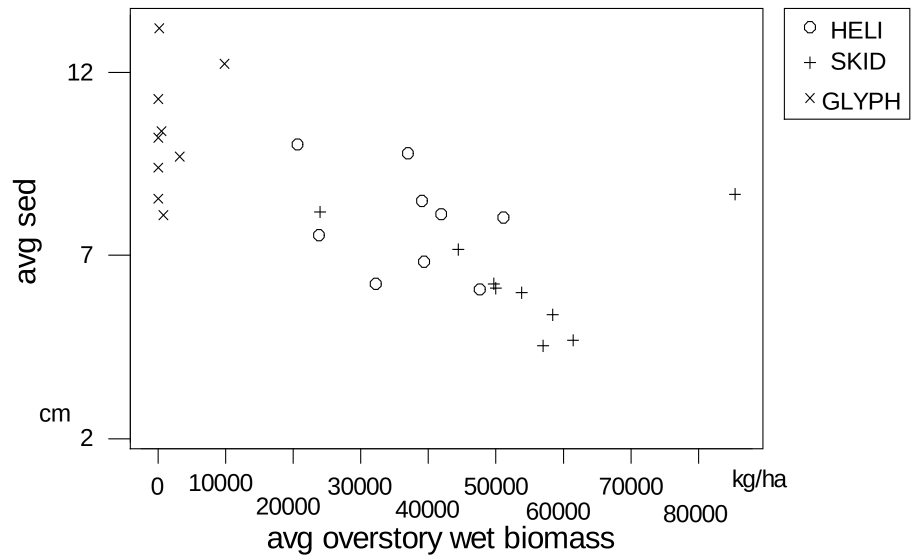


FIGURE 23. The effects of three harvesting treatments on sediment accumulation. Average overstory wet biomass (kg/ha) and average sediment accretion (cm) in HELI, SKID, and GLYPH plots in a tupelo-baldcypress swamp in the Mobile-Tensaw River Delta of southwest Alabama.

TABLE 5 – Average annual sediment accumulation (1988-1993) at 7 years by treatment (letters represent results of t-tests preformed between treatments at alpha = 0.10) Reproduced from Zaebst (1994).

Treatment	Average sediment (cm/year)	Average groundflora (kg/ha)
REF	**0.54 a**	**1516 a**
SKID	**0.92 b**	**5202 b**
HELI	**1.16 b**	**5649 b**
GLYPH	**1.49 c**	**13121 c**

Ten years after disturbance – 1996

Following ten years of recovery from harvesting, the ANOVA procedure detects only treatment differences. All harvest treatments contain significantly greater sediment than the REF area by both the parametric and nonparametric analyses (TABLE 2). The REF area trapped an average of 4.1 cm during the three years after the sediment rods were measured and relocated in 1993. The GLYPH treatment captured an average of 10.3 cm of sediment, significantly greater than the average sediment accretion of 7.8 and 6.8 cm in the HELI and SKID treatments, respectively (TABLE 2). In addition, parametric and non-parametric analyses yielded identical results for significance in tests for location shift.

Ranging from 1.3 to 2.6 cm/year, the annual sediment accretion for the HELI, SKID, and REF sites fall within the upper end of the published range of sediment accretion in wetlands (TABLES 3 and 4). The mean value of sediment accretion, 3.1 and the median 3.4 cm/year, measured in the GLYPH treatment exceed the upper limits of the published range.

The histograms in FIGURE 24 graphically display the significantly lower average sediment accretion in the REF plots when compared to the three harvest treatments. Similar to the 1993 sediment data, the significantly lower accretion in the reference plots may again be attributed to a closed canopy forest system with reduced amounts of ground flora, thereby lessening the wetland's ability to intercept suspended sediment.

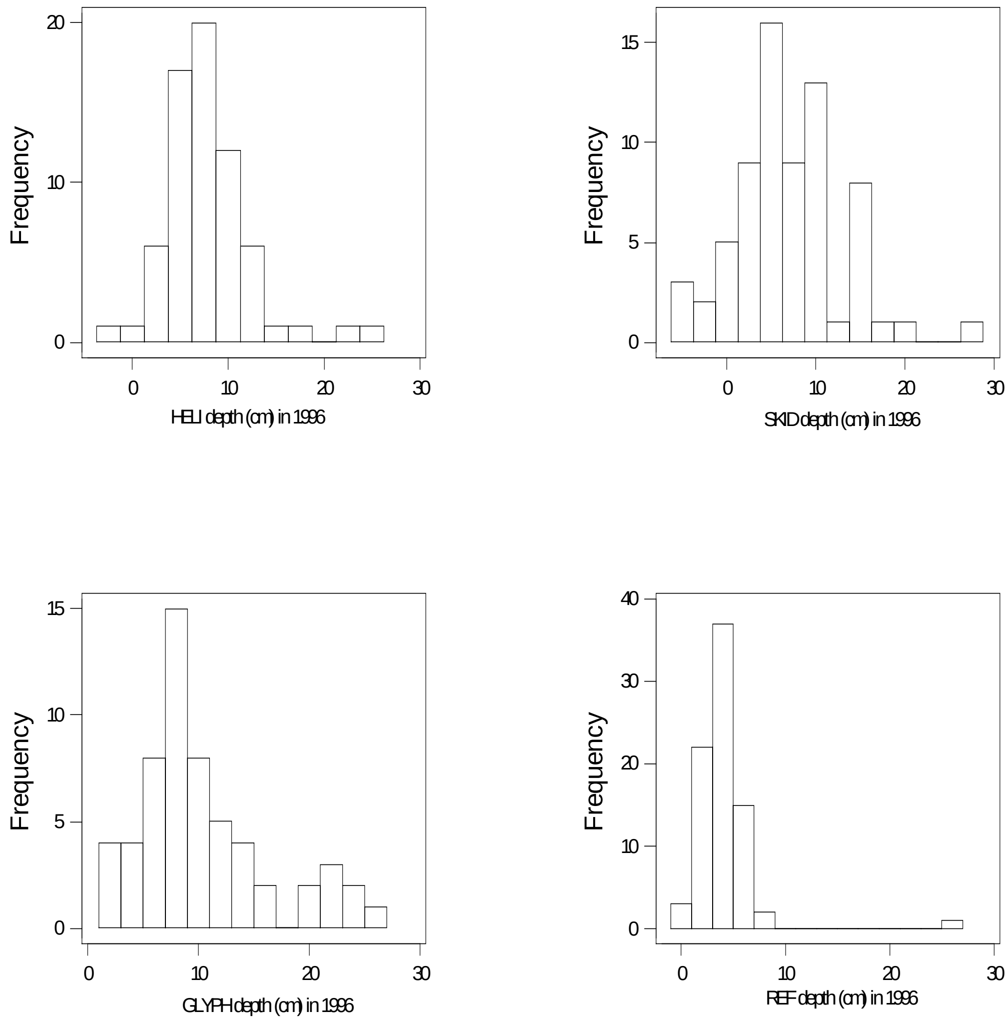


FIGURE 24. Histograms of sediment accretion and loss in cm between the years 1993 and 1996 for three harvest treatments and reference plot in a tupelo-baldcypress wetland in southwest Alabama.

Once again, the significantly greater sediment accretion in the GLYPH sites may be attributable to thick ground flora cover seen in FIGURE 25. As seen in FIGURE 26 the vegetative structure of the HELI treatment resembles a forested wetland more than a freshwater marsh.

FIGURE 25. GLYPH treatment 10 years after harvesting disturbance in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

FIGURE 26. Stems in the HELI treatment 10 years following harvesting disturbance in a tupleo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

12 years after disturbance – 1998

After 12 years of recovery from harvesting disturbance, ANOVA procedures detect significant effects within the blocks, columns, rows and treatments of the experiment (TABLES 2 and 6).

TABLE 6. Effects of BLOCKS, ROWS, and COLUMNS on sediment accretion (cm) recorded between 1996 and 1998 in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama

SECTION	MEAN(STD.DEV)	MEDIAN (95% CI on true median)
BLOCK		
1	3.0(3.3) b[1]	2.4(+/-0.6) b[2]
2	2.3 (3.0) ab	2.0 (+/-0.5) ab
3	1.5 (2.0) a	1.6 (+/-0.4) a
ROW		
1	2.4 (1.7) b	2.1 (+/-0.3) c
2	1.9 (1.6) a	1.6 (+/-0.3) b
3	1.5 (3.6) a	1.1 (+/-0.1) a
COLUMN		
1	1.8 (2.0) b	1.8 (+/-0.3) a
2	2.7 (3.5) c	1.9 (+/-0.3) a
3	1.4 (1.6) a	1.2 (+/-0.2) a

[1] for each column and section (i.e. BLOCK, ROW, COLUMN) values with different letters are significantly different according to a t-test at alpha of 0.05.
[2] for each column and section (i.e. BLOCK, ROW, COLUMN) values with different letters are significantly different according to the Mann-Whitney test at an alpha level of 0.05.

Prior to 1998 effects of blocks, rows and columns were not found significant so it is interesting that after 12 years of recovery from harvesting disturbance evidence of their influences are being witnessed. Wetland hydrology is complex and influenced by many factors, so while the results may seem unusual compared to previous analyses, this does not necessarily mean they are completely invalid.

In the histograms in FIGURE 27 a relationship between row and recorded sediment accretion in the two years between 1996 and 1998 can be seen. Both statistical procedures detect significantly higher sediment accretion row 1, located closest to the river channel when compared to row 3, located farthest from the river channel.

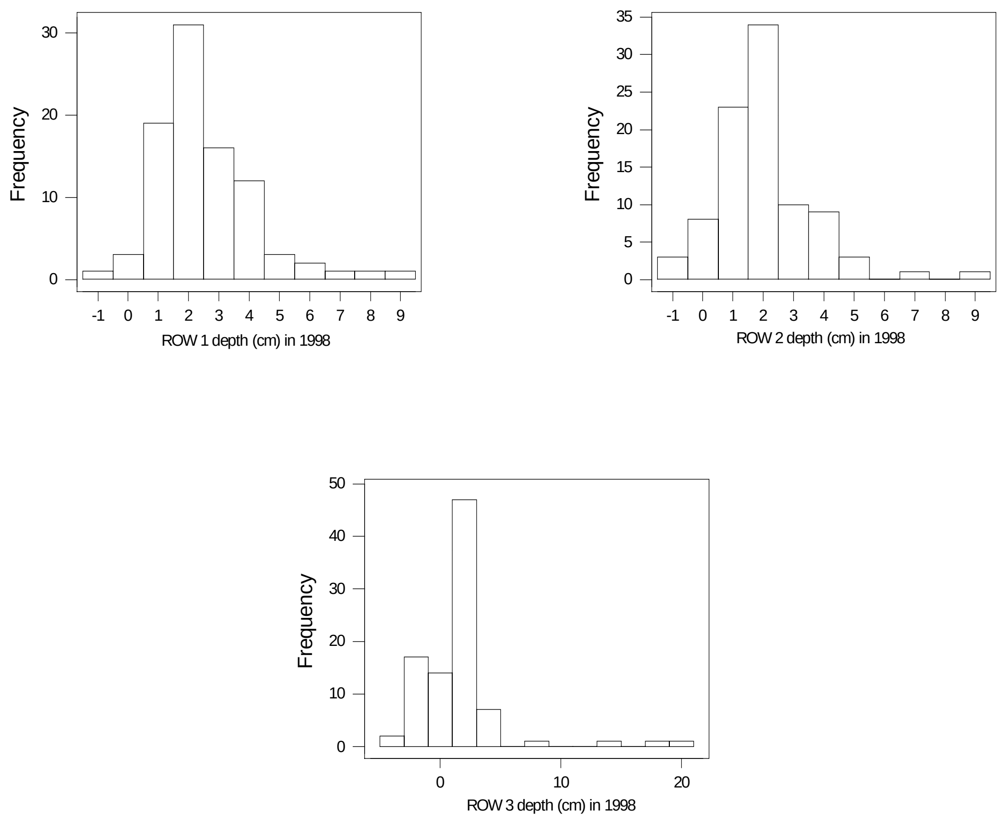


FIGURE 27. Histograms of sediment accretion and loss in cm between the years 1996 and 1998 for three rows in the Latin Square experiment in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

The parametric procedure fails to detect a significant difference between the mean of 1.9 cm of the second and mean of 1.5 cm in the third rows of the Latin Squares (TABLE 6). However, the nonparametric procedure found the median sediment accretion value of 1.6 cm in the second row significantly greater than the median of 1.1 cm in the third row (TABLE 6). The mean and median values are nonetheless, roughly similar and the non-Gaussian appearance of the data distribution in FIGURE 27 suggest that the conflict in procedure relates to violation of the Gaussian distribution assumption required in the parametric tests. Again, the nonparametric procedure will better protect alpha. Therefore, row 1, closest to the river channel accretes significantly more sediment than row 2 that in turn accretes significantly more sediment than row 3 during the two years between 1996 and 1998.

Due to the design of the experiment, column effects are a bit more complicated to explain. For each of the three Latin Squares the first column is the rightmost column of each block when looking at FIGURE 7. Since the river flow is from right to left when looking at FIGURE 7, column effects are related to location upstream or downstream the channel. Since the three Latin Squares are adjacent to each other column effects may be very difficult to sift from the data analysis. It would be expected that those columns upstream would capture more sediment that those down stream since sediments loads would be greater assuming factors such as slope and vegetative composition remained constant. The results in TABLE 6 do not support this idea, bearing in mind that vegetative composition is not constant making the results awkward to interpret. However, analysis of the columns from a single Latin Square may agree with the notion that columns upstream accrete more sediment than those downstream. Therefore, analysis of block differences may better address the issue of column effects in the realm of this experimental design.

Results from analysis of the blocks agree with the idea that stream location, be it up or down, has an effect on sediment accretion between 1996 and 1998. Block 1, which is the furthest upstream accretes and mean of 3.0 cm and a median of 2.4 cm of sediment during the two-year period (TABLE 6). On the other hand, block 3, located the furthest downstream accretes significantly less than block 1 with a mean of 1.5 cm and a median of 1.6 cm (TABLE 6). Block 2, with a mean of 2.3 cm and a median or 2.0 cm is not significantly different from either block 1 or 2.

As in analyses from previous measurement periods, harvest treatment contributes to differences witnessed in sediment accretion. Similar to data collected in 1993 and 1996 the GLYPH treatment with a mean of 2.9 cm and a median of 3.0 cm, continues to accrete significantly greater sediment than the other treatments and REF area (TABLE 2). In FIGURE 29, the GLYPH plot appears less-like a freshwater marsh than it was seen just two years prior in 1996 (FIGURE 25). However, the woody vegetation appears smaller than that of the HELI and SKID treatments (FIGURES 30 and 31).

The parametric procedure fails to detect a difference in sediment accretion in the HELI, SKID and REF area. This would suggest that the vegetation in the HELI and SKID plots is growing more like the mature forest in the REF area. In FIGURES 30 and 31, the HELI and SKID treatments appear dense with stems, though the plots still look "thicker" and container smaller trees than the REF area depicted in FIGURE 17.

The nonparametric analysis, similar to results from 1996, suggest that the HELI and SKID treatments continue to accrete significantly greater sediment than the REF area (TABLE 2). FIGURE 28 displays the distribution of sediment data collected between 1996 and 1988, once again the non-Gaussian appearance of the data is clear, suggesting that nonparametric analysis will better protect alpha.

The annual rates of sediment accretion based on 1998 measurements, range from 0.6 to 1.5 cm/year. These values fall within the published range for annual wetland sediment accretion (TABLES 2 and 3).

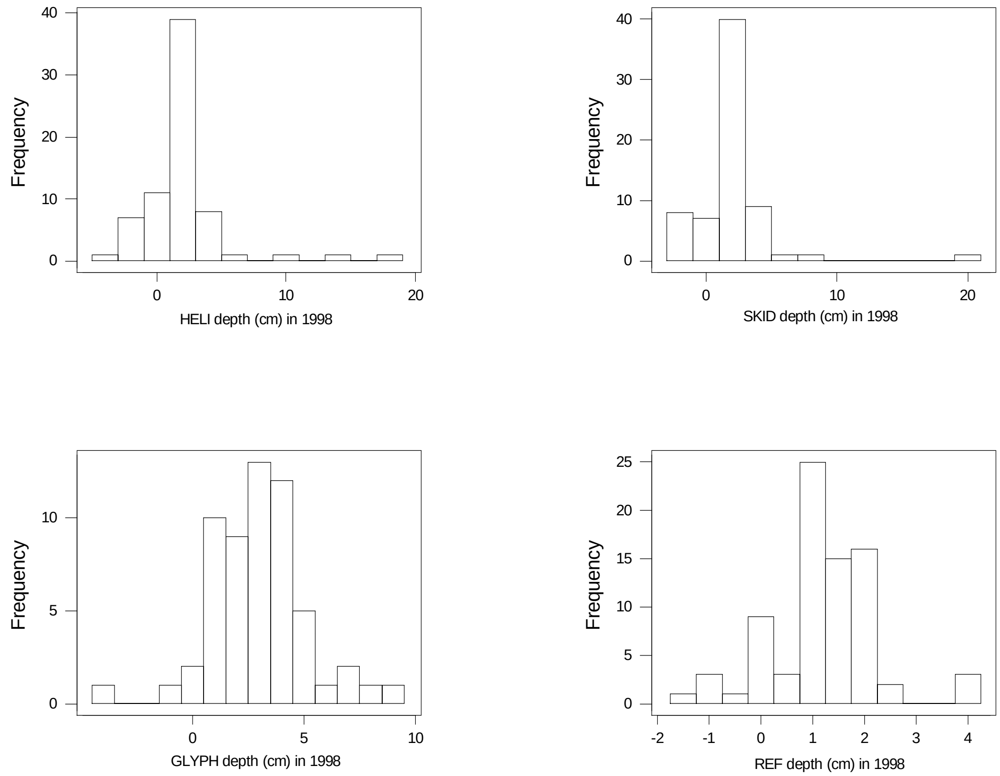


FIGURE 28. Histograms of sediment accretion and loss in cm between the years 1996 and 1998 for three harvest treatments and reference plot in a tupelo-baldcypress wetland in Mobile-Tensaw River Delta of southwest Alabama.

FIGURE 29. GLYPH treatment 12 years after harvesting disturbance in a tupelo-baldcypress wetland in Mobile-Tensaw River Delta in southwest Alabama.

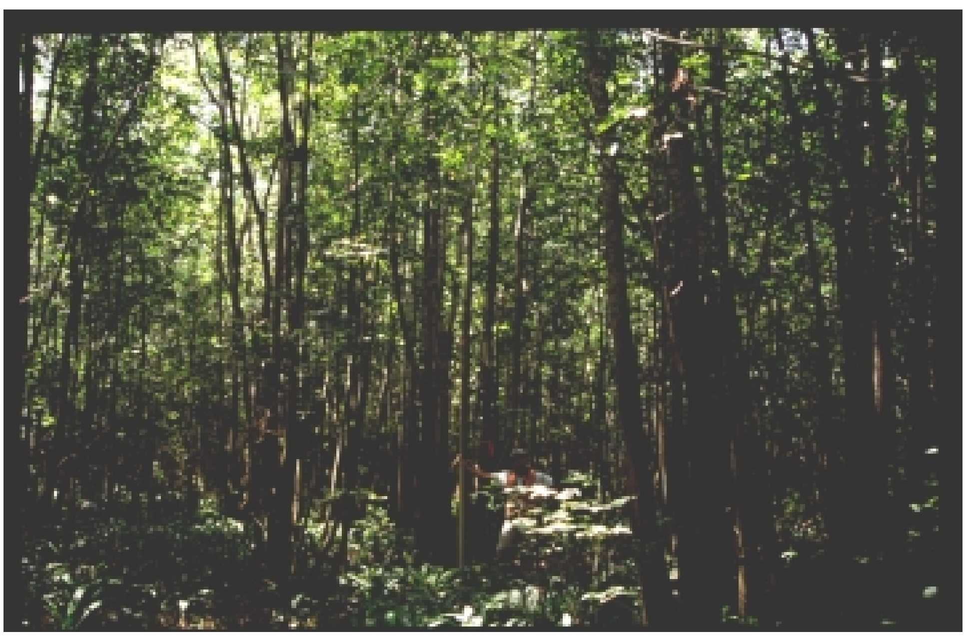


FIGURE 30. HELI treatment 12 years after harvesting disturbance in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

FIGURE 31. Stems in the SKID treatment 12 years after harvesting disturbance in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

TOTAL (1987-1998)

The ANOVA procedure detects significant differences only within the harvest treatments (TABLE 2). Over the twelve-year range of the study, the REF area accumulated the least amount of sediment with a mean of 10.2 and median of 9.5 cm (TABLE 2). The parametric procedure found the mean of 24.7 cm accreted by the GLYPH significantly greater than the means of the HELI and SKID, 20.3 and 16.3 cm, respectively (TABLE 2). However, nonparametric test found the median SKID accretion value of 15.1 cm significantly less than the HELI and GLYPH, 18.3 and 19.5 cm, respectively (TABLE 2). Since the distribution of sediment accretion data throughout the course of this study has been non-Gaussian in appearance, it is safe to assume that distribution of the total values departs form the normal as well, again suggesting the nonparametric statistical procedures better protect alpha. The noteworthy result of the nonparametric analysis is that the sediment accretion value of the SKID treatment is significantly less than the HELI and GLYPH, and furthermore that Mann-Whitney test failed to detect a significant difference between those two treatments (TABLE 2). A question is then drawn addressing why the SKID treatment failed to collect as much sediment over the twelve-year span when compared to the HELI and GLYPH. One possible explanation may be soil disturbance that occurred to the site as a result of skidder traffic. Ruts may have been formed that were subject to erosive waters and raindrop impact. In

FIGURES 19, 20 and 24 negative values of sediment accretion are visible, yet in FIGURE 28, negative values are associated with all treatments and REF area.

In 1998 data was collected on basal area (ft^2/acre) [BA] for the treatment plots. When average BA for each treatment plot is plotted against the average sediment accretion (1987-1998) for its corresponding plot in FIGURE 32 a direct linear relationship is difficult to observe, though a Pearson's Correlation Coefficient of (-0.426) is obtained when comparing the two variables. This value suggests a negative, though not strong, relationship between BA and total sediment accretion in the treatment plots. Though this relationship is not as strong as the correlation found between average overstory biomass and sediment accretion for the years between 1988 and 1993 (FIGURE 23), it is still apparent and complementary. The relationship suggests that greater amounts of BA are associated with less sediment accretion. Greater amounts of BA are associated with greater tree-diameter area, suggesting the presence of larger trees or several small trees. Again, when there are greater numbers of trees on a given plot, light is intercepted by the forest canopy with little remaining for low growing herbaceous species, which are important in sediment capture.

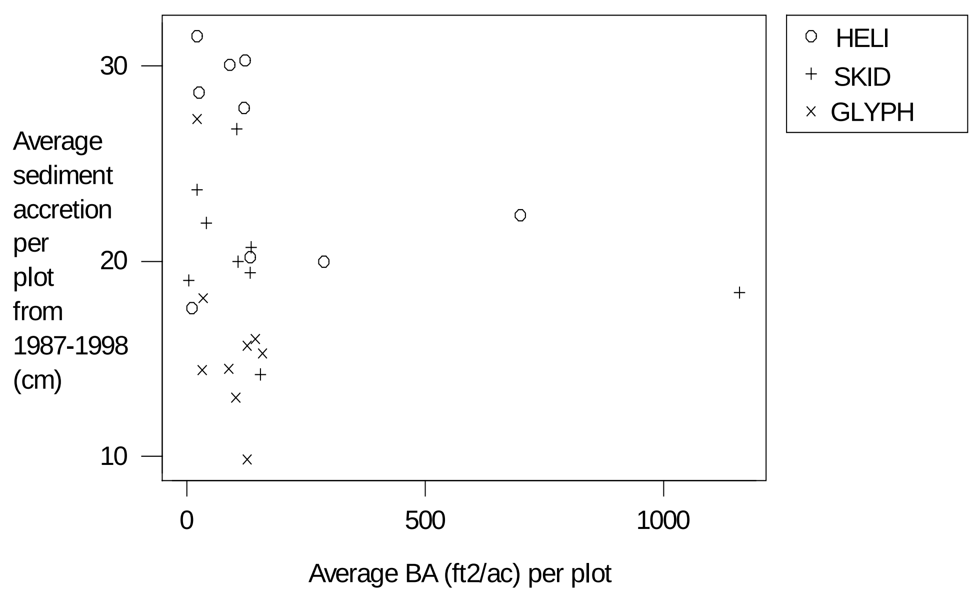


FIGURE 32. Effects of harvesting treatment on BA (ft^2/ac) and total sediment accretion (cm) between 1988 and 1998 in a tupelo bald-cypress wetland in the Mobile-Tensaw River Delta in southwest Alabama.

In 1998 the metal rods used to measure sediment accretion and loss were measured to determine the depth at which rust is found on the rod. The depth of iron oxidation yields a general idea of where the water table is found within the treatment plots. The results of these measurements are in TABLE 7.

TABLE 7. Depth to iron oxidation on sediment measurement rods [rust(cm)] and total sediment accretion (cm) between 1987 and 1998 following three harvesting treatments in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta of southwest Alabama.

Treatment	HELI	SKID	GLYPH	REF
Mean depth to rust (cm)	44.2 b[1]	43.8 b	42.0 b	24.0 a
(STD.DEV)	(13.2)	(13.7)	(12.2)	(9.7)
Mean total sediment accretion (cm)	20.3 b[1]	16.3 b	24.7 c	10.2 a
(STD.DEV.)	(10.1)	(12.5)	(12.8)	(6.3)
Median depth to rust (cm)	40.0 b[2]	40.0 b	38.3 b	24.0 a
(95% CI on true median)	(+/-4.9)	(+/-5.1)	(+/-4.5)	(+/4.5)
Median total sediment accretion (cm)	18.3 c[2]	15.1 b	19.5 c	9.5 a
(95% CI on true median)	(+/-2.0)	(+/-3.4)	(+/-3.1)	(+/-1.0)

[1] For this row, values with different letters indicate a significantly difference based on t-tests at and alpha level of 0.05.
[2] For this row, values with different letters indicate a significant difference based on Mann-Whitney tests at an alpha level of 0.05.

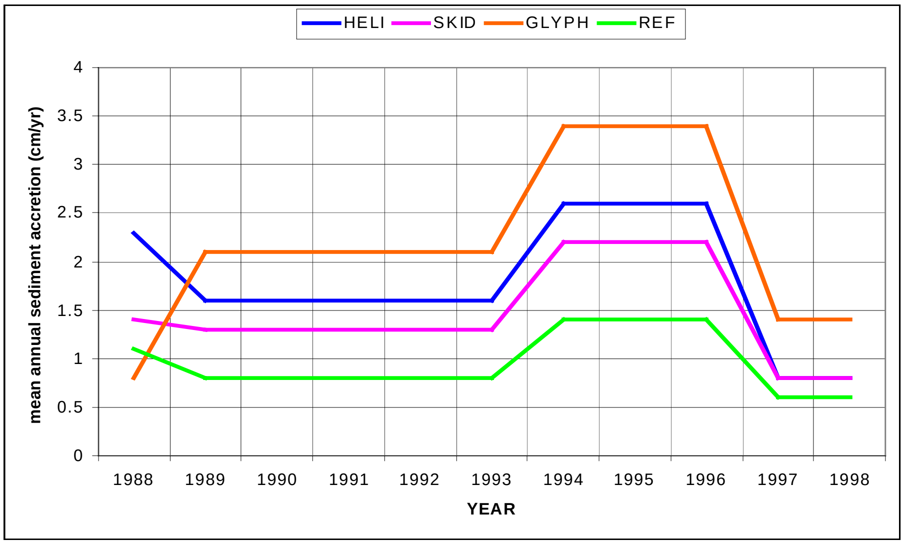
Both analyses find the location of rust on the metal rods significantly deeper in the harvest treatments compared to the REF area. In addition, a weak positive correlation (Pearson Correlation Coefficient of 0.248), is detected between the two variables. This correlation, though not very strong, suggests that those areas with a deeper depth to rust accrete more sediment than plots with shallower rust location. Furthermore, in analyses of total sediment accretion over the 12-year span of the study, both statistical procedures find the treatment areas accreting significantly more sediment than the REF area. Assuming that the water table does not fluctuate dramatically over the experiment area, deeper depths to rust may be associated with the greater amounts of sediment accretion in the treatment plots, suggesting changes in the microtopography, specifically increased elevation.

Temporal Patterns of Annual Sediment Accretion for each Harvest Treatment

Reference

The reference area is the basis by which to compare the three harvest treatments over the 12-year spans of the study. A wetland’s ability to capture sediments changes due to many variables. Within the harvest treatment area, significant differences in sediment accretion are associated with change in vegetation and soil disturbance. However, when examining annual sediment accretion over the span of the study it cannot be assumed that variables, such as upstream land use and weather remain unchanged. Effects of external variables are accounted for in the REF plot measurements. Therefore, significant change in annual sediment accretion in REF area does not necessarily suggest sampling error.

FIGURE 33 and TABLE 8 display a range of 0.6 to 1.3 cm/year of sediment accretion in the REF area across the span of the study. These numbers fit into the range of published sediment accretion rates in various wetlands (TABLE 3). Both statistical procedures detect similar significant changes. Results for location shift yield a significant decrease in mean and median annual sediment accretion between 1988 (mean 1.1, median 1.0 cm/yr) and 1993 (mean 0.8, median 0.7 cm/yr). Between 1993 and 1996 (mean1.4, median 1.3 cm/yr) there was a significant increase in annual sediment accretion. Finally, between 1996 and 1998 (mean and median 0.6 cm/yr) annual sediment accretion significantly increased.



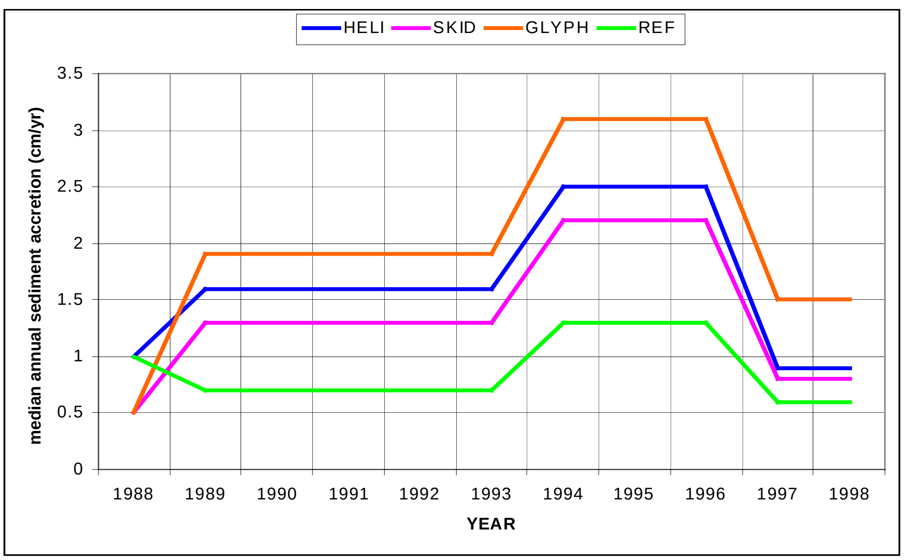


FIGURE 33. Mean and median annual sediment accretion (cm/year) in three harvest treatments and reference plot over 12-year course of study in a tupelo-baldcypess wetland in the Mobile-Tensaw River Delta of southwest Alabama.

TABLE 8. Annual sediment accretion in three harvest treatments and REF area between 1988 and 1998 in a tupelo-baldcypress wetland in the Mobile-Tensaw River Delta in Southwest Alabama.

		1987-1988 cm/year	1988-1993 cm/year	1993-1996 cm/year	1996-1998 cm/year
REF	MEAN	1.1 a[1]	0.8 b	1.4 c	0.6 d
	(STD.DEV.)	**(1.2)**	**(0.7)**	**(1.0)**	**(1.1)**
	MEDIAN	1.0 a[2]	0.7 b	1.0 c	0.6 d
	(95%CI on true median)	**(+/-0.5)**	**(+/-0.1)**	**(+/-0.1)**	**(+/-0.1)**
HELI	MEAN	2.2 a[1]	1.6 a	2.6 b	0.8 c
	(STD.DEV.)	**(5.07)**	**(0.8)**	**(1.6)**	**(1.2)**
	MEDIAN	1.0 a[2]	1.6 a	2.5 b	0.9 c
	(95%CI on true median)	**(+/-0.5)**	**(+/-0.2)**	**(+/-0.3)**	**(+/-0.2)**
SKID	MEAN	1.4 a[1]	1.3 a	2.2 b	0.8 c
	(STD.DEV.)	**(5.3)**	**(0.9)**	**(2.0)**	**(0.9)**
	MEDIAN	0.5 a[2]	1.3 b	2.2 c	0.8 d
	(95%CI on true median)	**(+0.5)**	**(+/-0.4)**	**(+/-0.5)**	**(+/-0.2)**
GLYPH	MEAN	0.8 a[1]	2.1 b	3.4 c	1.4 d
	(STD.DEV.)	**(3.0)**	**(1.2)**	**(2.0)**	**(1.0)**
	MEDIAN	0.5 a[2]	1.9 b	3.1 c	1.5 d
	(95%CI on true median)	**(+/-0.5)**	**(+/-0.3)**	**(+/-0.5)**	**(+/-0.3)**

[1] For this row, values with different letters indicate a significant difference of treatment means by paired t-tests at an alpha of 0.05. Comparisons were only made year-by-year; therefore interpretations of results based on comparing non-sequenced years are not statistically valid.
[2] For this row, values with different letters indicate a significant difference of treatment medians by Wilcoxon Signed Rank t-tests at an alpha of 0.05. Comparisons were only made year-by-year; therefore interpretations of results based on non-sequenced years are not statistically valid.

Helicopter

Over the twelve-year course of this study, the mean and median sedimentation rates in the HELI treatment exceed the REF area (FIGURE 33, TABLE 8). The greatest value of annual sediment accretion is 2.6 cm/yr, which occurs as the mean value for 1996, roughly ten years after initial harvest disturbance (TABLE 8). Similar to the REF plot, there appears to be an oscillating patter of decrease-increase-decrease in annual sediment accretion values over the span of the study. However, tests for location shift failed to detect a significant difference in mean and median values of annual sediment accretion between 1988 and 1993. Since a significant decrease is found in the REF area between 1988 and 1993, the lack of a difference in the HELI treatment suggest an improvement in the treatment area’s ability to capture and trap sediment. Increased sediment trapping ability is associated with thick ground flora that dominates a freshly cleared area.

Nonetheless, after 1993 the pattern of annual sediment accretion, both means and medians, follow the same pattern of increase and decrease found in the REF area. This suggests that increase and decrease in annual sediment accretion may be largely attributable to changes outside the control of the study such as weather and change in patterns of upstream land use.

Between 1997 and 1998 there was a very strong El Nino event (NOAA, 2001) (TABLE 9). El Nino produces conditions that increase rainfall in the lower reaches of the Mobile-Tensaw River Delta's watershed while a decrease in rainfall is witnessed in the upper reaches of southeastern United States. La Nina results in increased rainfall for the entire watershed (NOAA, 2001). One reason the sedimentation rates decreased may be the El Nino resulted in less rainfall in the upper portion of the watershed, thereby reducing the sediment load and flood intensity.

In TABLE 9 a pattern can be seen in the sediment accretion and weather event. There was a rapid succession of El Nino events between 1990 and 1994 (NOAA, 2001). In 1995 La Nina occurred, during the time of La Nina, rainfall increases over the range of the watershed. Between 1994 and 1996 a significant increase in annual sediment accretion is witnessed not only in the HELI treatment, but also in all treatments and the REF area (FIGURE 33). During the strong El Nino of 1997 to 1998, annual sediment accretion significantly dropped in all treatments. Data regarding the number of flood days witnessed at the Barry Steam Plant operated by the USACE is included at the bottom of TABLE 9 (USACE, 2001). The Barry Steam plant is located approximately seven miles from the study sites so flood data witnessed at its stream gage is indicative of conditions at the study site. The annual numbers of flood events recorded at the stream plant do not appear to correspond with the presence of El Nino or La Nina. One may expect more flooding events during La Nina since it is associated with increased rainfall though out the entire watershed of the Mobile River, yet the relationship between the annual number of flood days and the weather event is unclear. In fact, the years with the greatest amount of flood days, 1992 (291 flood days) and 1997 (256 flood days), both are associated with El Nino (TABLE 10). This situation, however, may not be entirely peculiar since El Nino, though associated with less precipitation in the upper portions of the Mobile River watershed, is also associated with increased precipitation in the lower, primarily delta portions of the watershed (NOAA, 2001). It seems logical that greater amounts of sediment would enter the water system in the upper portions of the watershed since this region contains mountainous terrain of the western Appalachians and agricultural and urban land of the Southeastern United States. This idea coincides with the data in TABLE 9, suggesting that when rainfall increases in the upper reaches of the watershed during La Nina, more sediment enters the stream and results in greater sediment accumulation in the study site.

TABLE 9. Median annual sediment accretion values (cm/yr) in the three treatment and REF area from 1987 to 1998, presence of El Nino and La Nina weather events (NOAA, 2001), and annual number of flood days recorded at the Barry Steam Plant (USACE, 2001).

Year											
1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998
HELI			**Median Annual Sediment Accretion (cm/year)**								
1.0	1.0	1.6	1.6	1.6	1.6	1.6	2.5	2.5	2.5	0.9	0.9
SKID											
0.5	0.5	1.3	1.3	1.3	1.3	1.3	2.2	2.2	2.2	0.8	0.8
GLYPH											
0.5	0.5	1.9	1.9	1.9	1.9	1.9	3.1	3.1	3.1	1.5	1.5
REF											
1.0	1.0	0.7	0.7	0.7	0.7	0.7	1.3	1.3	1.3	0.6	0.6
Event											
El Nino	La Nina ---→			El Nino----------------→				La Nina		El Nino -----→	
Number of floods days/year at Barry Steam Plant											
187	*71*	*237*	*231*	*237*	*291*	*194*	*195*	*185*	*232*	*256*	*211*

Skidder

The temporal pattern of sedimentation in the SKID treatment is similar to the HELI treatment (FIGURE 33). The greatest value of annual sediment accretion, 2.2 cm/yr, is found in 1996 roughly ten years after initial harvest disturbance. The remaining values of annual sediment accretion, ranging from 0.8 to 2.2 cm/yr, fall within the published range for wetlands (TABLE 3).

The parametric analysis fails to detect a significant change in mean annual sediment accretion between 1988 (1.4 cm/yr) and 1993 (1.3 cm/ yr), though the parametric procedure found a significant increase in the median values of 0.5 and 1.3 cm/yr, respectively. Again, between these years a significant decrease was detected in the REF area, suggesting an increase in the SKID treatment's ability to capture and trap sediment during this five-year period. During this period the SKID treatment is marked by thick herbaceous vegetation and stump sprouts which aid in slowing floodwaters and allowing suspended sediments to precipitate from solution. (FIGURES 16 and 21)

Except for the mean and median values between 1988 and 1993, the remaining values of mean and median annual sediment accretion follow the same oscillating pattern of increase and decrease present in the REF area, though the values of annual sediment accretion are higher in the SKID treatment. The higher values of annual sediment accretion in the SKID treatment compared to the REF area are associated with thick herbaceous ground flora, while the similar pattern of increase and decrease in annual sediment accretion is associated with factors such as land use change and weather patterns which are outside the control of the experiment.

Glyphosphate

The mean and median annual sediment accretion values in the GLYPH treatment follow a similar pattern of oscillation seen in the other treatments and REF area (FIGURE 33). The mean annual sediment accretion value of 3.4 cm/year in 1996 exceeds the published range for wetlands, however, the remaining values fall within the published range (TABLE 3).

Mean and median annual sediment accretion values, 0.8 and 0.5 cm/yr respectively, are low in the first two years following harvest disturbance compared to the latter years (TABLE 8). In 1988 the mean and median annual sediment accretion was less in the GLYPH treatment than the REF plot, though both statistical procedures fail to detect a significant difference (TABLE 2). Although the low value in the GLYPH plots is not significantly less than the values recorded in the other treatments and REF area, the relatively small value of annual sediment accretion may have been an influence of some other factor such as erosion. Since glyphosphate kills herbaceous cover and tree roots, both trees and ground must be re-established by seed. The low annual sediment accretion may be in indication of near barren land subject to erosion, and in FIGURE 19 some negative values of sediment accretion are present. However, while it is evident that erosion is present in the first few years following harvest, its effect is not found to be significant compared to the remaining treatments and REF area (TABLE 2).

Soon after the low sedimentation rates are experienced, the annual sediment accretion values increase dramatically until 1996, where they again fall (FIGURE 33). Between 1988 and 1993 there is a significant increase in mean and median annual sediment accretion (TABLE 8), mean values rise from 0.8 to 2.1 cm/year and median values rise from 0.5 to 1.9 cm/year. During these years a significant decrease was detected in annual sediment accretion in the REF area, suggesting a dramatic increase in the GLYPH treatment's ability to capture and trap sediment. The increase in sediment capture between 1988 and 1993, is associated with the thick marsh-like vegetation found within the treatment area. Again, the marsh-like vegetation that resulted as a complete loss of trees and roots, slowed floodwaters and allowed suspended sediment particles to precipitate from water.

A significant increase in annual sediment accretion is also experienced between 1993 and 1996, though this increase is witnessed in all treatments and the REF area. Similar to the HELI and SKID treatments, the highest value of annual sediment accretion, a mean of 3.4 and median of 3.1 cm/yr, is found in 1996, roughly ten years after harvest disturbance. This may suggest that the vegetation is at a physical form that promotes the greatest capture and trapping of suspended sediments in floodwaters. While this may seem reasonable by looking at the thick herbaceous cover in FIGURE 25, the greatest values of annual sediment accretion are found in all treatments and REF area during this period, suggesting that explanation may also lie outside of control of the experiment, perhaps in changing land use and weather patterns. Again, the weather event La Nina was present in 1995, which may have resulted in increased sediment loads in water draining through the watershed of the Mobile-Tensaw River Delta (TABLE 9).

Between 1996 and 1998 a significant decrease is found in mean and median annual sediment accretion values in the GLYPH treatment, coinciding with the significant decrease in annual sediment accretion values in the REF and other harvest treatments (TABLE 8). The photos in FIGURES 25 and 29 show the succession of trees into the marsh-like vegetation and suggest a loss in the treatment area's ability to trap and capture sediment based solely on the physical characteristics of the vegetation. However, since the decrease is detected in all harvest treatments and the REF area, it is possible that influences outside the control of the study, such as the presence of El Nino, are being witnessed as well (TABLE 9).

Conclusions

Wetlands are important components of the environment. The functions of wetlands are complex and often difficult to study, however attempts to study wetland functions aid in the furthering ability to understand the wetland ecosystem. Furthermore, with acquired knowledge, managers are able to test scenarios in investigation of possible management decisions. For instance, urban sprawl may ensue a high demand for timber in the Southern US and also be coupled with increased sediment loads from upstream land use change within the watershed of the Alabama and Tombigbee Rivers. Managers must be able to make decisions that will both supply and support the economy, while at the same time protecting the environment.

Improvement of water quality through sediment removal may be the most widely touted function and value of wetlands. Sediments in water may carry toxic substances, such as high amounts of phosphorus. In addition, suspended sediment also affects water temperature and dissolved oxygen levels. Removal of sediments may be considered the first step in improved water quality.

This study analyses the effects of three harvest treatments on sediment accretion for twelve years following harvest disturbance in a southwestern Alabama tupelo-baldcypress swamp. In the first two years following harvest, the harvested plots yield values of sediment accretion statistically similar to those recorded in the REF area. This suggests that when compared to the undisturbed REF forested wetland, there is no significant change on the wetland's ability to trap sediment during the "fragile period" shortly after harvest when the ground is exposed to the erosive forces of rainfall and floodwaters. However, it must be emphasized that the study area has less than a 1% slope and large quantities of logging slash provided some cover for the site.

After the second year following disturbance harvesting, has a positive effect on sediment capture compared to the REF forest. In 1993, seven years following initial disturbance, the HELI and SKID treatment yield average and median sediment accretion values significantly greater than the REF area, in addition the GLYPH treatment yields values significantly greater than the HELI and SKID treatments. The greater values of sediment accretion recorded in the treatment plots are associated with the presence herbaceous ground flora, which slows flood waters and allows suspended sediment particles to precipitate from overbank flow. Thick herbaceous vegetation is present in areas of the wetland where competition for light with trees is minimal. Harvesting, of course, removes the canopy and allows light to reach ground level, promoting growth of herbaceous vegetation. Furthermore, the application of glyphosphate kills woody vegetation and prevents the formation of stump sprouts, thereby retarding early woody re-growth and maintaining the GLYPH treatment in a marsh-like physical form.

At ten years of recovery the greatest values of annual sediment accretion are recorded. However, the relatively high values of sediment accretion may also be associated with factors outside the control of the study, such as those which effect sediment loads in floodwaters that enter the experiment area. Similar to the results after seven years of recovery, after an additional three years of harvest recovery, the treatment areas continue to yield sediment accretion values significantly greater than the REF forest. Furthermore, the marsh-like vegetation of the GLYPH treatment continues assist in the capture and trapping of sediment as it continues to yield mean and median values of sediment accretion significantly greater than the HELI and SKID plots.

In the twelfth year following harvesting disturbance the treatments continue to yield significantly greater values of sediment accretion compared to the REF forest (Note that the parametric analyses failed to detect a significant difference between the HELI, SKID and REF, but violation of the Gaussian distribution assumption suggests that the nonparametric results better protect alpha of 0.05). In addition, the GLYPH treatment continues to yield sediment accretion values significantly greater than the HELI and SKID treatments. By this time woody vegetation in the HELI and SKID treatments has grown from shrubby stump sprouts into tree form and the look of the vegetation in the treatments appears more like a forest with a distinguishable canopy and understory, though this characteristic is certainly more clear in the REF forest. Woody stems have appeared in the GLYPH treatment, though a separate overstory is not discernable.

Base patterns of annual sediment accretion over the course of the study were established in the REF forest. This pattern consists of a significant decrease in annual sediment accretion between 1988 and 1993, followed by an increase between 1993 and 1996, and finally another decrease between 1996 and 1998. These patterns, established by the REF area are mirrored in the treatment plots. However, between 1988 and 1993 analyses suggest a marked improvement in the treatment plots ability to capture and trap sediment compared to the REF forest. In addition, the annual sediment accretion rates appear to follow a similar pattern to El Nino and La Nina weather events. Since the amount of rainfall influences the intensity of flood events and also affects sediment loads, it is plausible that weather patterns as well as treatment effects are being seen in the results of this study.

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Teaching Assistant, Department of Forestry, Virginia Tech, Blacksburg, VA, Spring 2000

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