ABSTRACT

Title: AGRICULTURAL DRAINAGE DITCHES: SOILS AND IMPLICATIONS FOR PHOSPHORUS TRANSPORT AND RETENTION.

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Agricultural drainage ditches act as pathways for nutrients to local surface waters. Knowledge of ditch materials, the spatial variation and distribution of ditch soil phosphorus, is critical to effective ditch nutrient management strategies. Ditch materials from the University of Maryland Eastern Shore Research Farm in Princess Anne, Maryland were described and characterized using a pedological approach. The spatial variation of phosphorus was also investigated. The materials found within these ditches are natural soil bodies. Pedogenic processes operating in these soils include organic matter accumulation, structure formation, Fe oxidation and reduction, sulfuricization, sulfidization, and bioturbation. Soil phosphorus was well autocorrelated, and exhibited a high degree of spatial variation. Ditch soil phosphorus at depth ranged from 4 to 4882 mg kg^{-1} for total phosphorus, 4 to 4631 mg kg^{-1} for oxalate-extractable phosphorus, and 2 to 401 mg kg^{-1} for Mehlich-3 phosphorus. Future ditch management strategies should include a subsurface soils component.

AGIRCULTURAL DRAINAGE DITCHES: SOILS AND IMPLICATIONS FOR NUTRIENT TRANSPORT.

By

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Dedication

This thesis is dedicated to many people. First, I would like to thank God, for listening, when everyone else got tired. To my family, who have supported me along the way, even though they still don't understand what exactly I do. Kate and Charles, thanks for giving me a place to stay, and endless amounts of entertainment. Thanks to the past and present members of the SAWGAL lab. David, Mike, Mitch, Olivia, and Sumathi, thank you all for being yourselves, and making life in the lab fun. To Steve Burch in the UMD Pedology lab, Lou Saporito at ARS, and all others who helped me with my lab work, thanks so much for your help, I would still be running samples without you. Sue Dorcey, thank you for putting up with all of my late travel requests. I will miss you Sue! Thanks to all my undergraduate assistants for your help. To all the soils graduate students, thanks for your support and ideas.

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“The important thing is not to stop questioning.”
Albert Einstein (1879 - 1955)

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Chapter 1: Introduction

In 2000, the USEPA identified approximately 22,000 surface water bodies as impaired (USEPA, 2003). Eutrophication has caused widespread water quality impairment throughout the U.S., and agriculture has been the greatest source of these nutrients (USEPA, 1996). Nitrogen (N) and phosphorus (P), nutrients essential for agriculture production, are leading factors controlling the eutrophication of freshwater aquatic systems (Diaz and Rosenburg, 1995; Carpenter et al., 1998; USGS, 1999). In nearly all freshwater aquatic environments, P is the most limiting nutrient (Boers et al., 1998; Correll, 1998). The Chesapeake Bay is the largest estuary in the United States, but has suffered from the effects of N and P limited eutrophication for more than 50 years (Boesch et al., 2001). In addition, the Bay has been the subject of one of the most intensive large-scale ecosystem restoration projects in the world.

The Delmarva Peninsula, on the eastern shore of the Chesapeake Bay, is home to one of the most intense poultry broiler operations in the U.S., producing over 600 million broiler chickens annually (Sims et al., 2003). The substantial amounts of animal wastes produced by Delmarva's intense broiler operations and the use of commercial fertilizers has led to excessive P in soils on the Delmarva (Sims et al., 1996; Sims et al., 1998; Sims et al., 2003). Agriculture is responsible for 80% of the nonpoint source P load to the Chesapeake Bay (Taylor and Pionke, 2000). Greater than 90% of the agricultural P entering the Chesapeake Bay is the result of intensive agricultural practices such as broiler chicken production and cropland (Taylor and Pionke, 2000).

The lower Delmarva Peninsula contains an extensive network of agricultural drainage ditches used to lower the water table and speed the removal of overland flow from the poorly drained soils of the region. Most agricultural drainage ditches on the lower Delmarva Peninsula drain into streams and rivers that flow into the Chesapeake Bay and Delaware Bays. These drainage ditches have been shown to serve as key pathways for agricultural nutrient export (Sims et al., 1998). To date, most research on managing nutrients in drainage ditches has been focused on N. Mechanisms of P transport in drainage ditches are poorly understood.

Efforts to control agricultural nonpoint P into the Chesapeake Bay have focused on the implementation of both structural (e.g., animal waste lagoons and sheds) and agronomic (e.g., no-till, nutrient management plans) best management practices (BMPs) to control sediment-bound P inputs into the Bay (Boesch et al., 2001). However, some BMP strategies are not adequate by themselves. For example, the largely accepted and promoted BMP of no-till farming on the lower Delmarva has been shown to increase dissolved P in surface runoff and soil P in the upper-most horizons (Boesch et al., 2001). While long-term water quality improvement strategies in the watershed must include overall reductions in P inputs to the Delmarva, more effective BMPs need to be developed if nutrient reduction goals for the Chesapeake Bay are to be met (Boesch et al., 2001). The establishment of BMPs for agricultural drainage ditches have been overlooked in the past, yet they may provide additional nutrient reductions, especially for P.

Given the dearth of information on ditch P dynamics and management, a study to investigate the potential for the management of agricultural drainage ditches to mitigate nutrient transport out of a watershed was initiated in 2002 by researchers from multiple university, federal, and state agencies, to identify critical source areas within drainage ditch networks that show the greatest potential for the release of P to overlying waters. This thesis lays the foundation for critical source area detection through the systematic investigation of drainage ditch soil properties such as particle size distribution, Fe and Al contents, and their relationship to P release to overlying waters.

Specific objectives of my research were to (1) characterize soils within agricultural drainage ditches on the University of Maryland-Eastern Shore Farm, (2) examine the spatial distribution of surficial soil P within a drainage ditch network, and (3) conduct an intensive survey of near-ditch and research farm soils.

Chapter 2: Background and Literature Review

Eutrophication and the Chesapeake Bay

Accelerated eutrophication, the biological enrichment of surface waters arising from anthropogenic nutrient inputs, has been identified in the U.S. as the most widespread water quality impairment, and agriculture as the largest source of these nutrients (USEPA, 1996). In 2000, 11% of the 22,000 surface waters identified by the USEPA as impaired were the result of nutrients (USEPA, 2003). Nitrogen (N) and phosphorus (P), nutrients essential for agriculture production, are leading factors controlling the eutrophication of freshwater and estuarine aquatic systems (Diaz and Rosenburg, 1995; USGS, 1999; Carpenter et al., 1998). Due to the ecological, economic, and social impacts of eutrophication, there are ever-increasing concerns and efforts to control eutrophication in fresh and estuarine waters of the U.S.

The Chesapeake Bay is a highly productive estuary with an approximate watershed size of 167,000 km^2 over 6 states (Maryland, Virginia, Delaware, Pennsylvania, New York, West Virginia) and the District of Columbia, and a surface area of 11,000 km^2. The Bay is the largest estuary in the United States and has great social and economic value to surrounding states (Boesch et al., 2001). However, the Bay has been stricken with the ill effects of eutrophication for nearly 30 years, while the signs of eutrophication have been around for nearly 50 years (Boesch et al., 2001). Eutrophication, increases in diseases, as well as the increasing loss of subaquatic vegetation (SAV) since hurricane Agnes in 1972, has contributed to the sharp decline of shellfish and fishery stocks in the Bay.

More recently, outbreaks of Pfiesteria in the lower Pokomoke River in 1997 and other Bay tributaries lead to stricter nutrient management regulations for farmers within Maryland in the form of the Maryland Water Quality Improvement Act of 1998 (McCoy, 1999). The Maryland Water Quality Improvement Act mandates nutrient management plans for both N and P for agricultural producers with at least $2500 annual gross revenue or eight animal units (1 animal unit = 454 kg live weight) (Coale et al., 2002). In addition, the Maryland Phosphorus Site Index (PSI), a variation of Lemunyon and Gilberts (1993) original P index, was developed for the identification of sites with the greatest potential for P loss to surface waters (Coale et al., 2002). The Maryland Water Quality Improvement Act of 1998 set rigorous and ambitious goals for the reduction of agricultural nutrient inputs into the Bay; while setting a high standard for the rest of the country. Approximately 1.26 million acres in Maryland have been placed under nutrient management plans by 2001 (Maryland Cooperative Extension, 2005).

Intensive Animal Production on the Delmarva Peninsula

Several regions within the Chesapeake Bay watershed contain high amounts of intensive agricultural animal production. In states such as Pennsylvania and New York there is a very active dairy cow industry while the Delmarva Peninsula contains one of the most intensive poultry broiler industries in the United States. As a result, the lower Susquehanna and the lower Eastern Shore have been identified with the highest potential for P loss in the watershed (Taylor and Pionke, 2000).

In 2004, the Delmarva poultry industry produced more than 560 million broiler birds and 1,360 million kg of chicken (Delmarva Poultry Industry, 2005). Somerset

County, which is located on the lower Eastern Shore of Maryland, ranked 41st in broiler production among all U.S. counties (Delmarva Poultry Industry, 2005). As a result of high bird production, enormous quantities of poultry litter (comprised of both animal manure and bedding material) are produced. In general, most poultry litter has a low N to P ratio, so that when applied to meet crop N demands, leads to P applied in excess of crop demands (McCoy, 1999). P applied in excess of crop requirements may lead to excess soil P, which can enrich runoff and lead to eutrophication (Sharpley, 1999).

Agricultural Drainage Ditches

Agricultural drainage ditches are linear features that serve many practical and economic functions for agriculture (Janse and Van Puijenbroek, 1998). Ditches are most often located in environments where the landscape is predominantly flat and the ground water table is near or at the surface for extended periods of the year. In addition, drainage ditches are also utilized in landscapes where a water-restricting layer is close to the surface, such as the Atlantic Coastal Plain of North Carolina, preventing the downward movement of water in the soil profile. Drainage ditches serve as a method to lower the water table, and remove surface runoff in these environments. Intercepting the water table allows water to be moved quickly to local surface waters such as streams and rivers. During intense or prolonged periods of rainfall, open air drainage ditches speed the removal of excess runoff from fields.

The removal of both ground water and runoff water aids in avoiding saturated surface soil conditions and provides an optimal seed bed during the growing season. Drainage also allows farmers to enter their fields with heavy machinery earlier in the growing season. By controlling the water table and moisture status in cultivated fields, the farmer gains some control of an unpredictable natural system. Agriculture in regions with poorly drained soils would be economically impossible without land drainage. Therefore, management of the water table in agricultural production imparts an economically important role to drainage ditches in agriculture.

Artificial land drainage can be accomplished through several techniques. Two common methods in the U.S. are the installation of open-air drainage ditches and tile drains (Evans et al., 1995). Historically, open-air drainage ditches were constructed using manual labor, often slave labor in the case of the Eastern U.S., by horse-drawn digging devices, and more recently with gasoline-powered machinery (Shirmohammadi et al., 1995). Installation of tile drains in agricultural fields to intercept high water tables is another common method in the United States. While this system does not remove surface runoff, it does lower the ground water table so that the volume of surface runoff is reduced due to the increased infiltration capacity of soils with a lower ground water table (Simard et al., 2000). In tile drainage systems, plastic, clay, or concrete tiles are placed in the soil deep enough to intercept the water table, and draw it down just enough to permit crop growth. Tile drains lines often feed directly into open air drainage ditches that remove the drainage water to other surface waters or directly into streams (Evans et al., 1995).

Pedogenic Processes in Drainage Ditch Environments

Recent investigations of mineral and organic materials found within drainage ditches have described these materials as sediments and not soils (Sallade and Sims, 1997a,b; Nguyen and Sukias, 2002). Drainage ditches are apt to function similarly to wetlands (Bowmer et al., 1994; Nguyen and Sukias, 2002). Processes affecting drainage ditch soil formation and pedogenesis are analogous to those in wetland environments. Therefore, a pedological investigation of soils formed in agricultural drainage ditches may yield further insight to nutrient transport processes.

Simonson (1959) described four categories of soil formation processes as a framework for understanding soil development. The categories are: additions, removals, transfers, and transformations. These processes occur in all soils.

Additions of mineral and organic materials to drainage ditches may result from both overland flow and sidewall slumping processes (Nguyen and Sukias, 2002). During periods of intense precipitation, surface runoff including entrained sediments may flow into drainage ditches (Sallade and Sims, 1997a). Mineral and organic materials (coarse and dissolved) may settle out and accumulate over time with successive erosional events. In subaqueous soil environments, the organic component provides the most robust evidence for pedogenic additions (Demas and Rabenhorst, 1999). Further additions to drainage ditch soils may include such materials as decaying animal tissues, animal feces, algal biomass, and dissolved ions as Fe^{2+} and SO_4^{2-} from surrounding native soils (McCoy et al. 1999; Nguyen and Sukias, 2002).

Figure 1. Algae growing on the water surface in drainage ditch DX2 at University of Maryland Eastern Shore Research Farm in July 2003.

Soil losses and removals are also likely to occur in drainage ditch soils. These processes are modified by hydrological factors such as flow duration, frequency, and velocity. During storm events where surface runoff processes are most likely to occur, an increase in drainage flow volume and velocity are expected to take place. Increasing flow velocity promotes scouring, which removes and relocates sediments within the drainage ditch network (Sims et al. 1998). Suspended sediment may be completely removed from the drainage ditch network and carried into local streams (Sims et al. 1998).

Examples of transfers within soils include diffusion processes, bioturbation from insects or animals and eluviation (Fanning and Fanning, 1989). Within drainage ditches, diffusion may be an important process. For instance, the diffusion of soluble Fe from sub-surface horizons may occur when concentrations near the surface are depleted by overlying drainage waters (McCoy et al., 1999). In addition, the diffusion of SO_4^{2-} from underlying soils that contain high concentration of sulfides may also occur. Bioturbation and transfer of soil material from one horizon to another may also occur due to benthic microorganisms and animals such as insects and macroinvertebrates. Eluviation, primarily the eluviation of clay downward through the soil profile, is not likely a large contributing factor due to the relatively young age of drainage ditch soils.

Figure 2. Grey spots are comprised of sub-surface soil-evidence of biopedoturbation in drainage ditches at the University of Maryland Eastern Shore Research Farm (March 2003.)

Figure 3. Iron concentrations on a soil ped face, evidence of iron translocation, from a soil found within a drainage ditch at the University of Maryland Eastern Shore Research Farm.

Transformations occurring within soils are commonly thought of as changes in the mineral and organic fractions (Demas and Rabenhorst, 1999). Within wetland soils, and by extension, within drainage ditches, the reduction of Fe^{3+} to a more soluble form Fe^{2+} occurs when reducing conditions are present within the soil (Sims et al., 1998; McCoy et al., 1999; Nguyen and Sukias, 2002) Reduction of ferric Fe may lead to other transformations such as the dissolution of Fe-bound P. Phosphorus that is solubilized is believed to be more mobile in soil than P which is occluded by or bound to Fe. Denitrification of nitrate-N to nitrous oxide and N_2 gas is a transformation that also occurs in soils with low redox potentials.

Simonson's four basic soil development categories were found to exist in a subaqueous environment in Sinepuxent Bay, Maryland (Demas and Rabenhorst, 1999). While the relationship of drainage ditch soil formation with wetland soils has previously been discussed here, there are unmistakable similarities between drainage ditch soils and subaqueous soils.

Jenny's (1941) five factors of soil formation: parent material, climate, topography, biotic activity, and time are also helpful in understanding the formation of drainage ditch soils.

Several parent materials can lead to the formation of soil-like materials within agricultural drainage ditches. One source of both mineral and organic matter within drainage ditches is the overland transport of soil particles and organic particles by runoff from agricultural fields into drainage ditches (Sallade and Sims, 1997a; Nguyen and Skias, 2002). The slumping of soil from the sidewalls of drainage ditches after construction or dredging is an additional parent material source for drainage ditches (Sallade and Sims, 1997a; Nguyen and Skias, 2002). The native materials (e.g. fluvial deltaic and marine sediments in the Atlantic Coastal Plain) that underlie all drainage ditches are a further source of parent materials (Sims et al., 1998). In addition, organic material can enter the drainage ditch as debris after cultivation of crops in the fall, or as dead plant tissues from in situ plant material within the drainage ditch (Nguyen and Skias, 2002).

Climate factors, including air temperature, soil temperature regime, and length of growing season, all have significant biogeochemical affects on the development of wetland soils, hence drainage ditches, as well as plants growing in drainage ditches.

On the Atlantic Coastal Plain, the landscape in which drainage ditches are located is relatively flat. Even so, topography both within a drainage ditch and outside of a drainage ditch may have an affect pedogenesis within a drainage ditch mainly through water dynamics. Wetland soils are profoundly affected by water dynamics (Fiedler and Sommer, 2004). Micro-changes in topography may have an effect on the speed and scouring effects of flowing drainage ditch water, thereby affecting the underlying soils. Changes in the relief of the landscape will most certainly bring about changes in water table height and affect runoff amounts into drainage ditches.

Biotic factors such as vegetation and organisms affect the materials and pedogenesis that occur within drainage ditches. Vegetation may have a large impact on suspended sediments by creating turbulence in flowing water promoting sedimentation and the accretion of drainage ditch materials. Organisms such as insects and macroinvertebrates burrow into drainage ditch soils, bringing with them deeper materials to the surface.

Time is a dominate soil formation factor. Human interactions with drainage ditches such as the time between clean-outs may control the depth of drainage ditch materials. The time required for mineral and organic material buildup in the drainage ditches is an essential factor for their development. Organic material additions may be controlled by the frequency of herbicide application for controlling the growth of plants within drainage ditches, thereby limiting their growth and buildup. In general, drainage ditch materials are youthful in nature and very dynamic.

Additional factors of importance in potential drainage ditch soil development are the geometry of the drainage ditches themselves, and the water depth within the drainage ditches. Deeper drainage ditches tend to be wider at their base than shallower drainage ditches and provide more surface area for plant growth and deposition. Depth may also influence the depth of ground water interaction. Drainage ditch slope and width may influence the speed and lateral flow direction of drainage ditch waters impacting the depth of drainage ditch materials through scouring effects and hence, textural composition. Water depth and resonance time may also influence the rates of organic material decay by increasing the chance for creating a reducing environment.

Agricultural Drainage History

The development of land drainage systems began in Mesopotamia around 9,000 years ago (van Schilfgaarde, 1971). Surface-drainage system networks were also developed by the Egyptians and the Greeks around 400 B.C. (Shirmohammadi et al., 1995). The Roman Cato in the second century B.C. wrote about the need for land drainage consisting of many drains in low-lying areas to remove excess water (van Schilfgaarde, 1974). In Europe, drainage also has a long history. In England, the Romans installed drainage networks in 150 A.D., while in the eighth and ninth centuries, the Netherlands began to install extensive ditch networks along with their dike networks in efforts to reclaim land back from the ocean (Shirmohammadi et al., 1995). Around the thirteenth century, Holland began the construction of ditch networks for dealing with excess precipitation (Shirmohammadi et al., 1995). In France, the La Gironde project drained nearly 1,500,000 acres (Shirmohammadi et al., 1995).

Organized drainage for agricultural and public purposes has been around since the 1600's in the U.S. (Evans et al., 1996). Several states rely on drainage ditches to control ground water levels in both agricultural and urban areas. In the humid regions of Oregon and Washington, where precipitation averages up to 1,500 mm per year, surface drainage is essential (Backlund et al., 1995). Drainage networks were first began being installed in 1830, and by 1870 most of the land that could be drained by surface drainage had been drained (Backlund et al., 1995). After 1870, subsurface tile drainage became the preferred method of land drainage due to the desire of not losing land (Backlund et al., 1995). The 1960's brought about newer technologies such as flexible plastic materials for subsurface drainage, and drainage installations began to increase in this region (Backlund et al., 1995).

Many states in the Midwest rely heavily on drainage for agricultural proposes with nearly 37% of arable land requiring drainage (Fausey et al., 1995). Agricultural drainage began in the Cornbelt and Great Lake states, which is comprised of Illinois, Indiana, Iowa, Michigan, Minnesota, Missouri, Ohio, and Wisconsin around 1850 (Fausey et al., 1995). The Swamp Land Acts of 1849 and 1859 made available large tracts of wetlands and swamps, clearing the way for the development of drainage networks thought the region to aid in the conversion of land for agriculture and urban development (Fausey et al., 1995). Early in the 1900's, formal drainage districts began to be formed throughout the regions, which further aided in the development of drainage systems (Fausey et al., 1995). Since then, these states have drained more than 20.6

million hectares using both surface and subsurface drainage systems (Fausey et al., 1995). In Indiana alone, there are over 36,000 miles of drainage ditches (Evans et al., 1996).

The Northeastern U.S. and Southeast Canada, primarily Quebec and Ontario, has used land drainage since colonial times for both public and agricultural purposes. Early colonial and state laws provided authority for drainage projects in Massachusetts and New York, where both open ditches and subsurface drainage techniques were utilized (Ritter et al., 1995). In fact, the first commercial production of clay tiles for subsurface drainage in the U.S. was started in New York with techniques learned in England. New York has the most drainage, with approximately 45% of the 370,600 ha drained lands being open-air surface ditches. Drainage in Ontario and Quebec began in 1912, and since then more than 600,000 ha of land have been drained.

In the Mississippi delta region of the U.S. land drainage is an integral part of production farming in an areas which receives in excess of 1,500 mm of rain each year. However, drainage is not practiced in all areas which could benefit from land drainage due to the high cost and labor requirements of drainage network installations (Bengtson et al., 1995). In Arkansas, Louisiana, and Mississippi, total cropland with drainage exceeds 5 million ha, while more than 2.5 million ha have been identified as needing land drainage (Bengtson et al., 1995).

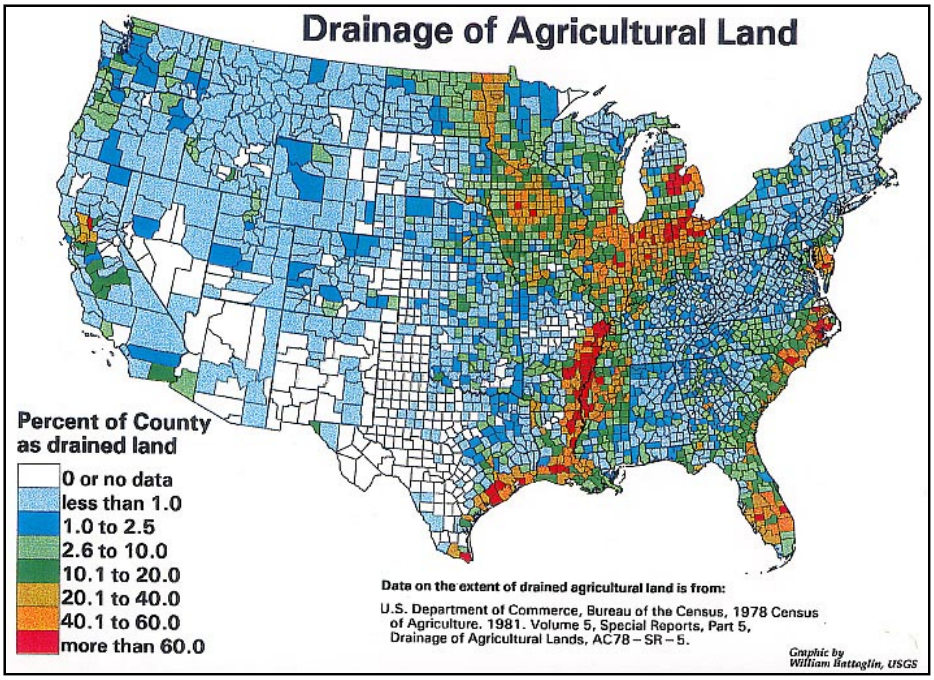
Agricultural drainage in the Southeastern U.S. Atlantic Coastal Plain is very extensive. Many soils in this region are poorly drained and require some form of land drainage for profitable agriculture. Drainage in this region began in North Carolina with the arrival of early colonial settlers. In 1763, the Dismal Swamp Canal Company, formed by George Washington and five associates, bought nearly 17,000 ha with the goal of draining the land. It was on this land where a 7.5 km canal referred to as the "Washington Ditch" was soon constructed (Evans and Skaggs, 2004). In 1805, work on the Dismal Swamp Canal was completed, providing navigation from the Chesapeake Bay to Albermale Sound (Evans and Skaggs, 2004). Following construction of the Dismal Swamp Canal, additional parallel ditches were constructed allowing lands to the east of the canal to become extensively cultivated (Evans and Skaggs, 2004). Today, the coastal plain of North Carolina has more than 2 million acres that rely on drainage ditches (Evans et al., 1996). More than 40% of all crop land in North Carolina today requires artificial drainage (Thomas et al., 1995). Soon after drainage started in North Carolina, land drainage began moving southward with the growing populations to states such as South Carolina and Georgia. The colony of South Carolina in 1754 passed an act for the draining of lands (Shirmohammadi et al. 1995). Today in South Carolina, roughly 710,000 ha of crop land are drained while in Georgia, 625,000 ha of crop land are drained (Thomas et al., 1995). Florida has one of the most extensive agricultural drainage networks in the Southeast, and possibly the U.S. The vast majority of this drainage is in the Florida Everglades Agricultural Areas located just south of Lake Okeechobee. This region is dominated by organic soils underlain by sand and limestone. Drainage began in 1883, while successful drainage for crop production did not occur till around 1900 (Thomas et al., 1995). Currently in Florida, there are more than 2.5 million ha that are affected by artificial land drainage (Thomas et al., 1995).

In New Jersey, Delaware, Maryland, and Virginia, artificial land drainage in the Northern Atlantic Coastal Plain has been around since colonial times and is essential for profitable agriculture. Many early settlers brought with them drainage techniques of the fifteenth through the seventeenth centuries of Europe to the region (Shirmohammadi et al., 1995). New Jersey in 1772 passed one of the first drainage laws in the colonies, and was later added to the state constitution in 1776 (Shirmohammadi et al., 1995). Most drainage ditches through the nineteenth century in this region were constructed using open-air surface ditches as well as a few subsurface drains constructed of local resources (Shirmohammadi et al., 1995). Many of the open-air ditches in the Mid-Atlantic region that were constructed by hand or horse-drawn devices before the nineteenth century are still functional (Shirmohammadi et al., 1995).

Maryland has an extensive history of drainage ditches for both public and private use. Ditching in Maryland began in the 1700's when the first publicly recorded drainage ditch, the Long Marsh ditch, was constructed in 1789 (Public Drainage Task Force, 2000). Many of the early drainage projects were constructed using slave labor (Public Drainage Task Force, 2000). During the Great Depression of the 1930's, drainage ditch construction became more widespread because farmers could not afford to lose a crop to flooding. Efforts by the Civilian Conservation Corps (CCC) lead to the construction of vast networks of agricultural drainage ditches, public drainage ditches, and stream straightening with the goal of job creation and greater control of agricultural production (Public Drainage Task Force, 2000; Sims et al., 1998). These efforts eventually led to the formation of several public and private partnerships for ditch maintenance and further construction.

There are currently 103 Public Drainage Associations (PDA's) and Public Watershed Authorities (PWA) in Maryland. PDAs and PWAs are charged with maintaining larger drainage ditches systems that drain networks of private and state owned drainage ditches in local drainage districts. These organizations were given authority by the state to levee taxes against farm owners and other citizens who benefited from their use as well as an easement on land surrounding the ditch so that maintenance could be performed. In addition, these ditches were constructed in a fashion so that all farms with drainage ditches in the area would drain into these larger ditches. This drainage network design gives the PDAs and PWAs the right to tax nearly every farm (Greg Williams, personal communication, 2003). Funds generated by the PDAs and PWAs are used to maintain ditch functionality by removing vegetation and cleaning out sediment that is impeding flow. Additional funds are appropriated by the State of Maryland Department of Agriculture for the maintenance of and implementation of drainage management practices within ditches controlled by PDAs and PWAs.

Since the construction of the Long Marsh ditch in 1789, drainage ditch construction in Maryland has lead to 821 miles of public drainage ditches and thousands of miles of privately owned ditches (Public Drainage Task Force, 2000). Roughly 40% of the drainage ditches monitored by PDAs in Maryland were constructed with both federal and state cost sharing funds in the 1950's and 60's. The last major drainage project to be completed in Maryland was the 1994 Nebo Road project which was 1.8 miles long and drained roughly 66 acres in Wicomico County (Public Drainage Task Force, 2000).



Data on the extent of drained agricultural land is from:
U.S. Department of Commerce, Bureau of the Census, 1978 Census of Agriculture. 1981. Volume 5, Special Reports, Part 5, Drainage of Agricultural Lands, AC78 – SR – 5.

Graphic by William Battaglin, USGS

Figure 4. Map of the U.S. showing the percentage of drained agricultural land in each county. Source: USGS. [Online] Available at http://d-outlet.coafes.umn.edu/education/default.htm (verified 8 Aug. 2005).

Agricultural Drainage Legislation

Since colonial times, land drainage has been subject to legislation. Several colonies passed early laws in an effort to promote land drainage for both urban development and agricultural purposes. The colonies of New York, Massachusetts, New Jersey, and South Carolina all created laws regarding land drainage. With the passing of the Federal Drainage Districts acts of 1885, 1903, and 1919, the federal government began to become involved in drainage projects throughout the U.S.

In 1944 and 1954, the passage of the Federal flood control act accelerated drainage projects in the U.S. (Ritter et al., 1995). This act provided funds for projects that limited damages due to flooding through activities such as stream straightening and deepening in watersheds less than 250,000 acres. The idea was to effectively speed the removal of water from the land and smaller streams to larger water bodies. In the 1960's, Federal Public Law 566 was passed that began a process of re-engineering and expanding of existing drainage networks (Public Drainage Task Force, 2000). The U.S. National Environmental Policy Act (NEPA) of 1969 and The Clean Water Act of 1977 were some of the first pieces of legislation that began to respond to the environmental consequences of land drainage.

The Food Security Act of 1985 (Public Law 99-198) and the 1990 Food, Agriculture, Conservation and Trade Act (FACTA) (Public Law 101-624), also known as the 1990 Farm Bill, created new rules regarding land drainage and how it could be practiced. The "swampbuster" provisions in the 1985 Bill, and amended by the 1990 farm Bill denied financial support and other farm program benefits to farmers who converted or drained wetlands for the purpose of commodity production. In the 1996 Farm Bill, the "swampbuster" provision was changed slightly to allow for more cooperation between the USDA-Natural Resource Conservation Service (NRCS) and farmers in determining wetland classification. In addition, the "swampbuster" provision provided a limit to the repair and maintenance of drainage systems. Drainage systems, both open-air ditches and tile drainage systems could only be repaired and cleaned out so far as to add no additional drainage capacity; and provided that wetland conditions had not returned to the area. The Tax reform Act of 1986, eliminated investment tax credits for land drainage which provided further disincentives to bring new lands into agricultural production through land drainage (Ritter et al., 1995). The U.S. Army Corps of Engineers is no longer issuing permits for the construction of new drainage ditches in the U.S (Public Drainage Task Force, 2000).

The State of Maryland has also passed its own legislation regarding drainage. In 1957, Maryland passed a law (Article 25, Sections 52-95 Annotated code of MD), which was amended in 1994, that created PDAs and required PDAs to have a current Operation and Maintenance Plan that has to be approved be the Maryland Secretary of Agriculture. Through USEPA Section 319 grant, the Maryland Department of Agriculture's Office of Resource Conservation has been providing funds to PDAs for the implementation of BMPs to reduce the environmental impact of land drainage. The 2002 Farm Bill, more specifically the Environmental Quality Incentives Program (EQIP), provided federal funds that are being used in Somerset County, Maryland for the establishment of flow-control structures in drainage ditches and PDAs for controlling nitrogen losses from agricultural areas. In addition, the Maryland Department of Agriculture's Office of Resource Conservation provides cost-sharing funds for drainage BMPs. Programs which are offered focus on weirs and water control-structures (which are cost-shared up to 87.5%), pocket wetlands, and expansion of vegetative buffers (cost-shared up to 100%).

Agricultural Drainage Management Practices

Several agricultural drainage ditch management practices have been developed. Water-control structures and ditch clean-outs are two such practices. Water-control structures involve a series of riser boards that are placed across the width of a drainage ditch to block the flow out of the drainage ditch. Oak or plastic boards are either added or taken away depending on the level of water control desired by the farmer. By removing boards and allowing water to exit the system, the farmer can control the water table in the fields to desired levels. This technique is also referred to as sub-irrigation. Water-control structures promote sedimentation of P-laden particles in suspension in drainage ditches as well as creating an increased potential for denitrification and anoxic conditions to form at the bottom of drainage ditches. However, anoxic conditions may create conditions favorable for the release of Fe-bound P (Sallade and Sims, 1997b; McCoy, 1999).

Figure 5. A flow control structure with removable wooded riser boards installed in a drainage ditch on the University of Maryland-Eastern Shore Campus.

Drainage ditch clean-outs are an invasive process whereby a machine dredges the drainage ditch, removing newly formed soils from the ditch and placing them either on the adjacent fields or in areas deficient in P. This process theoretically recycles the nutrients, such as P, back to the fields. However, drainage ditch clean-outs may lead to an increased sediment load in the drainage ditch effluent, which often is saturated in P. This is due to a decrease in ditch vegetation, which protects the ditch surface from scouring during ditch water flow events. Removal of material saturated in P from drainage ditches may lower the potential for P to be released to overlying waters (Nguyen and Skias, 2002).

Phosphorus Chemistry

Phosphorus is a dynamic and biologically active element found in both terrestrial and aquatic systems (Correll, 1998). Soil P is found in the +5 valence state (oxidized), due to thermodynamically instabilities at lower oxidation states which can quickly oxidize to PO_4^{-3}, even in reduced soil systems (Richardson, 1999). In all living organisms, P is an essential component of many organic macromolecules such as DNA, RNA, ATP, phospholipids of membranes, and monosesters (Wetzel, 1999). In the environment, P is most commonly found as an oxyanion that is pH dependent and can take several forms. In acidic systems (pH 2 to 7), inorganic P is found as $H_2PO_4^-$, while in alkaline systems (pH 7-12) P is found as HPO_4^{2-} (Brady and Weil, 2002). In the most alkaline systems (pH >12), P is most commonly found as PO_4^{3-} (Brady and Weil, 2002).

There are upwards of 350 unique P minerals that have been identified, with the most commonly found P minerals in the environment being apatite ($Ca_{10}(PO_4)_6F_2$), crandallite ($CaAl_3(PO_4)(OH)_2$), wavellite ($Al_3(PO_4)_2(OH)_3$), variscite ($Al_3(OH)_2H_2PO_4$), and strengite ($Fe(OH)_2H_2PO_4$) (Harris, 2002; Graetz and Nair 1999). In all of these minerals, P acts as an anionic component within the mineral (Harris, 2002).

The movement of inorganic P in the environment is generally controlled by adsorption/desorption and precipitation reactions. Adsorption is a two-dimensional reaction where there is a concentration of one species at an interface and, in the case of P, the soil-water interface (Rhue and Harris, 1999). Precipitation is a three-dimensional process where two or more species react in a solution to form an all together new solid-phase species (Rhue and Harris, 1999).

The adsorption of P on to soil particles is a biphasic reaction, with a rapid initial uptake that last just a few minutes, followed by slower reactions that may reach weeks or months (Rhue and Harris, 1999). The initial rapid uptake is controlled by ion and ligand exchange, where the period of slower uptake is controlled by the diffusion of P into the interior of minerals which is driven by both chemical and electrical gradients (Rhue and Harris, 1999).

Ion exchange occurs through the electrostatic attraction of phosphate ions to the positively charged sites that exist on variable-charged surfaces below the zero point of charge (ZPC, where the charge from cations and anions at the surface are zero). Additional sources of charge are due to hydrous oxides and clay edges. This process constitutes a small portion of P in soils and is a rapid, reversible, and nonspecific reaction (Rhue and Harris, 1999).

Ligand exchange occurs when a phosphate anion replaces a surface hydroxyl that is coordinated with a metal cation in a solid phase and is very specific (Rhue and Harris, 1999; McBride, 1994). As a result, this reaction results in the release of hydroxyls and an increase in negative surface charges (McBride, 1994). Ligand exchange reactions have also been shown to occur with sulfate and silicate minerals (Rhue and Harris, 1999).

Inorganic Phosphorus

In the environment, P is found both in inorganic and organic forms. Inorganic P in agricultural soils typically ranges from 50 to 75% of total soil P, but can range from 10 to 90 % (Sharpley, 1999). Inorganic P can be found in solution, absorbed to hydrous sesquioxides, amorphous, and crystalline Al and Fe compounds (acidic noncalcareous soils), adsorbed to Ca and Mg compounds (alkaline systems in calcareous soils), contained within P bearing minerals, or bound to humic substances (Sharpley, 1999; Graetz and Nair, 1999). In general, the vast majority of inorganic P in acidic systems is found sorbed to Al and Fe oxides and hydroxides. Amorphous Fe oxides can sorb more P that crystalline Fe minerals due to the large number of singly coordinated surface hydroxyl ions and a greater surface area. In addition, P can also be adsorbed onto Fe and Al oxides and hydroxides that are coatings on soil particles (Graetz and Nair, 1999).

The binding of inorganic P to organic molecules occurs through the formation of a complex association of Ca, Mg, Fe, and Al (pH dependent) bonds to humic substances (Petrovic and Kastelon-Macon, 1996). Organic anions in soils often compete strongly with phosphate anions for ligand exchange sites in soils (Afif et al., 1995; Kafkafi et al., 1988). Therefore, organic matter may negatively affect the ability of P to be adsorbed by soils. Adsorption sites on the surfaces of clay particles and hydrous metal oxide particles may be masked or blocked by larger humic molecules which prevents P adsorption. Organic acids produced by microorganisms and plants may serve as organic anions which compete for binding sites on soil particles with P. Chelates formed of organic molecules with Al and Fe remove potential P bonding sites from the soil system. P does not bind directly with OM; rather, complexes form where metal cations such as Al^{3+} and Fe^{3+}

bond simultaneously with functional groups on organic matter and an anion such as P (McBride, 1994; Zhou et al., 1997). Additionally, the competition for sorption sites on mineral surfaces by fulvic acids can increase P desorption by 10 to 20 % (Petrovic and Kastelon-Macon, 1996)

Organic molecules with varying molecular weights have been shown to alter P sorption/desorption reactions. Ohno and Crannell (1996) studied dissolved organic matter from animal manures, vetch and clover, and their relationship to P sorption and desorption. They found that dissolved organic matter from animal manures did not inhibit P sorption by acidic soils where as dissolved organic matter from vetch and clover did inhibit P sorption. They concluded based on ultraviolet absorbance and fluorescence data that the higher molecular weight of dissolved organic matter from animal manures (2000 to 2800) compared to vetch and clover (710 to 850) was thought to be a factor in its inability to compete with P for sorption sites.

Most inorganic P in wetlands is associated with Fe and Al and is not considered labile unless the soil system becomes reduced (Graetz and Nair, 1999). Most P in wetlands soils is associated with Fe and Al.

Organic Phosphorus

Organic P is found in the environment as relatively labile phospholipids, nucleic acids, inositols, and fulvic and humic acids (Sharply, 1999). In the soil environment, organic P is often associated with the positively charged sites on clay particles, organic matter, and cations in the soil solution (Wetzel, 1999). Organic phosphorus is considered a labile or short-term storage sink for P and its cycling is dependent on P mineralization rates (Richardson, 1999).

Oxidation-Reduction and Phosphorus

Oxidation-reduction reactions play a fundamental role in the biogeochemistry of saturated soils found in wetland environments. Soils are commonly referred to as "reduced" when there is a low oxidation-reduction potential in the soil (Ponnamperuma, 1972). After saturation by water, oxygen concentrations in the soil solution decrease. Oxygen is used as an electron receptor by microbes, and because oxygen diffusion through water is 10,000 times slower in water than air, it is not replenished as quickly as is consumed. Once redox potentials reach below about 120 mV (pH=5), the reduction of Fe minerals from Fe^{3+} to Fe^{2+} may occur, releasing P in the form of soluble P to overlying waters (Moore and Reddy, 1994; Ponnamperuma, 1972). However, research has shown that transformation of Fe^{3+} to Fe^{2+} may occur at moderate Eh values (200 to 300 mV) in acidic systems (pH 5-6) (Holford and Patrick, 1979).

Suter et al. (1998, 1991) described the reduction of Fe minerals from Fe^{3+} to Fe^{2+} via three mechanisms. In acidic conditions, the detachment of Fe^{3+} from the mineral lattice occurs as protons are adsorbed to the surface of the oxide, which facilitates the detachment and dissolution of Fe^{3+} to Fe^{2+}. Reduction of Fe^{3+} is also enhanced by specifically adsorbed chelate ligands. When Fe^{3+} is found on the surface of an oxide and a reductant (reducing organic ligands or metal complexes) is adsorbed to it, Fe^{2+} is released quicker than Fe^{3+} due to the weakened bonds between the Fe^{2+} and the O^- ions of the crystalline lattice. The reduction of crystalline Fe also results in changes to more amorphous Fe forms (Richardson, 1999).

The sorption of phosphates in soil is strongly influenced by the redox potential in soils. Vadas and Sims (1998) studied the effects of reducing conditions on soluble P losses from wooded and cultivated Pokomoke loamy sand (coarse-loamy, siliceous, thermic Typic Umbraquult) and Fallsington sandy loam (fine-loamy, mixed, mesic Typic Ochraquult) soils. Soils were incubated in reducing conditions (200-350 mV) for a 28-d period. Upon re-oxidization, soluble P concentrations decreased (1.67 mg/kg) substantially in the cultivated Pokomoke A as well as the Fallsington (0.39 mg/kg) relative to reduced conditions. Water-soluble P concentrations in both soils increased slightly (0.05 mg/kg) after they were re-oxidized relative to pre-reduction values indicating that after a oxidation-reduction cycle, water-soluble P losses may increase over time.

The dissolution of Fe minerals is a dominant process that controls P solubility in anaerobic conditions (Rhue and Harris, 1999; Holford and Patrick, 1979).While adsorption of P by ligand exchange with newly formed ferrous iron oxides may occur, P may also precipitate directly with Fe^{2+} to form other phosphate minerals (Rhue and Harris, 1999).

In acidic soil systems, pH tends to rise with prolonged anaerobic conditions due to the consumption of H^+ ions in redox reactions and the subsequent increase in activity of OH^- . With an increase in pH, water-soluble P decreases due to ferrous iron that precipitates into new mineral phases which have higher surface areas. This is why P sorption capacities and oxalate-extractable iron concentrations tend to be greater after soil reduction (Patrick and Khalid, 1974; Khalid et al., 1977).

Large quantities of P-reactive Fe can be released through formation of FeS (Roden and Edmonds, 1997). At low redox potentials which are necessary for Fe reduction, sulfides can react with Fe^{2+} to form FeS. Because the solubility of Fe^{2+} is lower than that of ferrous oxide minerals, precipitation of FeS will drive the dissolution of these minerals, which results in the release of sorbed P (Rhue and Harris, 1999)

Phosphorus Transport

Agricultural Ecosystems

Phosphorus is transported as either particulate P (PP) or dissolved P (DP). There is no gaseous form of P in the natural environment akin to nitrogen, and therefore atmospheric deposition is not commonly measured. Particulate P is P sorbed to soil particles and attached to organic matter that is eroded from the land by surface runoff (Sharpley, 1999). This fraction makes up 60 to 90% of P loss and is a long-term source of organic P to local streams and water bodies (Sharpley, 1999).

Dissolved P is released from soil particles and OM through desorption based on chemical gradients, dissolution from P bearing minerals, and through the extraction of P from plant and other soil organisms (Sharpley, 1999). Dissolved P is immediately available to algae; its concentration in runoff is dependent on P saturation of soils, clay, Fe and Al oxides, carbonates, and OM (Pote et al., 1996; Sharpley, 1999).

Drainage Ditches

Phosphorus in drainage ditches is transported in dissolved and particulate forms from both overland flow and in shallow ground water. Mozaffari and Sims (1994) found that drainage ditches on the Delmarva can serve as key pathways for subsurface movement of P indicating that the source of P in ditch drained-agroecosystems may not only be the result of surface runoff. Phosphorus sorption onto Fe hydroxides are some of the most substantial P retention processes within drainage ditches (Boers et al., 1998). The burial and immobilization of sediment-bound P in deeper sediments may also play a role in P storage in drainage ditches.

Drainage ditches act as both a nutrient sink and source of P. As a nutrient sink, drainage ditches serve as sink for inorganic, organic, and dissolved P that is transported via erosion/runoff and overland flow (Sims et al., 1996). Lateral flow (<5m) of shallow ground water from P-enriched soils adjacent to ditches may also increase P additions to drainage ditches (Sims et al., 1996). Sediments laden with P and organic matter may be retained in drainage ditches until storm flow conditions are present, at which time these materials may be transported out of the ditch system.

Ditches act as nutrient sources to surface streams and water bodies (Sallade and Sims, 1997a). Throughout the spring and summer, base flow and stagnant conditions in drainage ditches lead to anoxic conditions as a result of the decomposition of organic matter, high temperatures, and low dissolved oxygen concentrations (McCoy et al., 1999). If anoxic conditions are present, ferric iron-bound P is released and transported as dissolved P. Small rain events have been shown to increase the suspended sediment load, and with it, an increase in P concentrations due to sediment bound P (House et al., 1995;

McCoy et al., 1999). Low flow conditions promote the accumulation of material that is easily transportable once high flow resumes in drainage ditches during storm flow (McCoy et al., 1999).

Sallade and Sims (1997a) examined P-enriched soils in agricultural drainage ditches in Delaware's Inland Bays' watershed. They sampled 17 drainage ditches that ranged in length from 450 to 900 m and 1 to 2 m in depth. Samples were collected from an upper layer (0-5 cm) and a bottom layer (5-15 cm). Coarse organic debris on the sediment surface was removed prior to sampling. Characterization of ditch soils included particle-size analysis, pH, and organic carbon. Analyses for P included Mehlich 1-P, and sequential extraction of inorganic P forms (soil/solution, 1:150,17 h), biologically available P (BAP) (0.1 M NaOH-extractable P), water-soluble P, and Langmuir sorption isotherms.

Dry sediment pH values ranged from 4.4 to 7.0; with the lower soils being somewhat more acidic relative to upper soils. Upper soils had a median organic matter content of 8% while lower soils were found to have a median of 3% organic matter content. Upper soils were finer in texture than lower soils. Finer textures in the upper layer were attributed to selective erosion of adjacent field soils. Additionally, finer textures found in the upper soils were correlated to increases in total Fe and Al oxide contents over their respective lower soils.

Analysis of both the upper and lower soils P showed that Mehlich 1-P values were in fact lower in the drainage ditch soils compared to adjacent field soils. Drainage ditch soils were found to be higher in Mehlich 1-P than corresponding field soils at the same depth relative to the ditch. In the upper soils, mean BAP values (384 mg kg^{-1}) were more than two times the mean BAP values (170 mg kg^{-1}) of the corresponding lower soils. Sequential fractionation indicated that the greatest quantity of total P was stored as inorganic P sorbed to Fe and Al oxides. P extracted by NH_4F averaged 69% of the total P while NaOH extractions averaged 26%. HCl extracted P had a mean of 5% of the total P extracted.

In a follow up two-part study, Sallade and Sims (1997b) investigated the effects of prolonged reducing conditions on P release from drainage ditch soils and the influence of temperature on P release from soils. The samples used in this experiment were the same as those in Sallade and Sims (1997a). A total of 77 samples, 37 from (0-5 cm) and 40 from (5-15 cm) were analyzed. Initial Eh and pH measurements were taken, and then the soils were incubated for 21 days under N_2 gas to represent reducing conditions found in drainage ditches. In addition, six soils were analyzed for the effects of temperature by incubating in the dark in triplicate at 7 and 35°C (cool winter and hot summer temperatures in the region) and analyzed for Eh, pH, and soluble P, Fe, Al, Ca, Mn, and S.

In soils from (0-5 cm) pH increased 0.7 units and soils from (5-15 cm) increased 0.6 units as predicted for soils with decreasing Eh values. Mean soluble P concentrations were nearly four times greater than initial concentrations after the 21 day incubation period and were correlated ($r^2 = 0.66$) with increases in soluble Fe concentrations. Increases in soluble Mn were also noted, while soluble Al concentrations decreased after the incubation period.

The results of temperature effects on soluble P losses in general showed rapid increases in soluble P concentrations within 1-7 days, and a leveling off after day 21. At day 21 of the study, soluble P losses in one sediment at 7°C (0.28 mg P L^{-1}) were lower than at 35°C (0.69 mg P L^{-1}) indicating that sediments in winter-like temperatures may have the capacity to increase soluble P concentrations in overlying waters.

Spatial Distribution of Phosphorus in Wetlands

All soil properties demonstrate differences in their spatial dependence which can be determined by their semivariograms. The semivariance of a soil property should increase with increasing distance, or the lag distance (h) (McBratney and Webster, 1986). Semivariance should continue to increase until it reaches the maximum variance as an asymptote (the sill) (Cambardella et al., 1994). The lag distance where the sill is reached is called the range of spatial dependence. Values less than the range exhibit special dependence, while those greater than the range are not related spatially (McBratney and Webster, 1986). At $h = 0$, the semivariance is termed the nugget variance which corresponds to measurement and experimental errors, and random variability at ranges smaller than the closest sampling distance (Nielsen and Wendroth, 2003; Cambardella et al., 1994; McBratney and Webster, 1986).

The spatial distribution of P in riverine wetland ecosystems, and or that matter, in most terrestrial environments, has not been thoroughly investigated. The causes of spatial variability in P are very complex. The spatial variability of intertwined spatial dependencies of P and other soil properties such as soil texture, Al, Fe, Ca, and organic carbon make investigations very difficult. These factors might operate at different spatial scales in a particular wetland and distinguishing between spatial relationships and P sorption relationships among the variables can be problematic (Bruland and Richardson, 2004).

An investigation of forested riparian wetlands in North Carolina found that P sorption (Phosphorus sorption index) was spatially autocorrelated, and Al_{ox} and Fe_{ox} were also well correlated at 13.2 m and 7.6 m respectively (Bruland and Richardson, 2004). It was shown that due to the fact that these soil properties are spatially autocorrelated and that there is heterogeneity in soil properties including P in riparian wetlands, soil sampling designs for P studies in wetlands should be conducted with a spatial structure.

Objectives

The objectives of this study were to (1) describe and characterize soils formed in agricultural drainage ditches on the lower Delmarva Peninsula; (2) investigate the spatial and vertical distribution of P within an agricultural drainage ditch network; (3) create a soil survey of relevant areas on the University of Maryland Eastern Shore Research Farm.

Chapter 3: Morphology and Characterization of Ditch Soils at an Atlantic Coastal Plain Farm

ABSTRACT

The materials in drainage ditches, which have traditionally been referred to and studied as sediments, may in fact be soil bodies. In this study, we described and characterized materials found in vegetated drainage at the University of Maryland Eastern Shore Research Farm in Princess Anne, Maryland. Sixty-nine profile descriptions were performed along 10 agricultural drainage ditches ranging in length from 225 to 550 m. Particle-size, pH, and organic carbon were analyzed on 21 representative profiles. The materials meet the definition of a soil. Pedogenic processes operating in these soils include organic matter humification and accumulation, structure formation, iron oxidation and reduction, sulfuricization, sulfidization, translocations, and bioturbation. These ditch soils were generally A horizons formed in loamy alluvial sediments eroded from loess-derived topsoils over gravelly and sandy C horizons formed in Coastal Plain sediments. Soil structure was described in 75% of A horizons. Redoximorphic features were described in 41% of A and 63% of C horizons. Organic carbon ranged from 0.04 to 12.4%. Monosulfidic black oozes were observed on some soil surfaces; sulfidic materials were observed at depth. Shallow ditches (>1.5 m) tended to have structure and a layer in the substrata with a bright matrix color. Deep ditches (1.5 to 4 m) tended to have high n-value, structureless sola, and gleyed subsoil horizons. Studies of the chemical, physical, and biological processes operating in drainage ditches should integrate an understanding of the pedological processes operating in these soils.

Abbreviations: OC, organic C; MBO, monosulfidic black ooze; CRS, Chromium-reducible S; AVS, acid volatile S; UMES, University of Maryland Eastern Shore.

Agricultural drainage ditches are long linear features that are constructed to lower the water table and speed the removal of excess surface runoff to local streams and water bodies. They are found in poorly drained, tile-drained, and irrigated landscapes that require artificial drainage for profitable agricultural production (Janse and Van Puijenbroek, 1998). Ditches function as a hydrological link between surface runoff, ground water, and surface waters (Janse and Van Puijenbroek, 1998). In regions with intensive agriculture, ditches have the potential to act as key pathways for the export of nutrients from areas of intensive agriculture to surface waters (Sallade and Sims, 1997a; Vadas and Sims, 1998; Nguyen and Sukias, 2002).

Land drainage for both public and agricultural use has been adopted around the world. Land drainage systems were developed in Mesopotamia around 9,000 B.P. (van Schilfgaarde, 1971). Surface-drainage system networks were also developed by the Egyptians and the Greeks around 2400 B.P. (Shirmohammadi et al., 1995). In the U.S., organized drainage for agricultural and public purposes began in the 1600's (Evans et al., 1996). Many states rely on land drainage, in particular subsurface drainage (tile drainage) and surface drainage (open-air ditches), to control ground water levels in both agricultural and urban areas. In the Midwest, several states rely heavily on drainage for agricultural proposes with nearly 37% of arable land in this region requiring drainage (Fausey et al., 1995). Today, the coastal plain of North Carolina has more than 800,000 ha that rely on drainage ditches (Evans et al., 1996). Currently in Florida, there are more than 2.5 million ha that are affected by artificial land drainage (Thomas et al., 1995).

Previous investigations of open-air agricultural drainage ditches (Sallade and Sims, 1997a; Sallade and Sims, 1997b; Nguyen and Sukias, 2002) have described the organic and mineral materials found in drainage ditches as sediment rather than as natural soil bodies. Conceivably, if the materials within ditches are identified and described as natural soil bodies, they should be understood, modeled, and managed as soils. The ability to understand materials within drainage ditches as natural soil bodies and not as sediment would allow for their classification and facilitate their mapping. The ability to accurately map and characterize drainage ditch materials within drainage ditch networks would enhance the development of ditch nutrient management plans. Materials found within ditches that have the ability to support rooted vegetation and have distinguishable horizons or layers formed as a result of pedogenesis may meet the definition of a soil (Soil Survey Staff, 2003). Materials that lack evidence of pedogenesis and have unaltered layers as a result of sedimentation are sediments.

Due to periodic or continuous saturation, materials within drainage ditches may have properties similar to wetlands and subaqueous soils (Demas and Rabenhorst, 1999; Bradley and Stolt, 2003). Wetland properties such as surface or subsurface hydrological connections, high organic matter contents, reducing soil conditions, and periodic oxidizing events may occur in ditches. Therefore, drainage ditches are likely to function similarly as wetlands in terms of their nutrient cycling processes and retention capabilities (Bowmer et al., 1994; Nguyen and Sukias, 2002). Both drainage ditches and subaqueous soils are heavily influenced by overlying water bodies (Demas and

Rabenhorst, 2001). Therefore, the application of select pedogenic concepts developed through the study of subaqueous soils may apply to drainage ditches. Situated within a drainage channel, ditch materials may also be influenced by fluvial processes.

Mineral and organic materials from a variety of sources act as parent materials for drainage ditch materials. Mineral material can accumulate from the sedimentation of suspended soil from cultivated fields in surface runoff water, the slumping of drainage ditch sidewalls, and from precipitates such as iron oxides that are the result of solutes transported in ground water (Sallade and Sims, 1997a; Nguyen and Sukias, 2002). Organic additions to drainage ditches include particulate and dissolved organic matter from adjacent cultivated fields in surface and subsurface runoff and in-situ deposition of organic plant materials growing within the ditch including: algae, microbes, and macro invertebrates (Sallade and Sims, 1997a; Nguyen and Sukias, 2002).

Most large drainage ditches on Maryland's Eastern Shore are cleaned out periodically, approximately once every 10-30 yrs., or when a blockage occurs (Public Drainage Task Force, 2000). Clean-outs usually involve the removal of mineral and organic materials from ditches by mechanical means and placing it adjacent to the drainage ditch or in an adjacent cultivated field (Public Drainage Task Force, 2000). Drainage ditch clean-outs profoundly alter the properties of ditches. Smaller drainage ditches many only be cleaned when a blockage occurs, and therefore, may not be cleaned out on a regular basis.

Sulfidic materials are formed by the process known as sulfidization, the mineral transformation process where the products are most commonly iron-monosulfides (FeS) and pyrite (FeS_2) (Fanning and Fanning, 1989; Fanning et al., 2002). The conditions for sulfidization are a source of S, a bacteria capable of reducing sulfate, a source of reactive iron, organic matter as a microbial substrate, and anaerobic conditions [roughly an Eh (mV)=0 at pH=6] (Fanning and Fanning, 1989; Demas and Rabenhorst, 1999; Fanning et al., 2002). As sulfate reducing bacteria respire and oxidize organic matter sulfide ions are produced, which can then react with ferrous iron (Fe^{2+}) in the soil solution to form iron-monosulfides (FeS), often in the form of mackanawite (FeS) (Rabenhorst and James, 1992). If anaerobic conditions are maintained for long periods of time, pyrite (FeS_2) tends to form from iron-monosulfides and persist due to its stability in anaerobic environments (Fanning et al., 2002). When sulfidic materials are exposed to aerobic conditions, the oxidation of 1 mol of pyrite yields 2 mol of sulfuric acid, which can greatly acidify soils that do not contain enough bicarbonate to neutralize the acid (Fanning et al., 2002).

The objective of this study was to examine materials within agricultural drainage ditches through a pedological framework to determine if they are soils and to understand their properties.

MATERIALS AND METHODS

Study Area

This study was conducted on the University of Maryland Eastern Shore (UMES) Research Farm (N 38^{o} 12’ 22”, W 75^{o} 40’ 35”) located in Princess Anne, Somerset County, Maryland (Fig. 6). The farm is approximately 81 ha in size with more than 9 km of open-air agricultural drainage ditches. The farm has a 30-year history of poultry litter application as fertilizer. All drainage ditches on the farm drain into the Manokin Branch that runs along the western boundary of the farm. The Manokin Branch eventually forms the Manokin River one mile downstream of the farm. The research farm has an average elevation of 7 m above mean sea level with a relatively flat relief. The farm receives an average of 1110 mm of rainfall per year (NRCS, 2005). Agronomic crops grown on the farm are corn (*Zea mays* L.), soybean (*Glycine max* L.), and wheat (*Trticum aesitivum* L.). The parent materials of field soils at UMES are silt loam loess over sandy Atlantic Coastal Plain sediments. Field soils on the farm were mapped primarily as consociations named for the dominant soil series of Othello (Fine-silty, mixed, active, mesic Typic Endoaquults), Mattapex (Fine-silty, mixed, active, mesic Aquic Hapludults), Matapeake (Fine-silty, mixed, semiactive, mesic Typic Hapludults), and Portsmouth (Fine-loamy over sandy or sandy-skeletal, mixed, semiactive, thermic Typic Umbraquults) (USDA-NRCS, 2005; Matthews and Hall, 1966). The majority of field soils are poorly to very poorly drained and would be agriculturally unproductive without artificial drainage. Characteristics of the farm are generally representative of other ditch-drained farming operations in the region.

Drainage Ditch Selection

A total of ten drainage ditches were selected for this study to reflect the diversity of drainage ditch function, size, and hydrological properties found on the UMES Research Farm (Fig. 6). The term “primary ditch” was used to describe open-air ditches that drain surface runoff from cultivated fields and are connected to shallow ground water sources. Primary ditches tend to be shallow in nature (<1.5 m), often contain stagnant water until storm flow conditions are present, and dry-out periodically throughout the year. The term “collection ditch” was used to refer to ditches that function to transport flow from primary ditches. Two types of collection ditches were described: shallow-collection and deep-collection ditches. Shallow-collection ditches are 1.5 to 2 m deep, connected to shallow ground water, and dry out periodically. Deep-collection ditches are deep (>2m), connected to deep, regional ground water supplies, and have stagnant and continuous water flow throughout the year.

Seven primary, one shallow-collection, and two deep-collection ditches were included in the study. Five drainage ditches were oriented in an east-west direction while the remaining five drainage ditches were oriented in a north-south direction (Fig. 6). Drainage ditches are identified using the letter D, as well as the letter X for primary field ditches, XXS for shallow-collection ditches, and XXD for deep-collection ditches. A numerical identifier (e.g. 1, 2, 3…) was used to distinguish between individuals of the same type (e.g., DX1, DX2, DXXS1, and DXXD1). The drainage ditches ranged in length from 225 to 550 m with a mean of 325 m. Primary drainage ditches located within fields ranged in depth from less than 1 m (mean depth = 0.5 m) while collection drainage ditches ranged from 1.5 to 4 m (mean depth = 2 m).

Field Methods

Drainage ditch material description sites were identified within the center of each drainage ditch using a measuring wheel with a wheel circumference of 1 m. Each description site was located at 40 m intervals starting from the outlet of each drainage ditch. All descriptions were performed in the summer of 2004. Each ditch material profile was excavated to at least 1 m. A hand spade shovel was used to excavate the ditch materials to a depth of approximately 40 cm; the remainder of the profile was excavated using a 7.6-cm bucket soil auger.

At each drainage ditch material description site, morphological characteristics, horizonation, horizon boundaries, structure, moist consistence (where applicable), and redoximorphic features were described based on standard soil survey techniques (Soil Survey Staff, 1993). All profile descriptions were performed by the senior author. Color was described using a Munsell® soil color chart in the field, and field soil texture was performed by hand. Redoximorphic features were described as a percentage of the matrix by visual estimation (Soil Survey Staff, 1993). The presence of ferrous iron was assessed in all ditch horizons using alpha-alpha' dipyridyl dye in neutral, 1-normal ammonium acetate solution (Childs, 1981; Soil Survey Staff, 2003). Samples of at least 500 g of each identifiable horizon were collected. Samples of less than 500 g were collected if the horizon was thin. Samples were placed into labeled plastic bags, and packaged in coolers for transport. Samples were air-dried (25^{o} C), coarse organic debris was removed, and were ground to pass a 2-mm sieve. Ditch profiles were classified by Soil Taxonomy (USDA-NRCS, 2003).

A total of 69 profile descriptions were performed within 10 drainage ditches at UMES (Fig. 6). An average of 7 profile descriptions were performed in each drainage ditch with the fewest in DX8 (n=1) and the maximum number of descriptions in DXXD2 (n=10). A subset of 21 profiles was chosen to represent each drainage ditch for further laboratory analyses. At least one profile was selected from each drainage ditch.

In addition to the profile sampling described above, five samples of ditch materials were collected at 40 m from the outlet in ditches DX1, DX2, DX3, and 80 m from the outlet DX3 in September and October of 2004 for sulfur fractionation. Surface soil samples were collected from 0-5 cm and subsurface samples ranging in depth from 28-107 cm. Surface samples were collected by hand with a spade shovel and subsurface samples were collected using a 7.6-cm soil bucket auger. Samples were placed into plastic bags and placed on ice in a cooler. The samples were then brought back to the laboratory and stored at -15 ^{o}C until analyzed.

Laboratory Methods

Particle-size analysis was performed by pipette (Gee and Bauder, 1986) and moist pH was performed using a soil to water ratio of 1:1. Organic C (OC) was determined using a high temperature CNS-analyzer with an infrared detector (Bremner and Tabatabai, 1971). Acid volatile sulfur (AVS) and chromium reducible sulfur (CRS) fractionations were performed utilizing the Johnson-Nishita apparatus (Cornwell and Morse, 1987). Acid volatile sulfur fractions generally capture iron-monosulfides (FeS), while CRS captures pyrite, iron-monosulfides (FeS), and elemental S (Smith, 2004). This method specifically reduces different sulfur fractions to hydrogen sulfide gas. The hydrogen sulfide gas was transported through the apparatus using an N_2 gas carrier that was kept at a flow rate between 40 and 70 ml min^{-1} (Hussein and Rabenhorst, 1999). Approximately 30 ml of sulfide antioxidant buffer (SAOBII) was used to trap the hydrogen sulfide gas (Cornwell and Morse, 1987). Sulfur concentration in the SAOBII was determined by potentiometric titration with [$Pb(ClO_4)_2$] using an [AgS] electrode along with a double junction reference electrode for end point detection. Acid volatile sulfur was determined by digesting a sample (with an equivalent dry weight between 1 to

2 g) for 45 min using 30 ml of cold 6 N HCl (Cornwell and Morse, 1987). The SAOBII trap was removed after 45 min of digestion for titration. Chromium reducible sulfur was determined from the same sample following further treatment by 10 ml of ethanol, 40 ml of reduced chromium solution, and 20 ml of concentrated HCl. The mixture was brought to a slow boil for approximately one hour, after which the SAOBII trap was removed and potentiometrically titrated (Canfield et al., 1986).

Data Analyses

Classical descriptive statistics were used to describe the results of individual variable analyses. In order to compare characterization data between horizons, eight morphological horizon classes were defined based on similar morphological and genetic characteristics (Table 1). Kolmogorov-Smirnov tests and descriptive statistics were used to assess normality. Statistical analyses were performed using S-Plus (Insightful Corporation, 2001) and the SAS GLM procedure (SAS Institute, 1990). The CONTRAST statement in the SAS GLM procedure was used to test pre-planned one-way comparisons between the means of morphological horizon classes: Oi greater than A horizons (Dark A, Gley A); Dark A greater than Gley A horizons; A greater than subsoil C horizons (Gley C, Bright C, and Sulfidic C); Bright C greater than Gley C horizons; and Sulfidic C greater than Gley C horizons.

RESULTS

Morphology

Ditch materials were found to contain layers that followed a sequence consistent with soil horizons (i.e. A horizons, underlain by B and C horizons). For ease of reporting the results of profile descriptions, layers will be discussed in terms of soil horizons. Horizonation in general was found to be A horizons formed in recent alluvium over C horizons formed from fluvial deltaic and marine sediments with the exception of nine profiles that contained an Oi horizon at the surface and three full profiles described with C horizons throughout. Thirty-seven percent (69 of 185) of A horizons were gley (value 4 or more, chroma 2 or less) in color; only six of these horizons were found at the soil surface. The mean depth at which Gley A horizons first appear was 9 cm. Of the ditch C horizons, 68% were gleyed.

Ditch A horizons were, in general, dark in color. The hue in 90% of A horizons was 10YR. Color value ranged from 1 to 5 with a mean of 3.3. Chroma ranged from 1 to 4 with a mean of 1.8. Ditch C horizons were lighter in color relative to the overlying A horizons. Soil hue in C horizons was primarily 10YR (64%) or 2.5Y (29%). Color value ranged from 2.5 to 8 with a mean of 5.0. Chroma ranged from 1 to 8 with a mean of 2.0.

Redoximorphic features were found in 41% of ditch A horizons. Concentrations of Fe as soft masses, pore linings, and coatings were the only redoximorphic concentration types observed. Of the 76 A horizons described with redoximorphic features, 20 horizons (26%) contained at least 2 types of features and 7 horizons (9%) contained at least 3 types of features. Manganese concentrations were not observed.

Mean depth to first concentration was 8 cm. Only 6 of 56 surface horizons contained redoximorphic features, 4 of which were described as Fe concentrations along pore linings. Concentrations ranged from 1 to 10% of the matrix with a mean of 3%. Concentration color hues were primarily 10YR (55%) and 7.5YR (25%). Value ranged from 3 to 7 with a mean of 4.0, chroma ranged from 4 to 8 with a mean of 5.5. Concentrations were predominantly fine to medium (<5 mm) in size (90%), distinct in contrast (65%), and described as masses and pore linings (90%). Only 23% of all ditch A horizons (non-gleyed and gleyed) had depletions. Depletion color hues were 10YR (74%), values ranged from 4 to 7 with a mean of 5.6, chroma ranged from 1 to 2 with a mean of 2. Depletions were medium (<5 mm) (74%) in size, faint to distinct in contrast (Schoeneberger et al., 2002)

Sixty-three percent of ditch C horizons contained redoximorphic features of which 21% contained at least two types of redoximorphic features. Redoximorphic features in C horizons were described primarily as concentrations of Fe (73%), which ranged from 1 to 40% of the matrix with a mean of 5%. Concentration hues were primarily 10YR (67%) or 7.5YR (19%); values ranged from 2 to 7 with a mean of 5; chromas ranged from 4 to 8 with a mean of 6. The majority of concentrations were medium (66%) in size, prominent (70%) in contrast; they were found as soft masses of iron (97%) or pore linings or ped faces (3%). Depletions accounted for 17% of the redoximorphic features described in C horizons. Depletion hues were mainly 10YR (51%), values which ranged 4 to 8 with a mean of 6.1, and chromas ranged from 1 to 2 with a mean of 2. In zones where they occur, iron depletion were generally medium (81%) in size and were faint (32%), distinct (32%), or prominent (35%) in contrast.

Soil structure was found to be present in 138 of 185 A horizons (75%) (Tables 2 and 3). Sixty-eight percent of soil structure in A horizons was found to be of weak grade with 52% of the structure as subangular blocky and 48% as granular. In ditch A horizons where no grade of structure was observed, 55% were single-grained while 45% were massive. Ditch C horizons were dominantly structureless (97%) and single-grained (90%).

Roughly 36% (25 of 69) of ditch soil profiles described at UMES contained sulfidic materials as described by Soil Taxonomy (Table 4) (Soil Survey Staff, 1999). A total of 42 horizons were described as containing sulfidic materials. Of those, 93% were described as C horizons and 7% as either A or B horizons. The mean depth to sulfidic materials was 64 cm (Table 4). Two profiles in DXXD3 contained sulfidic materials within 25 cm of the surface and one profile in DX3 contained sulfidic materials within 28 cm of the soil surface.

Characterization

Overall, soil pH values ranged from 2.6 to 6.1 with a mean of 4.7 and a standard deviation of 0.7. Mean horizon class soil pH values were very acidic and ranged from 3.9 (Surface C horizons) to 5.3 (Organic Oi horizons) (Table 5).

A subset (n=126) of all samples collected (n= 449) were analyzed for particle-size distribution. Fifty-seven ditch A horizons were analyzed. These horizons were found to be loamy in texture: 46% sandy loam and loams and 13% clay loam and sandy clay loams. Sand content ranged from 10 to 95% with a mean of 56%. The sand fraction was generally coarse and medium sands (averaging 32% of sand fraction). Fine and very fine sands comprised a mean of 15% of the sand fraction. Silt content ranged from 3 to 79% with a mean of 28%. Clay content ranged from 2 to 43% with a mean of 16%.

Ditch C horizons were coarser textured and were dominated by very gravelly sands, gravelly sands, and sands (54%). Gravelly loamy sands and loamy sands were 28% of C horizons. Sand content ranged from 36 to 98%, with a mean of 86%. Medium and fine sands comprised more than half (51%) of the sand fraction. Silt content ranged from 0.4 to 40% with a mean of 8%. Clay content ranged from 1 to 24% with a mean of 16%.

Ditch A horizons were found to be enriched with organic C relative to ditch C horizons. Organic carbon contents ranged from 0.04 to 12.4% (mean = 2.4) in all soil horizons analyzed. Morphological horizon classes Dark A and Oi were significantly higher in organic carbon relative to all other horizon classes (P=<.0001) (Table 5). Moreover, Dark A horizons were found to be more enriched in organic carbon than Gley A horizons (P= 0.05). Surface C horizons were found to have the lowest organic carbon compared to Dark A, Gley A, and Oi horizons (Table 5).

Acid volatile sulfur contents of the four surface samples analyzed (0-5 cm) ranged from 0.037 to 0.13% and CRS contents ranged from 0.030 to 0.75% (Table 6). The two samples taken at depth (28-107 cm) had AVS contents ranging from 0.0026 to 0.0039% and CRS contents ranging from 0.15 to 0.20% (Table 6).

DISCUSSION

Soils or Sediments?

The mineral and organic materials found within agricultural drainage ditches on the UMES Research Farm qualified as soils that support vegetation and have distinguishable soil horizons formed as a result of pedogenesis. We found evidence of the following pedogenic processes operating in these soils: organic matter decomposition and accumulation, soil structure formation, iron transformations, sulfuricization, sulfidization, diffusion, and bioturbation.

Pedogenic Processes

Key to the formation of ditch soils is the growth, death, and decay of in-situ vegetation and fauna. Organic matter is added through processes such as the littering of in-situ vegetation and the death of algae and other microorganisms. Additionally, dissolved organic matter and particulate organic debris from adjacent fields may be deposited into ditches through erosion, ground water, and direct inputs from farm operations such as harvest and manure applications. Surface soils of drainage ditches (Oi, Dark A, Gley A) (mean= 4.6%) were higher in mean organic carbon than C horizons (Bright C, Gley C) (mean=0.38%) (P= <.0001). Ditch soils at UMES were similar to surface soil horizons (0 to 5 cm) from ditches in Delaware that were found to have a mean soil organic matter content of 8% (Sallade and Sims, 1997a).

The structure in ditch soils was weak to moderate grade. Soil structure was most commonly found in the A horizons of primary and shallow-collection ditches. Soil structure was rare in C horizons or deep-collection ditches. The development of soil structure in ditches is presumably the result of wetting and drying cycles as well as aggregation by plant roots and soil fauna (Fanning and Fanning, 1989). The high n-value in deep-collection ditches, which is common in silty/loamy subaqueous soils, indicates permanent saturation and indicates a connection to regional ground water (Demas and Rabenhorst, 1999).

Evidence of gleization, oxidation, and the translocation of Fe was found in ditch surface horizons and subsurface horizons in the form of Fe concentrations and depletions and gleyed matrices. Positive reactions to alpha-alpha' dipyridyl dye in selected ditch horizons indicated the presence of ferrous (Fe^{2+}) iron (Childs, 1981; Soil Survey Staff, 2003). With in-situ decaying plant matter acting as a source of oxidizable organic matter for microbes, stagnant water conditions and warm temperatures, low redox potentials were expected (McCoy et al, 1999). An anaerobic environment within drainage ditches may potentially have large consequences on nutrient solubility and mineral transformations due to the dissolution of ferric (Fe^{3+}) iron and the subsequent release of Fe-bound P (Reddy et al., 1995; Vadas and Sims, 1998).

The presence of a light C horizon with visually appreciable amounts of oxidized iron and a matrix value of 5 or more, chroma of 3 or more, was described in 18 ditch profiles (Table 4). This horizon was found in both primary and shallow-collection ditches at roughly 1 m. This horizon was also observed in field soils adjacent to primary and shallow-collection ditches at approximately 1 m in depth and within 40 m of the ditch (Vaughan, 2005). Research in North Carolina, USA indicated that in a period of less than 30 yr, ditching can alter the morphology of surrounding soils by significantly increasing the quantity of concentrations within 7 m of drainage ditches (Hayes and Vepraskas, 2001). Fluctuating water tables in primary and shallow-collection ditches on the UMES farm may be producing this hydromorphology. The connection of deep collection ditches to deep ground water prevents their drying out and subsequent oxidation, thereby preventing the formation of this horizon. A second explanation for the formation of this horizon is the high amounts of iron released during the oxidation of sulfidic materials at depth. The oxidation of pyrite in soils can yield substantial amounts of ferrous iron, which may move up the profile through wicking and diffusional processes (Fanning et al., 2002). This could occur as the shallow ground water fluctuates seasonally. In the summer, as the ground water is lowered due to higher evapotranspiration rates, sulfidic materials may become oxidized as the ditches dry out. In winter when evapotranspiration rates are at their lowest levels during the year, ground water containing the dissolved ferrous iron may rise towards the surface and precipitate at the shallow groundwater boundary.

In addition to sulfidic materials found at depth in ditch profiles at UMES, the presence of jet-black iron-monosulfides (FeS) was also discovered. The monosulfides are black (N 2.5/1) in color and found within or on top of ditch mineral soil surfaces, and often intermixed with coarse organic debris on top of ditch soils. Monosulfides were identified in several ditches in the fall of 2004, and confirmed through acid volatile sulfur (AVS) and chromium reducible sulfur (CRS) fractionations (Table 6). The ditches contained roughly 6 to 10 cm of water at the time of sampling. However, iron-monosulfides were not described on ditch soil surfaces in the summer of 2004; during this time the ditches were dry and presumably oxidized. Iron-monosulfide formation requires low redox potentials, a source of S, and oxidizable organic matter, and respiring sulfate reducing bacteria. Sulfide ions are formed by the respiring bacteria, which then can react with ferrous iron in the soil solution to form iron-monosulfides (FeS), often in the form of mackanawite (FeS) (Rabenhorst and James, 1992).

Two possible sources of sulfur in drainage ditches at UMES are geologic deposits of sulfidic materials from which sulfate may be wicking up the soil profile or recent (<30 yr) additions of poultry litter. The origin of this geologically deposited sulfidic material is thought to be a result of a past marine transgression. The last marine transgression is thought to have occurred either 82,000 years B.P. or 125,000 years B.P. (Toscano and York, 1992; Groot and Jordan, 1999; Wah, 2003). At UMES, depth to sulfidic materials within ditches ranged from 15 cm in DXXD3-6 to 151 cm in DX5-3 (Table 4). Prior to the construction of drainage ditches at UMES, the soils were poorly drained and existed presumably in an anaerobic environment. Anaerobic conditions in the soil preserved the iron-sulfides. Upon construction of the ditches and subsequent land drainage, the sulfidic materials were subjected to fluctuating redox environments, leading to the oxidation of pyrite, and the formation of ferrous sulfate and sulfuric acid. Sulfate can be transported with the soil solution or by wicking to the surface (Fanning et al., 2002).

Poultry litter has been applied as fertilizer at UMES for more than 30 years. A common manure application rate in this region is 6,726 kg ha^{-1} (moist) (F.J. Coale, personal communication, 2004). In 2001, mean S content of all poultry manure (moist) tested (with floor litter; n= 758) at the Maryland Cooperative Extension testing lab was 0.586% (Maryland Cooperative Extension, 2001). Assuming a poultry litter application rate of 6,726 kg ha^{-1} (moist) over 30 yr, with an S content of 0.586%, equals 39 kg ha^{-1}

yr^{-1} of S. Presumably over time, S could migrate towards ditches at UMES through shallow ground water due to the fact that SO_4^{2-} is an anion and is susceptible to leaching in soils. The erosion and transport of poultry litter after application to drainage ditches from adjacent fields may also occur. The direct input of poultry litter to drainage ditches during its application to fields also occurs due to the close proximity of the manure spreader to the ditch.

Chemical amendments such as alum ($Al_2(SO_4)_3$), gypsum ($CaSO_4$), and iron sulfate ($FeSO_4$) are currently being evaluated as a potential nutrient management strategy in the region to reduce the loss of soluble P in manure-amended soils (Arai et al., 2005; Johnson, 2003). Phosphorus is a leading contributor to the eutrophication of surface waters (Diaz and Rosenburg, 1995). Chemical amendments supply Al, Ca, and Fe minerals that can decrease the solubility of P in soils, hence the potential for loss in runoff (Johnson, 2003). However, the further addition of sulfate to ditch-drainage agroecosystems that already receive S through additions of poultry litter may increase rates of iron-monosulfide formation on the surface of drainage ditches.

Evidence of diffusion and bioturbation were found in ditch soils at UMES. The diffusion of ferrous iron through drainage ditch soils from an area of high concentration to an area of low concentration is evident in the high amount of redoximorphic features found in ditch soils. Possible sources of soluble ferrous iron may be from nearby shallow ground water inputs, and from drainage ditch soils themselves. Evidence of bioturbation by macro-fauna such as crawfish and other invertebrates was observed in all ditch soils.

Implications of Sulfides In Ditch Soils

The presence of iron-monosulfides and sulfidic materials at depth creates concerns for water quality in drainage ditches and downstream waterbodies. Iron-monosulfides are extremely labile and may oxidize in seconds to minutes when exposed to oxygenated water and agitated (Fanning et al., 2002). The oxidation of iron-monosulfides in water can strip nearly all dissolved O_2 out of the water column in minutes (Bush et al., 2004). The formation of monosulfides as a thick black ooze, also termed monosulfidic black ooze (MBO), has been documented in drainage ditches cut into sulfidic materials in areas of Australia (Bush et al., 2004; Smith, 2004).

In wetland and lake environments the presence of sulfides has been shown to have an affect on soluble-reactive P (Bridgham et al., 2001; Caraco et al., 1989, 1993; Curtis, 1989). Sulfides bind Fe, which results in the release of ferric iron-bound P into the water column (Bridgham et al., 2001; Caraco et al., 1989, 1993). Acid volatile sulfides measured in ditch surface soils at UMES were as high as .13%. The formation of iron-monosulfides in drainage ditches could be driving more P into solution while concurrently reducing the ditch soils capacity to act as a sink for P. Acid volatile sulfides may also be environmentally beneficial at concentrations as low as 0.02% due to their ability to bind significant amounts of heavy metals (van den Berg et al., 1998).

The excavation and subsequent exposure to oxygen of sulfidic materials and the disturbance of iron-monosulfides in drainage ditches can have potentially severe environmental consequences for surface waters originating from acid sulfate soil-affected areas (Smith, 2004; Fanning et al. 2002). The construction of drainage ditches in areas that contain sulfidic materials is a hazard that may lead to the oxidation of pyrite. Therefore, nutrient managers should closely examine soils for the presence of sulfidic materials prior to any management technique employed, especially clean-outs that involve the excavation of soil materials from drainage ditches by mechanical means. Failure to identify sulfidic materials may result in water quality problems downstream of the affected area. In areas that do contain deep sulfidic materials, deep clean-outs do not appear to be an advisable option due to the inherent risks of uncovering sulfidic materials and exposing them to an aerobic environment.

Fluvial Processes

Cumulization is the process whereby additions of material are made to the soil surface through hydrologic transport mechanisms (Fanning and Fanning, 1989). Addition of mineral material to ditches at UMES is the result of the erosion of finer-textured materials from adjacent cultivated fields, and the erosion of ditch banks. Precipitation of dissolved minerals such Fe from shallow ground water inputs may also occur. The total quantity of alluvial materials accumulated in drainage ditches varied considerably. The mean solum thickness in all drainage ditches was 27 cm. The maximum thickness of the mineral solum in all ten ditches was 78 cm (in profile DXXD2-6). Primary ditches contained the shallowest soils (mean=23 cm); the deepest soils were found in deep-collection ditches (mean=32 cm) (Table 4). The thickness of the solum may be dependent on factors such as the time from last clean-out, drainage water velocity, cultivation and management practices (e.g. vegetated buffer strips, cover crops, no-till), slope within the ditch and adjacent land, and erosion potential of adjacent field soils.

Ditch A horizon soil textures varied between ditch types. Primary ditch solum mean sand, silt, and clay contents were 47, 23, and 20% respectively. The shallow-collection ditch was higher in sand relative to primary ditches with a mean sand, silt, and clay contents of 71, 20, and 10%. Deep-collection ditches were similar to the shallow-collection ditch, with a mean sand content of 72% and a mean silt content of 18%. Differences in textures between ditch types (primary versus deep-collection and shallow-collection) are most likely the result of water velocity, and the relationship between the ditch depth and the textures of the substratum. Higher water velocities are generated in collection ditches as a result of increased water inputs during storm events from primary drainage ditches. These events may promote the likelihood of scouring and the suspension of fine mineral particles. Soils at UMES are underlain by a sandy substratum. When collection drainage ditches (> 1.5 m) are constructed, it results in the exposure of these sandy soils on their banks. These materials are not as exposed in the primary ditches. As erosion of collection ditch banks occurs, sandy-coarse materials are deposited onto ditch soils.

Taxonomic Classification of Ditch Soils

All ditch soil profiles described were classified to the suborder level as aquents. Endoaquents accounted for 70% of all profiles, with the subgroups being sulfic (20%), aeric (20%), hummaqueptic (20%), and typic (9%) (Table 8). Family particle-size classes were fine-loamy, fine-loamy over sandy or sandy skeletal, coarse-loamy, coarse-loamy over sandy or sandy skeletal, and sandy (Table 8).

Variability and Mapping

The primary geomorphic and morphological contrasts between ditch soils in the study were between soils formed in shallow (<1.5 m) primary and collection ditches, which tended to have structure and a layer in the substrata with a bright matrix color and soils formed in deep ditches (1.5 to 4 m), which tended to have high n-value, structureless solum, and gleyed substrata. Additional variables to consider when delineating ditch mapping units may be depth of recent alluvium and the presence or lack of presence of sulfidic materials within 1 m of the ditch soil surface (Table 7). A map showing the location of each profile description and the associated mapping unit is presented in figure 7. The delineation of ditch mapping units based on Soil Taxonomy was not useful due to the high variability of taxonomic differences between description sites.

CONCLUSIONS

Mineral and organic materials found within the vegetated ditches in this study should be conceptualized as soils so as to better understand them in terms of nutrient cycling, modeling, and mapping. Ditch soils are unique natural soil bodies that undergo pedogenesis and occur in both seasonally and permanently saturated ditch environments. We have shown that individual soil horizons can be differentiated and that identification of subsurface properties (e.g., sulfidic materials) may potentially impact soil characterizations.

The formation of iron-sulfide minerals near or at the surface of some drainage ditch soils may represent a potential threat to surface water quality in a region that contains a highly concentrated animal agriculture industry. Failure to identify materials that can affect overlying soil chemical properties and water quality, as in the case of sulfidic materials at UMES, may lead to misapplying ditch management practices such as mechanical clean-outs.

Future research of drainage ditches in this region should address the presence of sulfidic materials and the potential impact that iron-monosulfides have on water quality. Further research into the extent and distribution of sulfides within drainage ditches in this region is needed.

Table 1. Morphological horizon class names and descriptions. Each horizon class is a group of soil horizons with similar morphological characteristics. These classes were used to compare the means of groups of similar horizons.

Morphological Horizon Class	Description
Oi	Described in the field as fibric organic horizons (Oi). May be considered a mineral horizon when all coarse organic debris is removed.
Dark A	A horizons that are dark in color (value 3 or less, chroma 3 or less), and are enriched in organic carbon.
Gley A	A horizons that exhibit colors with a color value 4 or more, chroma 2 or less.
Bright C	C horizons with bright colors (value 5 or more, chroma 3 or more) and show signs of an oxidizing environment.
Oxidized C	C horizons that have bright colors (value 5 or more, chroma 4 or more). Appear to be contain oxidized forms of Fe and signs of an oxidizing environment.
Gley C	C horizons that exhibit gley colors with a hue of 2.5Y or 10 YR, value 4 or more, chroma 2 or less, and are sand to loamy sand in texture.
Surface C	C horizons with a texture of sand or gravelly sand throughout the entire profile and which lack an organically enriched surface horizon.
Sulfidic C	Sulfidic materials that have been identified through either an 8 week incubation period, acid volatile sulfur and chromium reducible sulfur analyses, or violent reaction to treatment with H_2O_2.

Table 2. Profile description of a typical drainage ditch soil located in a deep-collection ditch at the University of Maryland Eastern Shore Research Farm. This profile is a coarse-loamy, mesic, Humaqueptic Endoaquent.

Horizon	Depth	Description
	cm	
Oi	0-7	Black (10YR 2/1) Fibric materials
A1	7-16	Very dark brown (10YR 2/2) mucky silt loam (13% clay); no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= >1
Ag1	16-24	Dark grayish brown (10YR 4/2) sandy loam (9% clay) with common (5%) medium prominent yellowish red (5YR 4/6) concentrations as masses; no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= .7 to 1
Ag2	24-30	Dark grey (10YR 4/1) silt loam (10% clay) with few (2%) medium prominent yellowish red (5YR 4/6) concentrations as soft masses; no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= .7 to 1
A'2	30-38	Brown (10YR 4/3) sandy loam (10% clay); no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= <.7
A'3	38-47	Very dark grayish brown (10YR 3/2) loam (8% clay); no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= <.7
2Cg	47-68	Brownish grey (10YR 5/2) sandy loam (6% clay) with common distinct (10%) yellowish brown (10YR 5/8) concentrations as soft masses; no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= <.7
2C	68-87	Brown (10YR 5/3) coarse sand (3% clay); no structure; positive reaction to alpha'-alpha dipyridyl. *n* value= <.7

Table 3. Profile description of a typical drainage ditch soil located in a primary ditch at the University of Maryland Eastern Shore Research Farm. The classification of this profile is a coarse-loamy over sandy or sandy-skeletal, mesic, Humaqueptic Psammaquent.

Horizon	Depth	Description
	cm	
A1	0-2	Very dark brown (10YR 2/2) silt loam (11% clay); weak granular structure; very friable.
A2	2-8	Very dark grey (10YR 3/1) silt loam (12% clay); weak granular structure; very friable.
A3	8-20	Very dark grey (10YR 3/1) silty clay loam (29% clay) with few (2%) fine distinct strong brown (10YR 4/6) concentrations as porelinings; weak subangular blocky structure; very friable.
2Cg1	20-36	Light brownish grey (2.5Y 6/2) loamy sand (5% clay) with common (5%) medium prominent reddish yellow (10YR 6/8) concentrations as soft masses.
2C1	36-46	Olive yellow (10YR 6/6) sand (2% clay) with few (2%) medium faint strong brown (10YR 5/8) concentrations as soft masses; few (2%) fine medium distinct pinkish grey (10YR 7/2) depletions.
2C2	46-71	Pale yellow (2.5Y 7/3) sand (2% clay) with common (3%) medium prominent strong brown (10YR 5/8) concentrations as soft masses; common (3%) medium prominent white (2.5Y 8/1) depletions.
2C’g2	71-86	Light brownish grey (2.5Y 6/2) sand (2% clay) with few (1%) medium faint light brown (10YR 6/4) concentrations as soft masses.

Table 4. Depth of alluvium (A horizons), depth to the uppermost horizon containing sulfidic materials, and depth to the top of oxidized C horizon in each soil profile described in drainage ditches located at the University of Maryland Eastern Shore Research Farm.

	Depth of Alluvium				**Depth to Sulfidic Materials**				**Depth to Zone of Concentrations**			
Ditch	**Min**	**Max**	**Mean**	**n**	**Min**	**Max**	**Mean**	**n**	**Min**	**Max**	**Mean**	**n**
		cm				cm				cm		
DX1	16	32	26	8	82	89	86	2	32	90	51	4
DX2	15	24	20	8	76	98	87	2	55	63	59	3
DX3	7	24	16	8	28	76	52	3	18	29	24	2
DX5	13	41	26	4	48	151	100	2	85	85	85	1
DX6	23	56	36	6	30	82	56	5	0	0	0	0
DX7	11	58	31	7	45	64	55	3	89	89	89	1
DX8	27	27	27	1	51	51	51	1	0	0	0	0
DXXS1	1	35	17	8	47	84	66	3	19	64	38	6
DXXD2	21	78	46	10	44	44	44	1	62	62	62	1
DXXD3	0	50	20	9	15	78	47	3	0	0	0	0

Table 5. Chemical and physical characterization data for morphological horizon classes from samples collected at the University of Maryland Eastern Shore Research Farm.

					Particle Size Fractions		
Horizon Class		pH	OC	N	Sand	Silt	Clay
			%				
Oi	mean	5.3	4.0	0.6	48	34	18
(n= 4)	min	5.0	0.2	0.0	26	17	13
	max	5.7	7.6	1.3	70	51	23
	std.dev	0.4	3.6	0.6	31	24	7
Dark A	mean	4.9	5.1	0.6	54	28	18
(n= 38)	min	3.8	0.6	0.1	10	4	2
	max	6.1	12.4	1.5	95	79	4
	std.dev	0.6	2.8	0.4	25	18	11
Gley A	mean	4.7	3.3	0.5	61	26	13
(n= 19)	min	3.4	0.1	0.0	16	3	2
	max	6.0	9.6	1.8	95	62	43
	std.dev	0.8	2.6	0.5	23	15	10
Bright C	mean	4.8	0.2	0.0	90	5	5
(n= 12)	min	4.0	0.1	0.0	85	1	1
	max	5.6	0.4	0.1	95	13	9
	std.dev	0.6	0.1	0.0	4	3	2
Oxidized C	mean	4.6	0.2	0.1	87	8	5
(n= 6)	min	4.0	0.1	0.0	77	3	3
	max	5.5	0.2	0.1	93	16	8
	std.dev	0.6	0.1	0.0	7	5	2
Gley C	mean	4.3	0.3	0.1	87	8	5
(n=29)	min	2.6	0.0	0.0	71	0	1
	max	5.6	1.4	0.7	98	24	12
	std.dev	0.7	0.3	0.2	8	6	3
Surface C	mean	3.6	2.5	0.2	93	4	3
(n= 2)	min	3.3	2.4	0.2	92	3	2
	max	3.9	2.6	0.2	94	4	4
	std.dev	0.4	0.1	0.0	1	0	1
Sulfidic C	mean	4.6	0.4	0.1	79	11	10
(n= 10)	min	4.0	0.2	0.0	36	4	5
	max	5.2	0.6	0.4	91	40	24
	std.dev	0.4	0.2	0.1	16	11	6
Total	mean	4.7	2.4	0.3	72	17	11
(n=126)	min	2.6	0.0	0.0	10	0	1
	max	6.1	12.4	1.8	98	79	43
	std.dev	0.7	2.9	0.4	24	16	9

Table 6. The percent total acid volatile sulfur (AVS) and chromium reducible sulfur (CRS) in selected drainage ditch soil samples from the University of Maryland Eastern Shore Research Farm.

Ditch	**Distance from outlet**	**Depth**	**AVS**	**CRS**
	— m —	— cm —	——— %	———
DX1	40	0-5	0.037	0.030
DX2	40	0-5	0.087	0.039
DX2	40	28-50	0.0026	0.15
DX3	40	0-5	0.13	0.040
DX3	80	0-5	0.070	0.075
DX3	40	91-107	0.0039	0.20

Table 7. Description of preliminary map units developed for drainage ditch soils on the University of Maryland Eastern Shore Research Farm.

Map Unit Symbol	Map Unit Description
1	Ditch soils that contain less than 20 cm of alluvium.
1A	Ditch soils that contain less than 20 cm of alluvium and sulfidic materials within 1 m of the ditch soil surface.
2	Ditch soils that contain 20 to 30 cm of alluvium.
2A	Ditch soils that contain 20 to 30 cm of alluvium and sulfidic materials within 1 m of the ditch soil surface.
3	Ditch soils that contain 30 to 40 cm of alluvium.
3A	Ditch soils that contain 30 to 40 cm of alluvium and sulfidic materials within 1 m of the ditch soil surface
4	Ditch soils that contain greater than 40 cm of alluvium.
4A	Ditch soils that contain greater than 40 cm of alluvium and sulfidic materials within 1 m of the ditch soil surface.
5	Ditch soils that contain no alluvium.

Table 8. Family particle-size class and great group presented as the number of soil profiles described in each taxonomic property category per ditch from the University of Maryland Eastern Shore Research Farm.

					Drainage Ditch					
Taxonomic Properties	**DX1**	**DX2**	**DX3**	**DX5**	**DX6**	**DX7**	**DX8**	**DXXS1**	**DXXD2**	**DXXD3**
					Family Particle Size Class					
Coarse-loamy		3		1	2	3		5	6	4
Coarse-loamy over sandy or sandy skeletal	5	3	6	2	2		1		4	
Fine-Loamy	2		1		1					1
Fine-loamy over sandy or sandy skeletal	1	2		1	1	3		1		1
Sandy			1					2		3
					Great Group					
Sulfic Endoaquent	2	3	1		2	2	1	2		1
Aeric Endoaquent	3			3	1	1		2	3	1
Humaqueptic Endoaquent	1	3				2		1	4	3
Typic Endoaquent		2				1			2	1
Humaqueptic Psammaquent			6		1			1		1
Typic Psammaquent	2							1		1
Haplic Sulfaquent			1	1		1		1		1
Typic Sulfaquent					2				1	

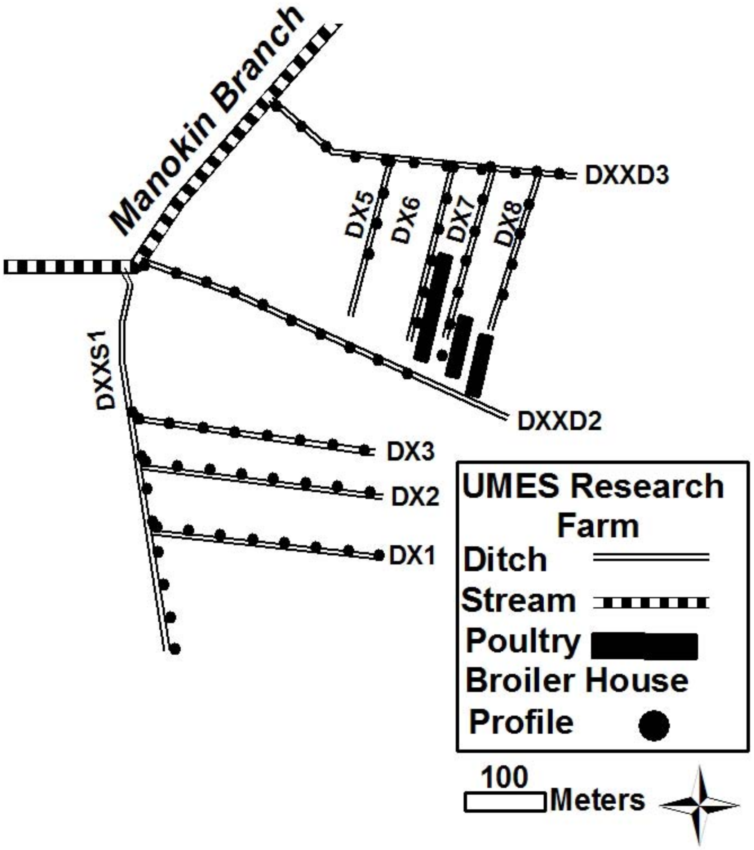


Figure 6. Overview map showing drainage ditch study area and soil profile description locations on the University of Maryland Eastern Shore Research Farm, Princess Anne, MD.

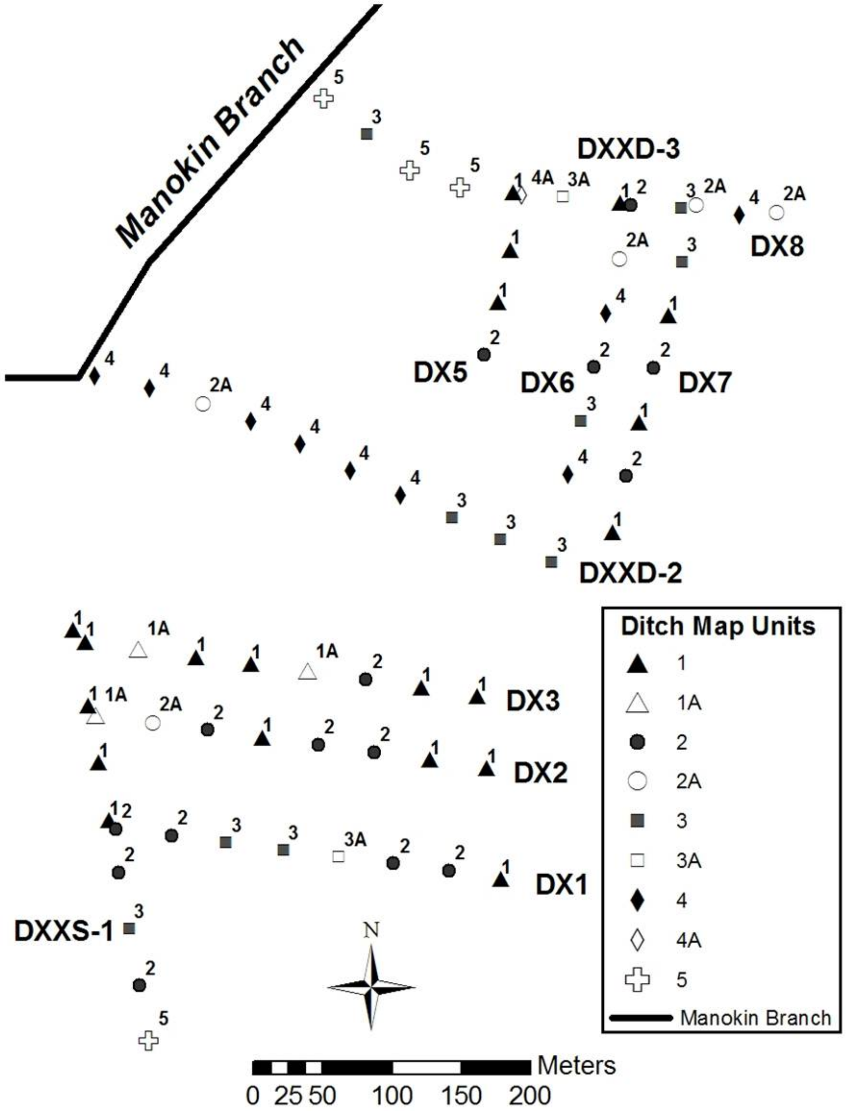


Figure 7. Map showing profile description sites and the map unit symbol within drainage ditch study area located on the University of Maryland Eastern Shore Research Farm, Princess Anne, MD. Ditch DX8 has only one profile description at the outlet of the ditch.

Chapter 4: Spatial Variation of Soil Phosphorus Within a Drainage Ditch Network

ABSTRACT

Agricultural drainage ditches serve as P transport pathways. The ditch soil-water interface may potentially be an important zone of interaction that is key to understanding P cycling in ditches. The spatial variation of soil P within drainage ditch networks has not been investigated. We conducted a study to quantify the spatial variation of surficial soil P within agricultural ditches on the University of Maryland Eastern Shore Research Farm in Princess Anne, Maryland. Soils (0-5 cm) from 10 drainage ditches were sampled at 10-m intervals and analyzed for acid ammonium oxalate-extractable P, Fe, Al (P_{ox}, Fe_{ox}, Al_{ox}) and pH. The P_{ox}, Fe_{ox}, $Al_{ox,}$ and pH variables were well autocorrelated spatially. Oxalate-P (min=135, max= 6919 mg kg^{-1}, mean= 700 mg kg^{-1}) exhibited a high standard deviation across the study area (overall 580 mg kg^{-1}) and within individual ditches (maximum 1383 mg kg^{-1}). Ditches contained distinct areas of high P_{ox}, which were associated with either point- or non-point P sources. Phosphorus was well correlated with Al_{ox} or Fe_{ox} within specific ditches, but Fe_{ox} (r=0.44; *P*=0.001) was not as well correlated as Al_{ox} (r= 0.80; *P*=0.001) with P_{ox} across all ditches. The spatial variation of soil P within the drainage network and the identification of areas particularly high in P_{ox} lead to important implications in the development of ditch sampling design, modeling, and future nutrient management strategies.

Abbreviations: P_{ox}, acid ammonium oxalate-extractable P; Fe_{ox}, acid ammonium oxalate-extractable Fe; Al_{ox}, acid ammonium oxalate-extractable Al; MBO, monosulfidic black ooze; DPS, percent degree of P saturation; UMES, University of Maryland Eastern Shore.

The eutrophication of both fresh and estuarine waters in the U.S. is a significant ecological and environmental problem. In the year 2002, 408 surface waters in Maryland were identified as impaired, 25% of those were the result of nutrients (USEPA, 2003). Degradation of water quality due to eutrophication can restrict recreational activity and commercial uses of water. The Chesapeake Bay is the largest estuary in the U.S. and has experienced the effects of eutrophication for longer than 30 years (Boesch et al., 2001). The ecological, economic, and social impacts of eutrophication are of increasing concern in the Chesapeake Bay watershed and throughout the U.S. (Boesch et al., 2001).

The southern Delmarva Peninsula has a relatively flat relief and is dominated by poorly drained soils. The water table in this region is close to or at the surface for extended periods of time during the year. To permit cultivation, the region relies on open-air drainage ditches to lower the ground water and quickly remove overland flow during periods of intense rainfall. Agricultural drainage networks are connected hydrologically to local streams and rivers and are a pathway for sediment and nutrients from agricultural ecosystems (Vadas and Sims, 1998).

The southern Delmarva Peninsula contains an intense poultry industry which produced more than 560 million broiler birds and more than 1.3 million Mg of chicken in 2004 (Delmarva Poultry Industry, 2005). Large quantities of poultry litter (poultry manure combined with woodchips, shavings or other bedding material) are produced each year on the Delmarva Peninsula, much of which is land applied as fertilizer for crops. Poultry litter has a low N to P ratio (Kleinman et al., 2005) such that application of litter at a rate suited to meet crop N requirements generally results in application of P well above that required by the crop. The continual application of poultry litter in excess of crop needs leads to the accumulation of P in soil and increased potential for P loss in runoff (Sharpley, 1999).

The development of land drainage systems for both public and agricultural use began in Mesopotamia around 9,000 B.P. (van Schilfgaarde, 1971). Around 2400 B.C., the Egyptians and the Greeks also developed surface-drainage system networks (Shirmohammadi et al., 1995). Organized drainage in the U.S. began in the 1600's (Evans et al., 1996). Many states in the Midwest rely heavily on drainage for agricultural proposes with nearly 37% of all cropland requiring drainage (Fausey et al., 1995). More than 40% of all cropland in North Carolina today requires artificial drainage (Thomas et al., 1995). Currently in Florida, there are more than 2.5 million ha of land requiring artificial land drainage (Thomas et al., 1995).

Ditches on the southern Delmarva Peninsula can be grouped based on their function, size, and hydrological properties (Vaughan, 2005). "Primary ditches" are open-air ditches located in cultivated fields that are shallow (<1.5 m), drain surface runoff and shallow subsurface flow, often contain stagnant water, and may dry-out periodically. "Collection ditches" transport outflow from primary ditches. Two types of collection ditches exist: shallow-collection and deep-collection ditches. Shallow-collection ditches (1.5 to 2 m) are connected to shallow ground water, and dry out during dry times of the year. Deep-collection ditches (>2 m) are connected to deep, regional ground water and contain water throughout most years.

Biogeochemical processes operating in riverine wetland soils are similar to processes occurring in some drainage ditch soils (Bowmer et al., 1994; Nguyen and Sukias, 2002). These processes include long periods of saturation and flooding, re-occurring oxidation-reduction cycles, mineralogical transformations, and the translocations of solutes such as ferrous Fe through the soil solution.

Ditch soils may act as both a sink and source of P (Sallade and Sims, 1997a; Sallade and Sims, 1997b; Nguyen and Sukias, 2002). Mechanisms that control this relationship include both sedimentation and re-suspension of organic matter and P-enriched soil particles, the sorption and desorption reactions of P in solution with mineral and organic compounds, and the uptake and release of P by plants and microorganisms (Johnston et al., 1997).

In acidic soils, such as those found in the Atlantic Coastal Plain, controls of P can be attributed to Fe and Al hydroxides and organic matter cycling (Vadas and Sims, 1998). Low redox potentials in drainage ditch soils develop during periods of warm weather and slow overlying water movement when the decomposition of organic matter is occurring (McCoy et al., 1999). Low redox potentials can lead to the dissolution of ferric (Fe^{3+}) iron and the subsequent release of Fe-bound P (Reddy et al., 1995; Vadas and Sims, 1998). Aluminum-bound P is unaffected by anoxic conditions (Darke and Walbridge, 2000).

The spatial variation of P in ditch soils has not previously been investigated. An understanding of the degree of variation (variance) and spatial patterns (autocorrelation) of soil P is necessary for sampling design, modeling, mass balance estimation, critical source area identification, management decisions, and understanding of basic transport and deposition processes (i.e. zones of sink, source, stability).

Our objectives were to 1) assess the spatial variation of oxalate-extractable P, Al, Fe, and pH from a depth of 0 to 5 cm within an agricultural drainage network and 2) examine the relationship between soil P variation, farm structure, and ditch properties.

MATERIALS AND METHODS

Site Selection

This study was conducted on the University of Maryland Eastern Shore (UMES) Research Farm (N 38° 12' 22", W 75° 40' 35") located in Princess Anne, Somerset County, Maryland (Fig. 6). Poultry litter as fertilizer has been applied to many areas of the farm for more than 30 years. For a more comprehensive description of the UMES farm, field and ditch soils, see Vaughan et al. (2005).

Drainage Ditch Selection

Drainage ditches at UMES were categorized as either a primary (<1.5 m), shallow-collection (1.5 to 2 m), or deep-collection (2 to 4 m) ditch (Vaughan et al., 2005). In Figure 6, drainage ditches were identified and labeled using the letter D, as well as the letter X for primary field ditches and XX for collection ditches. The letters S and D were used to distinguish between shallow, (i.e., S) and deep (i.e., D) collection ditches and a numerical identifier (e.g., 1, 2, 3…) was used to distinguish between individuals of the same type (e.g., DX1, DX2, DXXS1, DXXD1).

Ditch soil profiles are generally A horizons formed in recent alluvium over C horizons formed in original coastal plain sediments. Ditch A horizons are dark in color (mean value=3.3; chroma=1.8). Drainage ditch C horizons are lighter in color (mean value=5; chroma= 2). Ditch A horizons are loamy in texture while ditch C horizons are coarser in texture, and are dominated by very gravelly sands, gravelly sands, and sands. Soil structure is found in three-quarters of all A horizons; ditch C horizons are primarily structureless. Redoximorphic features such as iron concentrations are commonly found in ditch A horizons; while zones of iron depletion are less common. Greater than half of all drainage ditch C horizons contain redoximorphic features, which are found primarily as iron concentrations. Drainage ditch soils are generally acidic. Ditch horizons are enriched with organic C. Iron-monosulfides (FeS) are present on the surface of drainage ditches DX1, DX2, and DX3 when they are submerged for extended periods of time. Additionally, sulfidic materials that contain pyrite (FeS_2) are found at depth below most drainage ditches located on the farm. It is thought that the origin of this geologically deposited sulfidic material is due to a past marine transgression. The last marine transgression is thought to have occurred either 82,000 years B.P. or 125,000 years B.P. (Toscano and York, 1992; Groot and Jordan, 1999; Wah, 2003).

All ditch soil profiles described were classified to the suborder level as aquents. Endoaquents accounted for 61% of all profiles, with the subgroups being sulfic, aeric, and hummaqueptic. Particle size family classes were coarse-loamy or coarse-loamy over sandy or sandy-skeletal.

Field Methods

Soil samples were collected in the spring (March-April) of 2004. At the time of sampling, all drainage ditches contained >8 cm of water. Sampling sites within each ditch were located by measuring 10-m intervals using a wheeled measuring device in the field adjacent to each ditch, and then marking the sampling site within the ditch with either a flag or spray paint on the side of the ditch. Three cores (0-5 cm) from the bottom of each drainage ditch were extracted at each sampling site using a 7.6-cm open-face gouge auger in three evenly spaced distances perpendicular to the water flow direction in the ditch. At every third sampling location (30-m intervals) an additional composite sample was collected within 5 cm of the original sample. The cores were placed into plastic sampling bags and composited by hand. Samples were transported back to the laboratory at air temperature and air-dried. Upon returning to the laboratory and prior to air-drying, concentrations of what was presumably ferric iron were observed on the inside of the sealed sample bags. Coarse organic debris was removed and the sample was ground to pass a 2-mm sieve. All analyses were performed on crushed, air-dried samples. A total of 405 samples across all drainage ditches were collected and analyzed.

Laboratory Methods

All soil samples were analyzed for acid ammonium oxalate-extractable Al, Fe, P (Al_{ox}, Fe_{ox}, P_{ox}) that were extracted by 1:40 soil:(0.1M $(NH_4)_2C_2O_4 \cdot H_2O$ + 0.1M $H_2C_2O_4 \cdot 2H_2O$) that was adjusted to pH of 3, shaken in darkness for 4 hours, and measured by inductively coupled plasma atomic emission spectroscopy (ICP-AES) (Ross and Wang, 1993). Oxalate-extractable P was used rather than the commonly used

Mehlich 3-P as measure of soil P because of its ability to extract a greater proportion of P that is occluded by or tightly sorbed to Fe oxides that can form in soils with frequent oxidation-reduction cycles. Phosphorus that is occluded by Fe may become soluble under prolonged reducing conditions, therefore making acid ammonium oxalate-extractable P a better measure of P that can be potentially released to overlying drainage waters (Rhue and Harris, 1999). The degree of P saturation (DPS) was estimated from acid ammonium oxalate-extractable Al_{ox} $Fe_{ox,}$ and P_{ox} as :

$$DPS = (P_{ox}/[Al_{ox} + Fe_{ox}]) \text{ X } 100\% \qquad [1]$$

where $Al_{ox,}$ $Fe_{ox,}$ and P_{ox} are in mmol kg^{-1} (Breeuwsma and Silva, 1992). All samples were analyzed for pH (dried soil) at a soil to water ratio of 1:1 using a pH meter. These dry soil pH measurements may be lower than the pH of the soils in a moist condition because these soils may contain oxidizable sulfides. Previous investigations at UMES have revealed the presence of monosulfides at or near the surface of drainage ditch soils as well as sulfidic materials at depth; both of which are known to produce acidification upon oxidation (Vaughan, 2005).

Statistical Analyses

Statistical analyses were performed using S-Plus and S+ Spatial Stats (SAS Institute, Inc. 1990; Insightful Corporation, 2001). Statistical analyses were performed in two parts (1) all drainage ditches combined and (2) individual drainage ditches analyzed independently. Kolmogorov-Smirnov tests and descriptive statistics were used to assess normality. Only soil pH was found to be normally distributed; all other variables were found to be normal after log-transformation (Press et al., 1989). Correlations were performed by calculating regression coefficients presenting the relationship between the variables. Spatial autocorrelation was described using semivariance analysis (McBratney and Webster, 1986). A pooled variogram was generated for each variable by normalizing semivariance values for each ditch by the variance of that ditch. This was necessary due to the linear correlation between the mean and the variance (proportional effect). Semivariogram bins, or classes between point pairs, were set at 10-m increments to a maximum of 200 m. A minimum of 103 point pairs were present in all bins.

RESULTS AND DISCUSSION

Surficial Soil P Variation

Variograms

Pooled semivariograms indicated that all variables exhibited spatial autocorrelation at the scale of this survey (Fig. 8). Mean nugget semivariance values, calculated from adjacent samples, were lower than spatially separated mean values for all variables. In particular, P_{ox} and Al_{ox} exhibited high short-range variability. Excluding nugget values, Fe_{ox}, Al_{ox}, and P_{ox} exhibited a linear increase in semivariance with increasing distance. The semivariance of the 10-m bin in the pH and DPS semivariograms are low; the remaining bins increase linearly with lag spacing. The lack of an observed sill indicates that the range of spatial autocorrelation has not been reached.

The semivariograms support that the spatial patterns observed in the samples are real trends in the population.

Drainage ditch network

All drainage ditches were found to be high in P_{ox}, with a mean of 700 mg kg^{-1}, a median of 584 mg kg^{-1}, and a standard deviation of 580 mg kg^{-1} (Table 9). Drainage ditch soils (0-5 cm) studied in Delaware's Inland Bays' watershed total P (TP) concentrations ranged from 34 to 1285 mg kg^{-1} with a mean of 391 mg kg^{-1} (Sallade and Sims, 1997a). Oxalate-extractable Fe ranged from 431 to 7503 mg kg^{-1}, with a mean of 2041 mg kg^{-1}, a median of 1739 mg kg^{-1}, and a standard deviation of 1148 mg kg^{-1} (Table 9). Mean Fe_{ox} values of riparian wetlands in the Coastal Plain of North Carolina ranged from 2770 to 4540 mg kg^{-1} (Bruland and Richardson, 2004). Sallade and Sims (1997a) reported mean total Fe values of 1290 mg kg^{-1}. Acid ammonium oxalate-extractable Al_{ox} ranged from 239 to 13496 mg kg^{-1}, with a mean of 1128 mg kg^{-1}, a median of 929 mg kg^{-1}, and a standard deviation of 1185 mg kg^{-1} (Table 9). Mean total Al concentrations of 1815 mg kg^{-1} were reported by Sallade and Sims (1997a). The DPS ranged from 9 to 48%, with a mean of 22%, a median of 21%, and a standard deviation of 7% (Table 10). Soil pH ranged from 3.4 to 6.3, with a mean of 4.9, and a median of 4.9, and a standard deviation of 0.4 (Table 10).

Identification of Areas of High P

There were distinct ditch reaches of low and high P_{ox} in the study area relative to other reaches within the drainage network. These areas are identified in figure 9 using uppercase letters (i.e. A, B, C).

Area A and B

Area A encompasses all of ditch DX8 and is consistently high in P_{ox} (Fig. 9). The mean P_{ox} of DX8 was relatively high (mean= 862 mg kg^{-1}) and consistent throughout the entire length of the ditch (C.V.=34.6%). Mean Fe_{ox} of DX8 was relatively low as compared to all other drainage ditches (Fig. 10). Oxalate-extractable Al_{ox} in area A was moderate (Fig. 11). The percent degree of P saturation (DPS) in DX8 was very high thought the entire ditch (Fig. 12). Mean soil pH in DX8 was slightly acidic (Fig. 13), but was higher than most other ditch soils.

One reason for the high P_{ox} levels in ditch DX8 is the presence of a poultry manure storage shed 25 m to the west of the head of the ditch (Fig. 6). The shed was used to store poultry manure throughout the year until its application to adjacent fields and other farms during the spring planting season. The shed has a large footprint around it, so that farm equipment can operate in and around the structure. Due to the surrounding topography, it is likely that nutrient-rich runoff from this shed is directed to ditch DX8 and area A.

In area A, a moderate correlation was observed between P_{ox} and Fe_{ox} (r=0.65**) and a strong relationship between P_{ox} and Al_{ox} (r=0.84***) (Table 11). These correlations are important for understanding possible mechanisms of P_{ox} storage in this area. Under acidic soil conditions, Fe and Al-bound P would be expected to be the dominate fractions of P (Reddy et al., 1995). High DPS values (DPS> 30%) indicate that soils in area A (DX8) are highly saturated with P (Kleinman et al., 1999; Butler and Coale, 2005).

Area B comprises the eastern 100 m of the head of deep-collection ditch DXXD2. Area B contained the highest P_{ox} concentration (6919 mg kg^{-1}) found within the study area as well as the largest coefficient of variation of any ditch (C.V.=122.5 %). The mean P_{ox} of this 100 m zone at the head of DXXD2 was 2617 mg kg^{-1}. Oxalate-extractable Fe was moderate while Al_{ox} was very high relative to other areas (Fig. 11). The DPS was also very high (Fig. 12). Soil pH was varied; with one-third (30 m) containing small areas of low pH (3.9-4.3) and other small areas of higher pH (Fig. 13). Overall, the mean of pH values of soils in area B (5.05) was low, a value common in unammended soils of the Atlantic Coastal Plain (Vadas and Sims, 1998).

As with Area A, P_{ox} in Area B was likely affected by an adjacent farm structure, or point-source of P. In this case, a poultry broiler house (operational until spring of 2004) lies to the north in close proximity (Fig. 6). A door where poultry manure is removed and where harvested chickens are brought out is no more than 15 meters to the north. During heavy rain events, it has been observed that poultry manure was being carried in runoff through a gully and into area B. Thus, direct inputs of poultry manure into area B through runoff could account for the high concentrations of P_{ox}.

As was the case of area A, a significant relationship between Fe_{ox} and P_{ox} (r=0.66***) and a significant relationship between Al_{ox} and P_{ox} (r=0.98***) was observed in DXXD2. Soils in area B had very high DPS values, which might indicate again that sorption sites are becoming saturated with P.

There is a rapid transition from area B in ditch DXXD2 that is high in P_{ox}, Fe_{ox}, and Al_{ox}, to an area low in P_{ox}, Fe_{ox}, and Al_{ox} when compared to area B (Fig. 9-11). The lack of high P_{ox} observations near the outlet of DXXD2 suggests that the source of P_{ox} in DXXD2 is not the result of variations in P application rates to adjacent fields. Losses of P in drainage waters from DXXD2, especially from area B, are of particular concern due to the high DPS values throughout the length of the ditch (Fig. 12).

Area C, D, and E

Area C is located within the first 150 m of ditch DX3 starting from the outlet. Area C was high in P_{ox} and Fe_{ox}, but moderate in Al_{ox} and DPS when compared to other study areas within the drainage network. Soil pH in area C was considerably lower than other areas in the study area.

High P_{ox} values in area C do not appear to be the result of farm structure or point-sources of P as was the case with areas A and B. This portion of the farm does not contain any poultry broiler houses or manure sheds (Fig. 6). The high P_{ox} concentrations in this area may be a result of pedogenic mechanisms. This area is high in Fe_{ox} relative to other ditch areas (Fig. 10). P is most commonly found associated with Fe in acidic coastal plain ecosystems, therefore the high concentrations of poorly crystalline Fe may be acting as a source of binding sites to retain P_{ox}.

The cause of such high concentrations of Fe_{ox} in areas C, D, and E relative to areas A and B may be due to the oxidation of sulfidic materials found at depth in this area of the farm and the oxidation of iron-monosulfides found at the soil surface (FeS). These surficial iron-monosulfides are referred to as monosulfidic black oozes (MBO) (Smith, 2004). Sulfidic materials containing pyrite (FeS_2) at depth below drainage ditches as well as the formation of MBOs on the surface of drainage ditches have been documented in the drainage ditches at UMES (Vaughan, 2005). Sulfidic materials that contain pyrite (FeS_2) can be oxidized by lowered water tables due to drainage ditches, and during drainage ditch clean-outs (Smith, 2004). The oxidation of sulfidic materials containing pyrite (FeS_2) can produce significant quantities of Fe in a ferrous (Fe^{2+}) form which can be converted to a ferric (Fe^{3+}) form upon exposure to oxygen (Fanning et al., 2002). This process also releases appreciable amounts of sulfuric acid (Fanning et al., 2002). The production of sulfuric acid through pyrite oxidation could explain why the lowest mean soil pH (4.73) of any ditch was observed in DX3. The oxidation of MBOs can also lower soil pH (Smith, 2004).

Area D is located within the first 200 m of DX2 starting from the outlet, and area E is located within the first 150 m starting from the outlet of ditch DX1. Area D and E are very similar to area C in terms of their P_{ox}, Fe_{ox}, Al_{ox}, and pH characteristics. Area D and E contained high concentrations of P_{ox} (Fig. 9) as well as Fe_{ox} (Fig. 10). Only moderate concentrations of Al_{ox} were found in area D and E (Fig. 11). Similarly, DPS and pH in these areas was low to moderate (Fig. 11-12).

The additional Fe_{ox} produced by the oxidation of sulfidic materials may be having a favorable effect on controlling P losses in ditch DXXS1. This ditch receives water from DX1, DX2, and DX3 (Fig. 6) and is enriched with Fe_{ox} due to underlying sulfidic materials, but is not enriched with P_{ox} relative to DX1, DX2, and DX3. The enrichment of Fe_{ox} in DX1, DX2, and DX3 at the outlets may be buffering losses of P to DXXS1.

Implications of P Variation in Ditch Soils

If the spatial variation of soil P found at UMES is representative of ditches in agroecosystems with a history of manure application and intensive animal agriculture, it has important implications for the management and understanding of P losses. Traditionally, drainage ditch management has focused on clean-outs and woody vegetation control to maintain hydrologic function. New management practices are being developed to maintain and improve hydrologic function while increasing nutrient retention and denitrification (Evans et al., 1995). Currently these management practices are applied without sampling or characterization of ditch soils and their geomorphic environment. Knowledge of the spatial variation of P within drainage ditches may allow for more precise implementation of management techniques such as clean-outs. Areas within the drainage network identified as containing high P concentrations or high DPS values could be selectively targeted for clean-outs.

If sampling strategies for ditch soil P were to be developed, sampling design decisions would include point versus compositing, number of samples, sampling depth, P and other analyses, and full-ditch versus zonal sampling. Sampling strategies and data interpretation developed to understand P loss potential from field soils may not apply to ditches due to redox fluctuations, high organic matter content, and the different hydrology of ditches. There are currently no models available to estimate P transport processes within and through ditch soils.

In order to manage and model P losses from ditches, an improved understanding is needed of P transport pathways from the landscape to and through ditch soils, P retention processes in ditch soils, and direct P losses from ditch soils to overlying waters.

Areas of very high P and DPS (e.g., area A and B) may be critical source areas of P to downstream water bodies. Targeting of critical source areas with best management practices may yield large improvements in water quality. Areas of lower P and DPS in the ditch network (e.g., ditch DXXD3) may sorb P from these areas, providing a natural mitigation mechanism. However, ditch soils may also serve primarily as P sinks, in which case sampling and management should be performed to maximize this role.

At UMES ditches DXXD2, DX1, and DX3 exhibited substantial within-ditch zonation; in each case division of the ditch into thirds would have adequately captured this variation. The zonal nature of the variation of these ditches seems to have been caused by either local farm structure (DXXD2) or geomorphic setting (DX1 and DX3). It may not be necessary to sample ditches in zones that are unaffected by locally variable factors, but this would require methodology to consistently predict which ditches have substantial within-ditch variation.

CONCLUSIONS

Agricultural drainage ditches may operate as key sources and pathways for the transport of environmentally significant quantities of P from watersheds with intensive agriculture. Our results show that drainage ditch surface soils can accumulate very high concentrations of P_{ox}. Within the drainage network, P_{ox} had a high variance (overall standard deviation 580 mg kg^{-1}) and distinct low and high areas. Spatial variation of soil P within the drainage network may be important for the understanding and management of P losses from these systems. Future investigations of drainage ditches need to address the possible mechanisms for the linkages of farm structure P and Al, and geomorphic variation P with Fe.

Table 9. Summary characterization data of drainage ditch soils (0-5 cm) collected from the University of Maryland Eastern Shore Research Farm, Princess Anne, MD.

		DX1	DX2	DX3	DX5	DX6	DX7	DX8	DXXS1	DXXD2	DXXD3	Total†
	n	40	41	44	31	33	35	26	46	55	54	405
P‡	Min	348	510	225	364	208	269	227	144	135	161	135
	Max	1959	1359	1943	1505	811	1790	1635	1192	6919	1329	6919
	Mean	794	794	715	651	534	588	862	551	1085	407	700
	CV,%	45	22	50	35	24	43	35	46	123	57	83
	Std.	353	173	356	227	128	255	298	253	1329	234	580
Fe‡	Min	1089	1042	662	572	811	744	791	957	472	431	431
	Max	7504	4536	7070	6306	2289	2827	1795	5715	4664	7026	7504
	Mean	2935	2820	2240	2075	1411	1329	1349	2350	1750	1821	2041
	CV,%	50	31	62	55	25	33	23	36	52	74	56
	Std.	1457	872	1383	1142	350	432	307	834	905	1346	1148
Al‡	Min	612	597	560	505	514	381	571	367	239	309	239
	Max	1445	2032	13496	2203	1348	1890	2091	1400	13042	1711	13496
	Mean	976	1260	1725	805	900	1055	968	768	1678	850	1128
	CV,%	20	28	118	46	26	30	31	33	144	42	105
	Std.	198	350	2028	366	234	311	296	253	2408	357	1185

† All drainage ditch data combined.

‡ Acid ammonium oxalate-extractable P, Fe, and Al (mg kg^{-1})

Table 10. Summary characterization data of drainage ditch soils (0-5 cm) collected from the University of Maryland Eastern Shore Research Farm, Princess Anne, MD.

		DX1	DX2	DX3	DX5	DX6	DX7	DX8	DXXS1	DXXD2	DXXD3	Total†
	n	40	41	44	31	33	35	26	46	55	54	405
DPS‡	Min	44	38	18	41	30	34	41	22	36	27	18
	Max	77	66	59	91	74	110	108	80	96	72	110
	Mean	56	53	47	65	59	60	90	49	67	41	57
	CV,%	16	12	16	17	15	24	19	23	22	17	28
	Std.	9	6	7	11	9	14	17	11	15	7	16
pH	Min	4.2	4.5	4.0	4.4	3.4	4.2	5.0	4.3	3.8	3.8	3.4
	Max	5.0	5.3	6.3	5.6	5.2	5.9	5.7	5.6	5.8	5.6	6.3
	Mean	4.7	4.8	4.7	5.2	4.5	5.1	5.4	4.9	5.2	4.8	4.9
	CV,%	5	4	9	6	11	7	3	6	10	8	9
	Std.	0.2	0.2	0.4	0.3	0.5	0.4	0.2	0.3	0.5	0.4	0.4

† All drainage ditch data combined.

‡Expressed as a %. Percent degree of P saturation (DPS=$P_{ox}/(Fe_{ox} + Al_{ox})$)

Table 11. Correlation coefficients of drainage ditch soils (0-5 cm) by ditch and all ditches combined collected from drainage ditches at the University of Maryland Eastern Shore Research Farm, Princess Anne, MD .

Ditch	pH-Al	pH-Fe	pH-P	pH-DPS	Al-Fe	Al-P	Al-DPS	Fe-P	Fe-DPS	DPS-P
DX1 (n=40)	-0.04	0.09	0.17	0.25	-0.19	-0.09	-0.39*	0.98***	0.82***	0.86***
DX2 (n=41)	-0.07	-0.40*	-0.20	0.34*	0.02	0.50**	-0.27	0.68***	-0.18	0.25
DX3 (n=44)	0.60***	-0.30	0.13	-0.44*	0.00	0.62***	-0.57***	0.59***	0.01	-0.04
DX5 (n=31)	-0.55*	-0.19	-0.09	0.53*	0.78***	0.78***	-0.37*	0.87***	-0.42*	0.03
DX6 (n=33)	-0.81***	-0.56**	-0.67***	0.13	0.65***	0.75***	-0.29	0.81***	-0.05	0.33
DX7 (n=35)	-0.40*	-0.18	-0.15	0.17	0.23	0.48*	-0.11	0.76***	0.46*	0.75***
DX8 (n=26)	0.14	0.08	0.15	0.21	0.25	0.84***	0.47*	0.65**	0.062**	0.83***
DXXS1 (n=46)	0.10	0.00	0.15	0.25	0.64***	0.84***	0.45**	0.73***	0.07	0.79***
DXXD2 (n=55)	-0.40*	-0.31*	-0.36*	0.03	0.56***	0.98***	0.48**	0.66***	0.50***	0.59***
DXXD3 (n=54)	-0.43**	-0.66***	-0.55***	0.23	0.50***	0.77***	0.17***	0.88***	-0.05	0.31*
Total † (n=405)	-0.5	-0.30***	-0.07	0.03***	0.18**	0.80***	0.09	0.44***	-0.25	0.44***

*, **, *** Significant at the 0.05, 0.01, and 0.001 probability levels, respectively.

† All drainage ditch data combined.

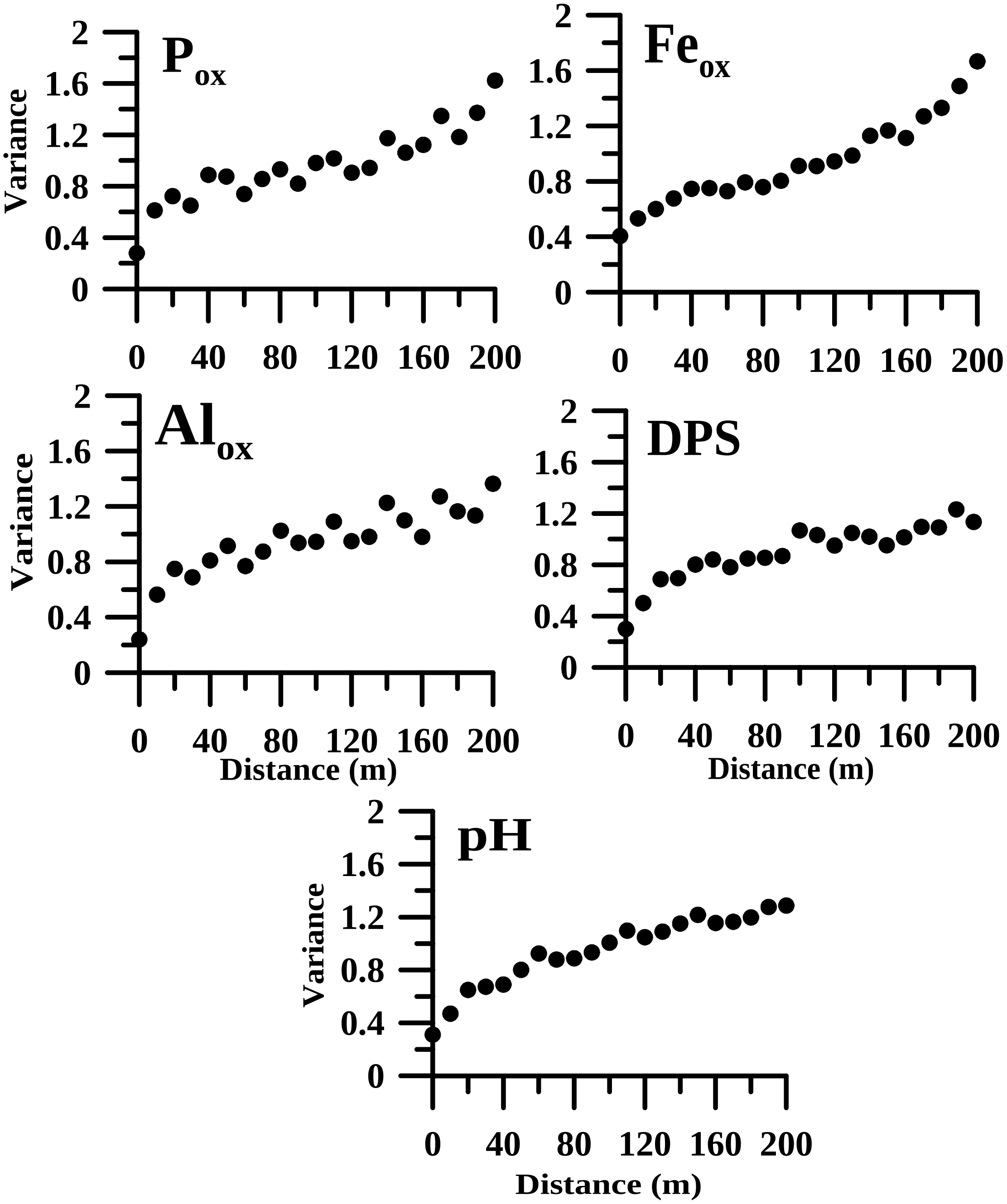


Figure 8. Pooled semivariograms of drainage ditch soil ammonium oxalate-extractable Fe_{ox}, Al_{ox}, P_{ox}. Percent degree of P saturation (DPS= P_{ox}/(Fe_{ox} + Al_{ox}) and pH are also presented.

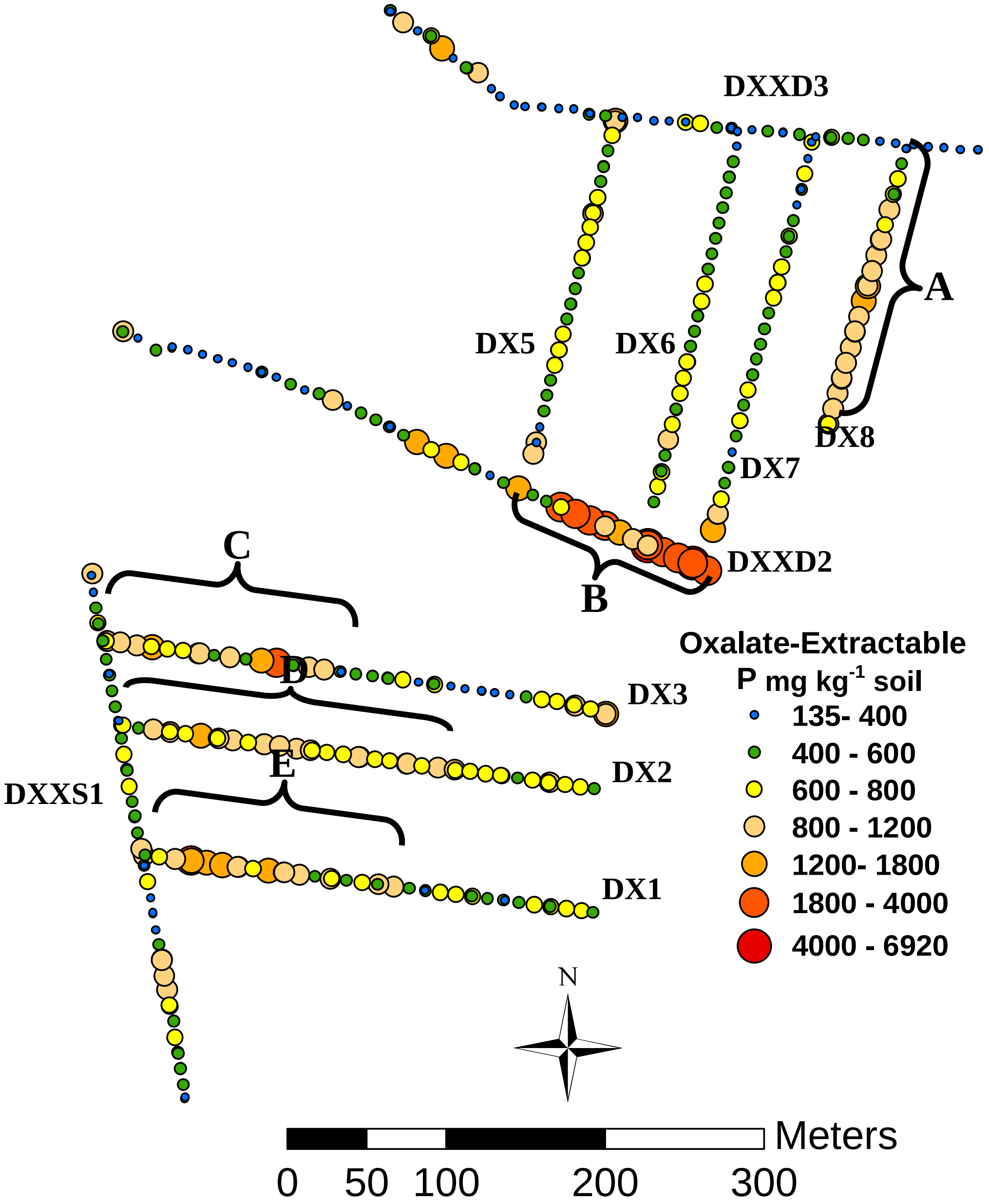


Figure 9. Map shows soil oxalate-extractable P distribution within a drainage ditch network.

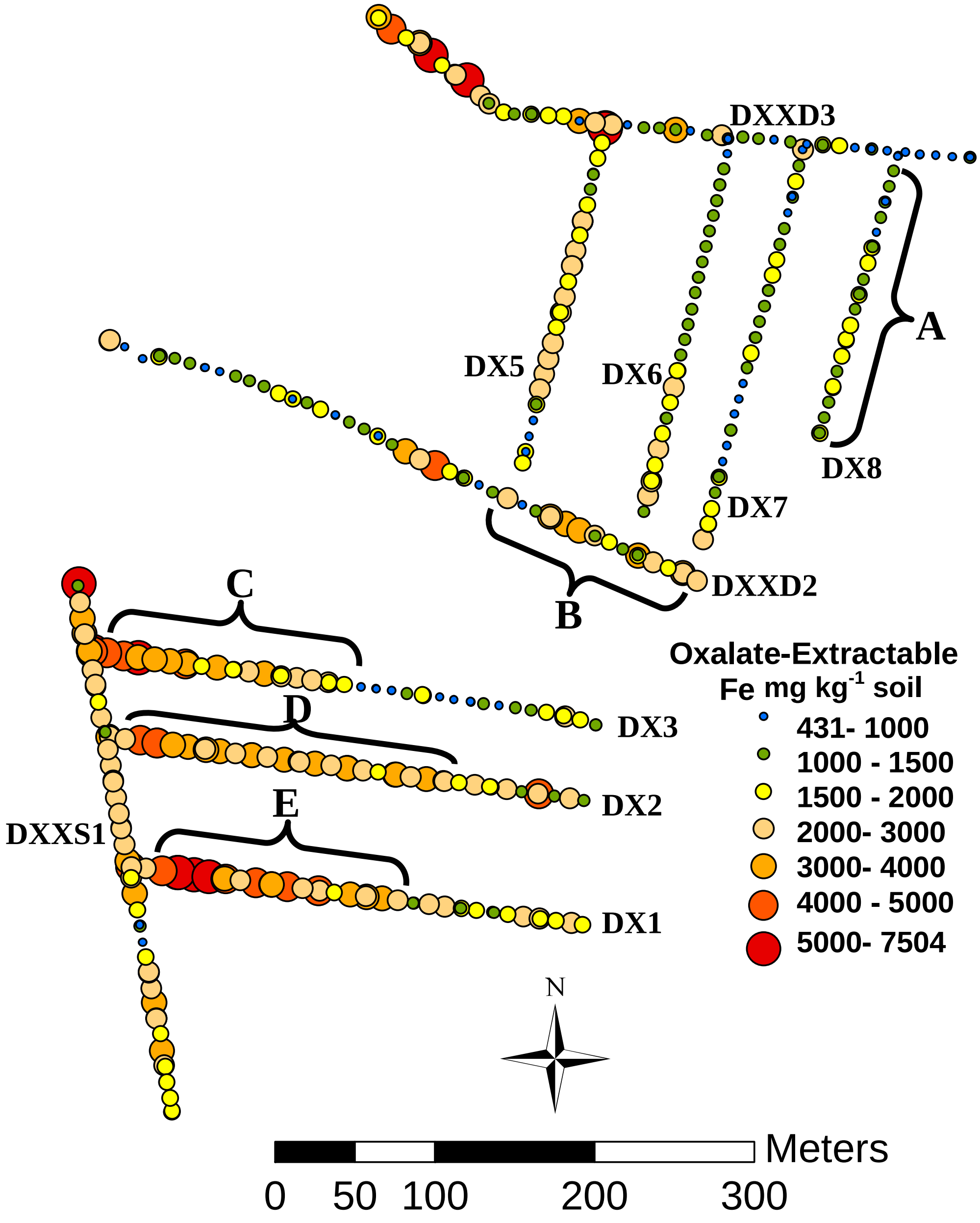


Figure 10. Map shows soil oxalate-extractable Fe distribution within a drainage ditch network.

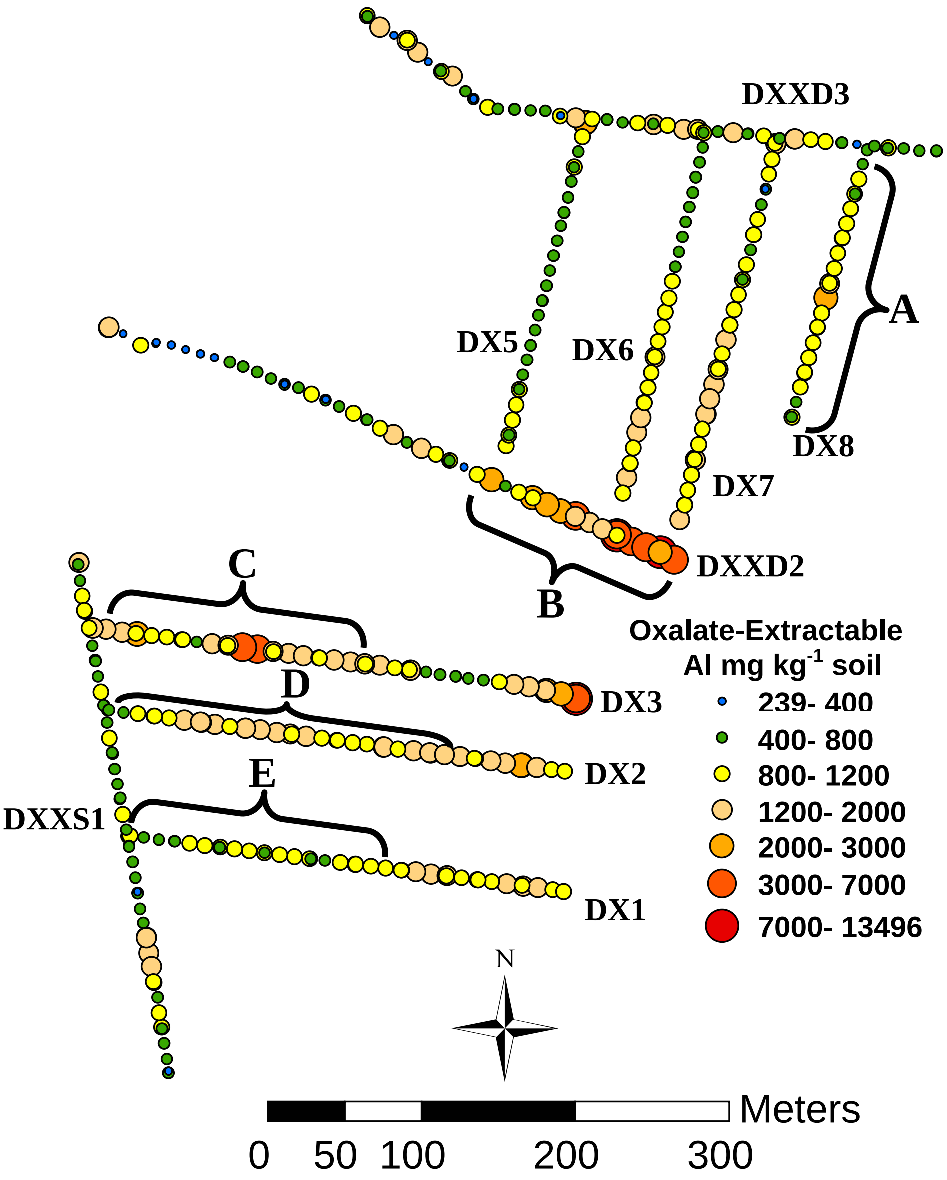


Figure 11. Map shows soil oxalate-extractable Al distribution within a drainage ditch network.

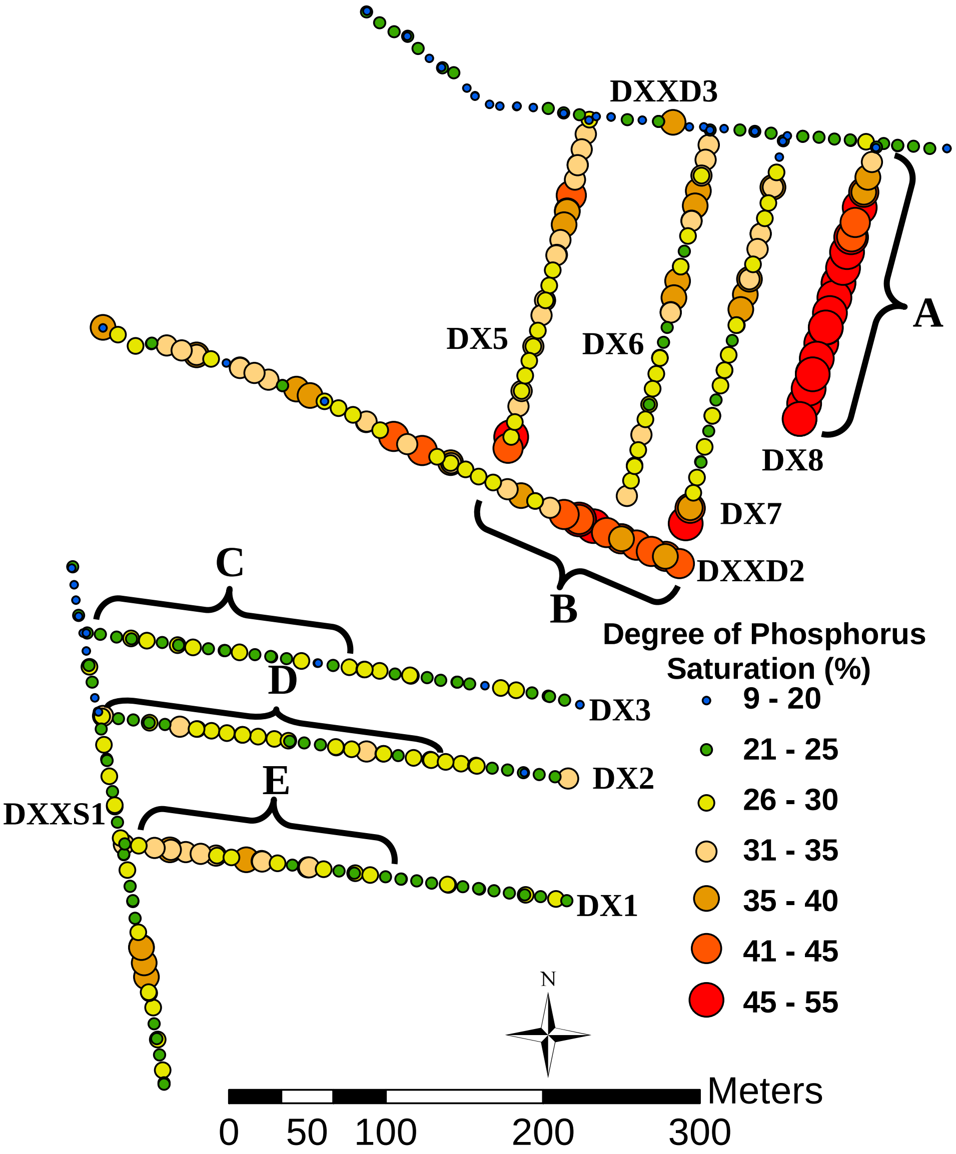


Figure 12. Map shows percent degree P saturation distribution within a drainage ditch network.

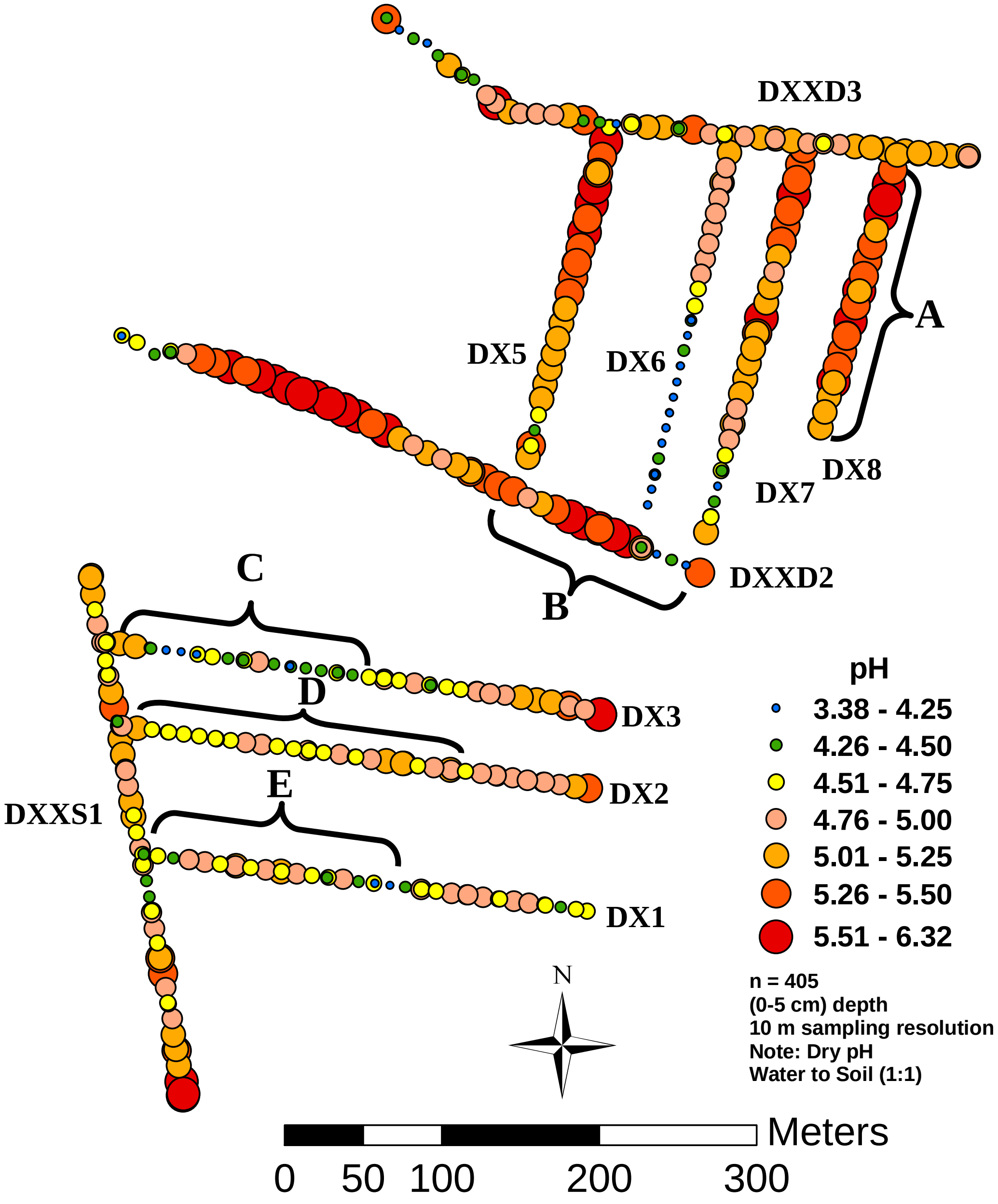


Figure 13. Map shows soil pH distribution within a drainage ditch network.

Chapter 5: Vertical Distribution of Phosphorus in Agricultural Drainage Ditch Soils

ABSTRACT

Pedological processes such as gleization, organic matter accumulation and decomposition, sulfidization, and sulfuricization may affect the vertical distribution of P within agricultural drainage ditch soils. The objective of this study was to assess the vertical distribution of ditch soil P as a function of depth and horizonation in ditch soils at the University of Maryland Eastern Shore Research Farm in Princess Anne, Maryland. Twenty-one profiles were sampled from 10 agricultural drainage ditches ranging in length from 225 to 550 m. Horizon samples were analyzed for total P, water-extractable P, Mehlich-3 P, acid ammonium oxalate-extractable P, Fe, and Al (P_{ox}, Fe_{ox}, Al_{ox}), pH, and organic C (n = 126). Total P ranged from 27 to 4882 mg kg^{-1}, oxalate-extractable phosphorus from 4 to 4631 mg kg^{-1}, Mehlich-3 P from 2 to 401 mg kg^{-1}, and water-extractable P from 0 to 17 mg kg^{-1}. While decreases in P with depth were observed, soil forming processes that result in pedological differences between horizons had the most significant effects on P fraction concentrations and sorption capacity. Organic horizons had the greatest P_{ox}, Fe_{ox}, and Al_{ox} concentrations, while dark A horizons were greater than gleyed A horizons (Dark A:Gley A; P_{ox}= 2.0, Fe_{ox}= 2.7, Al_{ox}= 1.5). Alluvial A horizons were greater in P_{ox}, Fe_{ox}, and Al_{ox} than subsurface C horizons (A:C; P_{ox}= 12.6, Fe_{ox}= 5.5, Al_{ox}= 3.8). Variation in P due to pedological differences between horizons may be essential for the understanding of short- and long-term P cycling, transport, and retention in ditch soils.

Abbreviations: TP, total P; WEP, water-extractable P; M3P, Mehlich 3-P; P_{ox}, acid ammonium oxalate-extractable P; Fe_{ox}, acid ammonium oxalate-extractable Fe; Al_{ox}, acid ammonium oxalate-extractable Al; DPS, percent degree of P saturation; MBO, monosulfidic black ooze; UMES, University of Maryland Eastern Shore.

The degradation of both fresh and estuarine waters in the U.S. as a result of eutrophication has lead to significant ecological and environmental problems. In the year 2000, 11% of 22,000 surface waters identified as impaired by the USEPA were the result of agricultural N and P (USEPA, 2003). The largest estuary in the U.S., the Chesapeake Bay, has experienced the effects of eutrophication for more than 30 years (Boesch et al., 2001). The study of eutrophication and its effects in the Chesapeake Bay has been unequaled in any other coastal ecosystem (Boesch et al., 2001). However, the ecological, economic, and social impacts of eutrophication continue to be an increasing concern in the Chesapeake Bay watershed and throughout the U.S. (Boesch et al., 2001).

Soils in humid regions with poorly drained soils require land drainage systems for profitable agricultural production (Shirmohammadi et al., 1995; Janse and Van Puijenbroek, 1998). Land drainage uses a system of open-air drainage ditches, subsurface drains (e.g., tile drainage), or a combination of both to lower the water table and speed the removal of excess surface runoff to local streams and waterbodies. Open-air drainage ditches are a hydrological link between surface runoff, ground water, and surface waters (Janse and Van Puijenbroek, 1998). Ditches have the potential to act as key pathways for the export of nutrients from areas of intensive agriculture to surface waters (Sallade and Sims, 1997a; Vadas and Sims, 1998; Nguyen and Sukias, 2002).

Throughout the world, land drainage has been adopted for multiple uses. Land drainage systems are thought to have been first developed 9,000 yr. in Mesopotamia and by the Egyptians and Greeks (van Schilfgaarde, 1971; Shirmohammadi et al., 1995). In the U.S., organized drainage began around the 1600's (Evans et al., 1996). Throughout the U.S. from 1900 to 1985, the installation and use of surface and subsurface drainage for agricultural purposes increased (Pavelis, 1987; Shirmohammadi et al., 1992; Shirmohammadi et al., 1995). The increase in land drainage has led to thousands of miles of open-air drainage ditches around the country. Currently, many states rely on land drainage, in particular subsurface drainage (i.e. tile drainage) and surface drainage (i.e. open-air ditches), to control ground water levels in both agricultural and urban areas. Thirty-seven percent of farmable land in the Midwest U.S. rely heavily on drainage for agricultural proposes (Fausey et al., 1995). Drainage ditches in the coastal plain of North Carolina have drained roughly 800,000 ha (Evans et al., 1996). Approximately 2.5 million ha of land in Florida are affected by artificial land drainage (Thomas et al., 1995).

The poultry industry of the southern Delmarva Peninsula produced more than 560 million broiler birds and more than 1.3 million Mg of chicken in 2004 (Delmarva Poultry Industry, 2005). As a result, vast quantities of poultry litter (poultry manure and woodchips or shavings) are produced each year on the Delmarva Peninsula, much is land applied as fertilizer for crops. Poultry litter has a low N to P ratio, and is commonly applied in excess of crop P requirements (Sharpley, 1999). The continual application of poultry litter in excess of crop needs leads to surplus soil P (Sharpley, 1999).

Investigations of non-point source P loss from agricultural watersheds and the eutrophication of surface waters have focused primarily on surface erosion and runoff

(Sims et al., 1998). The potential for P loss through subsurface runoff to local surface waters is most often underestimated (Sims et al., 1998). The most significant instances of downward movement of P through the soil profile have been through the accumulation of excessive amounts of P in agricultural systems involving the continuous application of manure and fertilizer (Sims et al., 1998). Several studies in regions with intensive animal agricultural production have shown the potential for subsurface soil P leaching and losses to shallow ground water in field soils. Mozaffari and Sims (1994) found that environmentally significant quantities of P had leached to depths near 75 cm in soils of a Delaware watershed where frequent applications of poultry litter as fertilizer were common. Phosphorus reaching shallow ground water may move laterally to drainage ditches between and during storm events and may constitute a significant transport pathway in these systems. The interaction of this P-laden ground water with subsurface ditch soil horizons may influence surficial water quality.

Drainage ditch soils may play a critical role in the water quality of the overlying water. Ditch soils may act both sinks and sources of P (Sallade and Sims, 1997a; Sallade and Sims, 1997b; Nguyen and Sukias, 2002). Mechanisms that control this relationship include both sedimentation and re-suspension of organic matter and P-enriched soil particles, sorption of P onto metal oxides such as Fe and Al in acidic soils and Ca in alkaline soils, oxidation-reduction (redox) potentials, and biotic uptake by plants and microorganisms (Johnston et al., 1997). Due to the acidic nature of soils found in the Atlantic Coastal Plain, drainage ditch soil P is primarily found sorbed to or occluded by Fe and Al hydroxides or as organic P (Vadas and Sims, 1998). Low redox potentials in drainage ditch soils develop upon organic matter decomposition during periods of warm weather and slow overlying water movement (McCoy et al., 1999). Low redox potentials can lead to the dissolution of ferric (Fe^{3+}) iron and the subsequent release of Fe-bound P (Reddy et al., 1995; Vadas and Sims, 1998). However, prolonged highly reducing conditions may lead to the precipitation of the ferrous phosphate mineral vivianite ($Fe_3(PO_4)_2 \bullet 8H_20$), which has been reported in to occur in both soils and sediments (Lindsay et al., 1989; Harris et al., 1994; Harris, 2002)

The focus of recent investigations regarding P in soils of open-air ditches has been limited to surficial (0-15 cm) soils (Sallade and Sims, 1997a; Sallade and Sims, 1997b; Nguyen and Sukias, 2002). These studies did not attempt to address the role of deeper soil horizons in P transport and retention. Drainage ditch surficial soils (0-15 cm) are the most likely to interact chemically with overlying drainage waters. However, shallow lateral subsurface flow pathways may bring storm flow into contact with deeper ditch soil horizons and inter-event diffusion and pedoturbation processes may mix P between surficial and deeper layers.

The objective of this study was to examine the vertical distribution of P fractions, ammonium-oxalate Fe and Al (Fe_{ox} and Al_{ox}), organic C, and pH in ditch soils as a function of depth and of ditch soil morphology.

MATERIALS and METHODS

Study Area

This study was located in Princess Anne, Somerset County, Maryland on the University of Maryland Eastern Shore (UMES) Research Farm (N 38° 12’ 22”, W 75°

40’ 35”) (Fig 6). For a more comprehensive and detailed description of the UMES farm, drainage ditches, drainage ditch function, size, and hydrological properties found on the UMES Research Farm, the reader is directed to Vaughan et al., (2005).

Field and Laboratory Methods

For a detailed description of field methods used at this site and the methods used to perform field soil profile descriptions the reader is directed to Vaughan et al., (2005).

Soil pH was performed using a soil (moist) to water ratio of 1:1. Particle-size analysis was performed by pipette (Gee and Bauder, 1986). Acid ammonium oxalate-extractable Al, Fe, P ($Al_{ox,}$ $Fe_{ox,}$ P_{ox}) were extracted at 1:40 soil:(0.1M $(NH_4)_2C_2O_4 \cdot H_2O$ + 0.1M $H_2C_2O_4 \cdot 2H_2O$), and measured by inductively coupled plasma atomic emission spectroscopy (ICP-AES) (Ross and Wang, 1993). The degree of P saturation (DPS) was estimated from acid ammonium oxalate extractable Al_{ox} $Fe_{ox,}$ P_{ox} as:

$$DPS = (P_{ox}/0.5[Al_{ox} + Fe_{ox}]) \text{ X } 100\% \qquad [1]$$

where $Al_{ox,}$ $Fe_{ox,}$ and P_{ox} are in mmol kg^{-1} (Breeuwsma and Silva, 1992).

Mehlich-3 P extractions were conducted by shaking 2.5 g of soil in 25 mL of Mehlich-3 solution (0.2 N CH_3COOH + 0.25 N NH_4NO_3 + 0.015 N NH_4F + 0.013 N HNO_3 + 0.001 M EDTA) for 5 minutes. The supernatant was then filtered through a Whatman #1 paper and the filtrate was analyzed for P colorimetrically (Mehlich, 1984). Water-extractable P analyses were performed using a deionized water extraction of 0.5 g of soil in 100 mL of distilled water for 1 hr followed by colorimetric analysis of filtered extract (Kuo, 1996). Total P was analyzed by a modified semimicro-Kjeldahl procedure with P in digests determined by a modified method of Murphy and Riley (1962), with a spectrophotometer wavelength of 712 nm (Bremner, 1996). Organic C and total N was determined using a high temperature CNS-analyzer with an infrared detector (Bremner and Tabatabai, 1971).

Data Analyses

Statistical analyses were performed using S-Plus (Insightful Corporation, 2001) and the SAS GLM procedure (SAS Institute, 1990). In order to compare the means of characterization data between horizons, eight horizon classes were defined based on similar morphological and genetic characteristics (Table 1). Kolmogorov-Smirnov tests and descriptive statistics were used to assess normality. The DPS, pH, and organic C variables were found to be normally distributed. Other variables were found to be normal following log-transformation; log-transformed data were used for statistical inferences (Press et al., 1989). The CONTRAST statement in the SAS GLM procedure was used to test pre-planned one-way comparisons between the means of morphological horizon classes: Oi greater than A horizons (Dark A, Gley A); Dark A greater than Gley A horizons; A greater than subsoil C horizons (Gley C, Bright C, and Sulfidic C); Bright C greater than Gley C horizons; and Sulfidic C greater than Gley C horizons.

RESULTS AND DISCUSSION

Phosphorus concentrations were highly variable between profiles and across depths with ranges of 4855 mg kg^{-1} for TP, 4627 mg kg^{-1} for P_{ox}, 399 mg kg^{-1} for M3P, and 14 mg kg^{-1} for WEP. Oxalate-extractable Fe and Al, and organic C were also highly varied (Table 12). Across all samples, P_{ox} comprised a mean of 44% of TP, M3P 13% of TP, and WEP 1% of TP (Table 12). The percentage of P_{ox} and M3P of TP was greater in A horizons than subsoil C horizons (Table 12); the percentage of WEP of TP was lower in A horizons than subsoil C horizons (Table 12). Across all horizons, Fe_{ox} concentrations were on average about three times Al_{ox} concentrations (Table 12). Soil pH ranged from 2.6 to 6.1, with a mean of 4.7 and a standard deviation of 0.7. For comparison, mean soil pH (0-5 cm) of ditches in Delaware was 5.2 (Sallade and Sims, 1997a).

Depth Distribution

Data from three representative primary ditch profiles are presented in Table 13 and three ditch soil profiles from shallow-collection and deep-collection ditches are presented in Tables 14. In primary drainage ditches (<1.5 m), P, Al_{ox}, $Fe_{ox,}$, and DPS generally decreased with depth with the greatest decreases being observed between the alluvial A horizons and the subsoil C horizons (Table 13). However, the pattern of decreasing P, Al_{ox}, Fe_{ox} concentrations and DPS with depth was not consistent within A horizon or C horizon layers. For example, in ditch DX2-2 there was increasing TP and M3P within the four C horizons (Table 13). In ditch DX1-3 the greatest P_{ox} concentration was found in the A’3 horizon at 23-30 cm (Table 13). The surface Ag1 horizon of ditch

DX2-2 had substantially lower TP and P_{ox} concentrations than did the underlying A1 horizon (Table 13).

Within shallow-collection and deep-collection ditches, decreases in P, Fe_{ox}, and Al_{ox} as a function of depth were more consistent than those found in primary ditches (Table 14). The greatest decreases were observed at the transition between Oi and A horizons and between A and C horizons. Decreases in DPS as a function of depth were not consistent in all pedons. There were several exceptions to the trend of decreasing concentrations. Drainage ditch profile DXXS1-2 contained high concentrations of TP in the upper soil horizons that steadily decreased down to a depth of 44 cm below the soil surface. However, the Ag2 horizon was higher in WEP than the overlying Ag1 horizon (Table 14). Similar trends to DXXS1-2 were found in profile DXXD2-4, although there was a slight increase in TP of the A’2 horizon over the Ag2 horizon (Table 14). Oxalate-extractable Fe in DXXS1-2, DXXD2-4, and DXXD3-6 were generally decreasing with depth. In DXXS1-2 and DXXD2-4, Al_{ox} also generally decreased with depth. Organic C in profile DXXD2-4 was found to be irregular with depth while in DXXS1-2 and DXXD3-6 organic C was found to consistently decrease with depth (Table 14).

Possible explanations for the irregular distribution of P, Fe_{ox}, and Al_{ox} might include solutes concentration (Fe and P) differences in shallow and deep ground water, and soil heterogeneity (i.e. grain sizes and mineralogical differences) in alluvial A horizons. Solute concentration (primarily ferrous Fe) differences in ground water entering drainage ditches may result from differences in redox chemistry and source area parent materials, which may affect the amount of ferrous Fe present in ground water. Alluvial A horizons are formed from mineral materials that come from a variety of sources that act as parent materials for drainage ditch soils. Mineral materials can accumulate from the sedimentation of suspended soil from cultivated fields in surface runoff water, the slumping of drainage ditch sidewalls, and from the formation of precipitates from solutes transported in ground water (Vaughan, 2005; Sims et al, 1998; Nguyen and Sukias, 2002).

Morphological Horizon Class Differences

Alluvial A Horizons vs. Subsoil C Horizons

The most substantial differences were observed when contrasting A horizons and subsoil C horizons. Mean P fraction, Al_{ox}, and Fe_{ox} concentrations and DPS were substantially greater in alluvial A horizons than subsoil C horizons (Table 12; Table 15). Organic carbon was significantly greater in alluvial A horizons when compared to subsurface C horizons (Table 14). There are several possible explanations for the these differences. First, there was a general textural difference. The A horizons were generally loamy while the subsoil C horizons were coarser textured, dominated by very gravelly sands, gravelly sands, and sands (Vaughan, 2005). Fine-textured soils generally retain greater quantities of P than coarse-textured soils due to greater surface area. A high surface area facilitates the binding of more Fe and Al hydroxides to the mineral surface, allowing for more binding sites for P. Second, the accumulation of P in A horizons may be a result of exposure to P-laden surface runoff and direct poultry manure inputs. As the ditches accrete through mineral and organic debris deposition, P may be retained in these alluvial layers. Finally, the alluvium may just be enriched with P when deposited. The greater mean DPS in A horizons than in subsoil C horizons (Table 12; Table 15), suggests that there may not be sufficient sorbable P passing through these C horizons to saturate all available P sorption sites.

Oi Horizons vs. A Horizons

The Oi horizons are thin (0-9 cm) organically rich layers that are found in sufficient quantities to sample on the surface of ditch soils in the study area. Oi horizons were not present in all profiles sampled and in some cases were not present in sufficient quantity for laboratory analyses; therefore this analysis is based on four Oi samples (Vaughan, 2005). These horizons contain coarse organic debris, algal mats, decomposing organic materials and, in some cases, monosulfidic black oozes. The Oi horizons may be of particular importance for P losses as it lies at the soil-water interface. Contrasts indicated that TP, WEP, P_{ox}, Al_{ox}, and Fe_{ox} in Oi horizons were significantly greater than in A horizons (Dark A and Gley A); these differences were not significant for M3P, DPS, or OC (Table 12; Table 15). A mean of 81% of the total P was in the P_{ox} fraction in these samples. The Fe_{ox} concentrations were on average 7.4 times Al_{ox} concentrations, which was the greatest of any horizon class.

The high concentrations of P in Oi horizons may be the result of higher Fe_{ox} and Al_{ox} concentrations. The mean Al_{ox} concentration was nearly twice and the mean Fe_{ox} concentration was over four times the mean concentrations of Dark A horizons (Table 12). Phosphorus would be expected to be found most closely associated with Al and Fe hydroxides in the acidic soil environment of the drainage ditches at UMES (Vadas and Sims, 1998). The extra sorption sites provided by reactive Fe and Al in humic compounds may be retaining P (Petrovic and Kastelon-Macon, 1996). One explanation for the lack of significant differences in organic carbon between Oi horizons and alluvial A horizons is that the removal of coarse organic debris during sample preparation removed most of the sources for organic carbon in the sample resulting in an artificially low organic carbon percentage.

Dark A Horizons vs. Gley A Horizons

The A horizons in this study were classified into Dark A and Gley A based on value and chroma. The low chroma colors of the Gley A horizons are presumably a result of gleization. We hypothesized that gleization would reduce P sorption capacity and concentrations through the loss of reduced iron. Significantly greater organic C, Fe_{ox}, and Al_{ox} concentrations were observed in Dark A horizons than in Gley A horizons (Table 12; Table 15). The greatest difference was in Fe_{ox} with a mean concentration 2.6 times greater in Dark A than Gley A horizons. Total P and P_{ox} were significantly greater in the Dark A horizons than in Gley A horizons; however significant differences of M3P, WEP, and DPS were not observed (Table 12; Table 15). These differences indicate that reduced conditions may lower the P sorption capacity and TP and P_{ox} concentrations of ditch A horizons due to the dissolution of Fe-bound P. Management practices that lower redox potentials or extend periods of reduced conditions within drainage ditch systems, such as water-control structures, may increase gleization rates thereby reducing P retention.

Bright C vs. Gley C Horizons

The bright C horizon class encompasses C horizons that do not have depleted matrixes or significant accumulations of Fe concentrations; Gley C horizons did have reduced matrixes. We hypothesized that gleization would reduce P sorption capacity and retention of Gley C horizons relative to Bright C horizons. Significantly greater concentrations of TP and WEP were found in Bright C than in Gley C horizons. However, no significant differences were observed for these horizons for M3P, P_{ox}, Al_{ox}, Fe_{ox}, DPS or organic carbon. The reason for the higher concentrations of TP and WEP in the Bright C horizon class is not clear and is not supported by higher concentrations of other P fractions or a higher P sorption capacity.

Oxidized C horizons

The Oxidized C horizons are layers with a visible accumulation of Fe concentrations, thought to be the result of either an oxygenated ground water table at this depth or of the oxidation of sulfidic materials at depth, which may produce appreciable quantities of Fe when oxidized (Fanning et al. 2002). We hypothesized that these Fe concentrations may be acting as P sinks within the subsoil C layers. However, no significant differences between the Oxidized C horizons and other C horizons were detected for any variables, including Fe_{ox}.

Sulfidic C horizons

The presence of iron sulfide-bearing sulfidic C horizons in the ditch soil profiles may pose a water quality risk due to released acidity upon oxidation of the iron-sulfide rich materials. A commonly found iron sulfide mineral in geologically deposited materials in this region is pyrite. Ferrous iron is also released upon iron sulfide oxidation; this ferrous iron can move into solution or may be oxidized to insoluble ferric iron forms. The only statistically significant difference observed between Sulfidic C horizons and Gley Dark C horizons was a slightly greater Al_{ox} concentration (Table 12). While the acidity released upon oxidation of iron sulfides in this horizon may be affecting general ditch soil properties, it does not appear that these horizons differ significantly from other C horizons in their P retention characteristics.

CONCLUSIONS

Drainage ditches at UMES are high in P. Decreases in P with depth in drainage ditches were recognized. However, pedological differences appear to have the most significant effect on P concentrations and retention in Oi and A horizons. Differences in mean P concentrations between A horizons and C horizons were substantial. Furthermore, significant differences (except M3P) were also seen between Oi, Dark A horizons, and Gley A horizons. Differences observed between C horizons that are morphologically diverse were significant in some instance with respect to TP, M3P, and WEP, but were not statistically different with respect to P_{ox}.

In these soils, P accumulates at depth below 15 cm which is a common sampling depth To more accurately estimate the total P load in these ditch soils, sampling should continue at least to the contact between the alluvium and underlying coastal plain sediments. Information of P at depth in drainage ditches is necessary to estimate total P retention in ditch soils and may be critical when choosing between management strategies such as a mechanical ditch clean outs verses water-control structures.

Table 12. Chemical characterization data for morphological horizon classes which are comprised of soil horizons with similar morphological properties.

							Oxalate-extractable			
Horizon Class		pH	OC	TP	M3P	WEP	P	Fe	Al	DPS
			%			mg kg^{-1}				%
Oi	mean	5.3	4.0	2580	165	9	2093	13403	1809	25
(n= 4)	min	5.0	0.2	832	59	3	714	2796	1149	18
	max	5.7	7.6	4882	277	17	4631	37265	3525	36
	std.dev	0.4	3.6	1778	89	6	1830	16097	1150	8
Dark A	mean	4.9	5.1	1180	120	5	679	3293	975	23
(n= 38)	min	3.8	0.6	414	25	1	54	65	158	11
	max	6.1	12.4	4600	401	12	3165	23682	3201	36
	std.dev	0.6	2.8	815	63	2	680	5113	600	7
Gley A	mean	4.7	3.3	646	105	4	334	1235	630	22
(n= 19)	min	3.4	0.1	220	30	1	57	274	93	12
	max	6.0	9.6	1743	266	7	1102	2713	1897	30
	std.dev	0.8	2.6	356	56	2	256	717	381	6
Bright C	mean	4.8	0.2	355	24	2	48	401	186	11
(n= 12)	min	4.0	0.1	414	3	0	8	70	92	2
	max	5.6	0.4	1839	62	3	132	913	284	23
	std.dev	0.6	0.1	561	21	2	36	295	59	6
Oxidized C	mean	4.6	0.2	101	9	1	24	406	203	6
(n=6)	min	4.0	0.1	84	3	0	12	223	163	3
	max	5.5	0.2	158	16	3	42	723	303	11
	std.dev	0.6	0.1	28	5	1	11	180	57	3
Gley C	mean	4.3	0.3	113	15	0	49	337	227	8
(n= 35)	min	2.6	0.0	27	2	2	4	62	38	2
	max	5.6	1.4	315	73	6	570	1322	1249	26
	std.dev	0.7	0.3	71	14	2	102	250	239	5
Surface C	mean	3.6	2.5	168	17	1	46	461	158	10
(n= 2)	min	3.3	2.4	138	9	0	25	160	137	10
	max	3.9	2.6	197	24	1	6	762	179	11
	std.dev	0.4	0.1	42	11	1	29	426	30	0
Sulfidic C	mean	4.6	0.4	123	18	2	34	463	279	6
(n=10)	min	4.0	0.2	65	3	1	12	124	160	2
	max	5.2	0.6	230	45	4	56	1104	522	8
	std.dev	0.4	0.2	55	16	1	16	373	125	2
Total	mean	4.7	2.4	618	66	3	344	1800	561	15
(n=126)	min	2.6	0.0	27	2	0	4	38	38	2
	max	6.1	12.4	4882	401	17	4631	37265	3525	36
	std.dev	0.7	2.9	808	68	3	639	4490	582	9

Table 13. Soil characterization data of representative ditch soil profiles located within primary drainage ditches.

							Oxalate-extractable					
Horizon	Depth	Horizon Class	pH	TP	M3P	WEP	P	Al	Fe	DPS	OC	Texture Class
	cm			mg kg^{-1}						%		
						DX1-3						
A1	0-5	Dark A	4.4	1267	124	5	411	447	1664	29	6.9	LS
A2	5-15	Dark A	4.0	414	75	4	70	158	169	25	3.7	S
Ag1	15-23	Gley A	4.3	385	70	3	126	329	364	22	4.5	SL
A'3	23-30	Dark A	4.4	912	222	6	446	859	885	30	5.5	SL
2Cg1	30-52	Gley C	4.8	213	34	3	54	267	256	12	0.3	grLS
2Cg2	52-71	Gley C	4.2	136	9	0	14	179	158	5	0.6	LS
2Cg3	71-87	Gley C	3.9	133	20	1	27	177	154	9	0.5	S
						DX2-2						
Ag1	0-5	Gley A	6.0	220	86	3	104	290	996	12	6.1	SIC
A1	5-15	Dark A	5.1	1532	105	6	1387	1311	5944	29	1.7	SCL
A'g2	15-23	Gley A	4.6	734	40	4	484	681	2178	24	0.3	grLS
2Cg1	23-33	Gley C	4.9	112	19	3	44	215	541	8	0.3	SL
2Cg2	33-43	Gley C	4.8	69	16	1	30	158	292	9	0.1	SL
2Cg3	43-71	Gley C	4.0	71	14	2	28	143	272	9	0.1	LS
2C1	71-98	Bright C	4.7	103	15	2	28	252	529	5	0.2	LS
2C2	98-110	Sulfidic C	4.4	114	23	2	42	238	721	6	0.2	LS
						DX6-2						
A1	0-13	Dark A	4.6	762	142	9	548	746	1496	32	2.3	SIL
Ag1	13-29	Gley A	4.5	588	128	7	381	656	1334	25	1.1	SIL
2Cg1	29-47	GleyC	4.6	315	73	6	110	517	522	13	0.4	SL
2C1	47-62	Bright C	5.1	158	27	3	27	125	70	15	0.1	S
2C'g2	62-72	Sulfidic C	4.7	230	43	4	49	479	160	8	0.4	grLS
2C'g3	72-89	Sulfdic C	4.6	170	45	4	56	522	165	8	0.5	LS

Table 14. Soil characterization data of drainage ditch soil profiles located within shallow-collection and deep-collection ditches.

							Oxalate-extractable					
Horizon	Depth	Horizon class	pH	TP	M3P	WEP	P	Al	Fe	DPS	OC	Texture Class
	cm			mg kg^{-1}						%		
DXXS1-2												
Oi	0-3	Oi	5.6	3003	154	11	2233	1173	8729	36	1.9	ND
A1	3-13	Dark A	4.7	987	124	5	362	649	1479	23	1.9	SL
Ag1	13-17	Gley A	4.8	747	118	5	291	628	950	23	1.3	SL
Ag2	17-26	Gley A	5.3	737	138	7	299	741	607	25	0.1	SL
2C1	26-44	Bright C	5.3	236	51	3	56	108	160	17	0.1	S
2Cg1	44-67	Gley C	5.6	88	19	2	27	212	128	8	0.1	S
2Cg2	67-90	GleyC	4.8	54	7	1	17	257	176	4	ND	S
DXXD2-4												
Oi	0-7	Oi	5.1	1605	169	3	796	1388	2796	25	7.6	SL
A1	7-16	Dark A	4.7	1063	62	2	378	817	2624	16	6.5	SL
Ag1	16-24	GleyA	4.6	566	34	2	148	468	1107	13	2.1	S
Ag2	24-30	Gley A	4.6	463	48	1	125	378	929	13	3.3	grLS
A'2	30-38	Dark A	4.4	504	39	1	136	460	1013	12	3.4	grLS
A'3	38-47	Dark A	4.4	434	28	1	72	256	449	13	4.4	LS
2Cg	47-68	Gley C	4.4	125	10	1	26	194	296	7	1.4	grS
2C	68-87	Bright C	4.3	77	5	0	9	92	74	6	0.1	grS
DXXD3-6												
A1	0-6	Dark A	5.5	1049	149	2	487	1363	1506	20	8.3	SL
A2	6-15	Dark A	5.4	442	82	2	159	613	542	16	4.5	SL
Ag1	15-30	Gley A	5.2	412	65	2	125	588	654	12	4.0	SL
2Cg1	30-73	Sulfidic C	5.2	192	28	1	46	304	434	8	0.5	grS

Table 15. One-way probability values for contrasts on differences between morphological horizon class means.

Contrast	TP	P_{ox}	M3P	WEP	Al_{ox}	Fe_{ox}	DPS	OC
Oi vs. A's	0.0008	0.0007	NS†	0.0408	0.0036	0.0001	NS†	NS†
Dark A vs. Gley A	0.0008	.0201	NS†	NS†	0.0115	0.0349	NS†	0.0013
A's vs. Subsoil C	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001
Bright C vs. Gley C	0.0023	NS†	NS†	0.0230	NS†	NS†	NS†	NS†
Oxidized C vs. subsoil C's	NS†	NS†	0.0492	NS†	NS†	NS†	NS†	NS†
Surface C vs. A's	0.0200	0.0480	0.0222	0.0008	0.0284	NS†	0.0019	NS†
Surface C vs. subsoil C's	NS†	NS†	NS†	0.0452	NS†	NS†	NS†	NS†
Sulfidic C vs. Gley C	NS†	NS†	NS†	NS†	0.0247	NS†	NS†	NS†

† NS; not significant at α=0.05

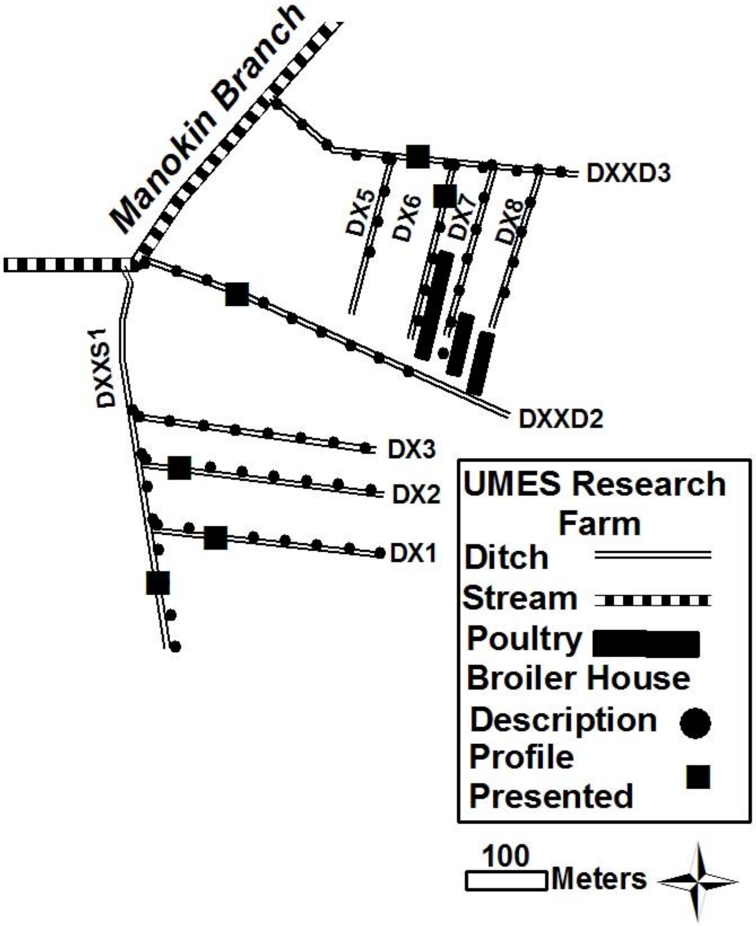


Figure 14. Overview map showing drainage ditch study area and selected profile description locations on the University of Maryland Eastern Shore Research Farm, Princess Anne, MD (Adapted from Vaughan, 2005).

Chapter 6: UMES farm soil survey, subsoil transects, sulfidic material incubations, and conclusions

ABSTRACT

An investigation of agricultural drainage ditch soils through a pedological framework was conducted at the University of Maryland Eastern Shore Research Farm (UMES). A soil survey of the UMES farm was needed to confirm previous mapping efforts conducted by the USDA-NRCS. While performing this soil survey, a subsoil horizon was identified that was enriched with what is thought to be ferric iron. In addition, what appeared to be sulfidic materials were also found at depth. The results of the investigations of UMES farm soils, the ferric iron enriched horizon, and the possible presence of sulfidic materials at UMES are present here in order. In addition, x-ray diffraction data are also presented from samples collected from the alluvium of UMES drainage ditches. Conclusions, summary thoughts and insights are also presented

UMES Farm Soil Survey

INTRODUCTION

A soil survey of the University of Maryland-Eastern Shore Research Farm (UMES) was initiated in an effort to confirm the mapping units presented in the Somerset County Soil Survey (Matthews and Hall, 1966). In addition, the soil survey would provide useful information for research being conducted on the farm. The last official soil survey of Somerset County was performed in 1966, and is currently in the process of being updated by soil scientists from the USDA-NRCS office in Princess Anne, MD.

MATERIALS AND METHODS

Site Description

This study was conducted at the University of Maryland Eastern Shore Research Farm (UMES) (N 38° 12’ 22”, W 75° 40’ 35”) located in Princess Anne, Somerset County, Maryland. For a comprehensive site description including ditch soils, please see Vaughan, 2005.

Field Methods

Confirmation of existing soil map units was performed by conducting soil profile descriptions and matching the soil profile data with existing map unit descriptions. Soil profile descriptions were performed by both traditional auger holes and soil profile faces on the stream banks of the Manokin Branch. Soils were excavated to a depth of 2-m using a 6-cm bucket soil auger. At each field description site, soil morphological characteristics such as horizonation, structure (where applicable), moist consistence (where applicable), and redoximorphic features, were identified and described in the field based on standard soil survey techniques (Soil Survey Staff, 1993). Soil color was described using a Munsell® color chart in the field, and field soil texture was performed by hand. Samples from each horizon described at each sampling location were collected and placed into plastic bags, packaged in coolers, and brought back to the laboratory for analysis. Soils were then air-dried, coarse organic debris removed, and ground to pass a 2-mm sieve.

Mapping

A digital copy of the 1966 Somerset County Soil Survey (Matthews and Hall, 1966) was obtained through Susan Demas, a soil scientist with the USDA-NRCS, Somerset County, MD. In addition, a 0.3-m contour interval laser-collected elevation data set of the UMES research farm and campus was provided by the geospatial lab at UMES (Fig. 16). The layer file containing soil survey map units and lines as well as the topography data were placed into a GIS.

RESULTS AND DISCUSSION

Roughly 30 soil profile descriptions were performed using a bucket auger. Several more auger holes without descriptions were performed as well. About 27 of these were adjacent to DX1, DX2, DX3. Other profiles were excavated in the fields west of DX5 and east of the Manokin Branch. Three profile descriptions were performed on the banks of the Manokin Branch, a photo of one appears in Figure 16.

The soil survey performed by Matthews and Hall (1966) was reasonably accurate (Fig. 17). For example, a small map unit of Portsmouth (very poorly drained) in a large expanse of map unit Othello (poorly drained) near the head of DX1 was correctly identified. However, no identification of sulfidic materials at depth throughout the farm was made because the definition of sulfidic materials as described by Soil Taxonomy (2003) was non-existent. Therefore, Matthews and Hall (1966) did not identify any areas on the UMES farm that contained sulfidic materials. However, sulfidic materials at depth were present throughout the farm (Fig. 17). This oversight was easy to make considering the variation in depth, particle-size distribution, and color of this material. Near the outlet of DX3, sulfidic materials come within less than a meter of the soil surface, where sulfidic materials near the outlet of DX8 are nearly 2 m below the soil surface. Sulfidic materials are also exposed in the Manokin Branch Bank west of DX5, with some of these materials below the water surface. The presence of this material and its depth around the farm should be included in the next soil survey of the area.

Independent of this oversight, the soil map units that were described by Matthews and Hall (1966) were accurate enough to keep the original lines. No large shifts or changes in the lines are necessary at this time. Further exploration of the sulfidic materials at depth could certainly change this suggestion in the future.

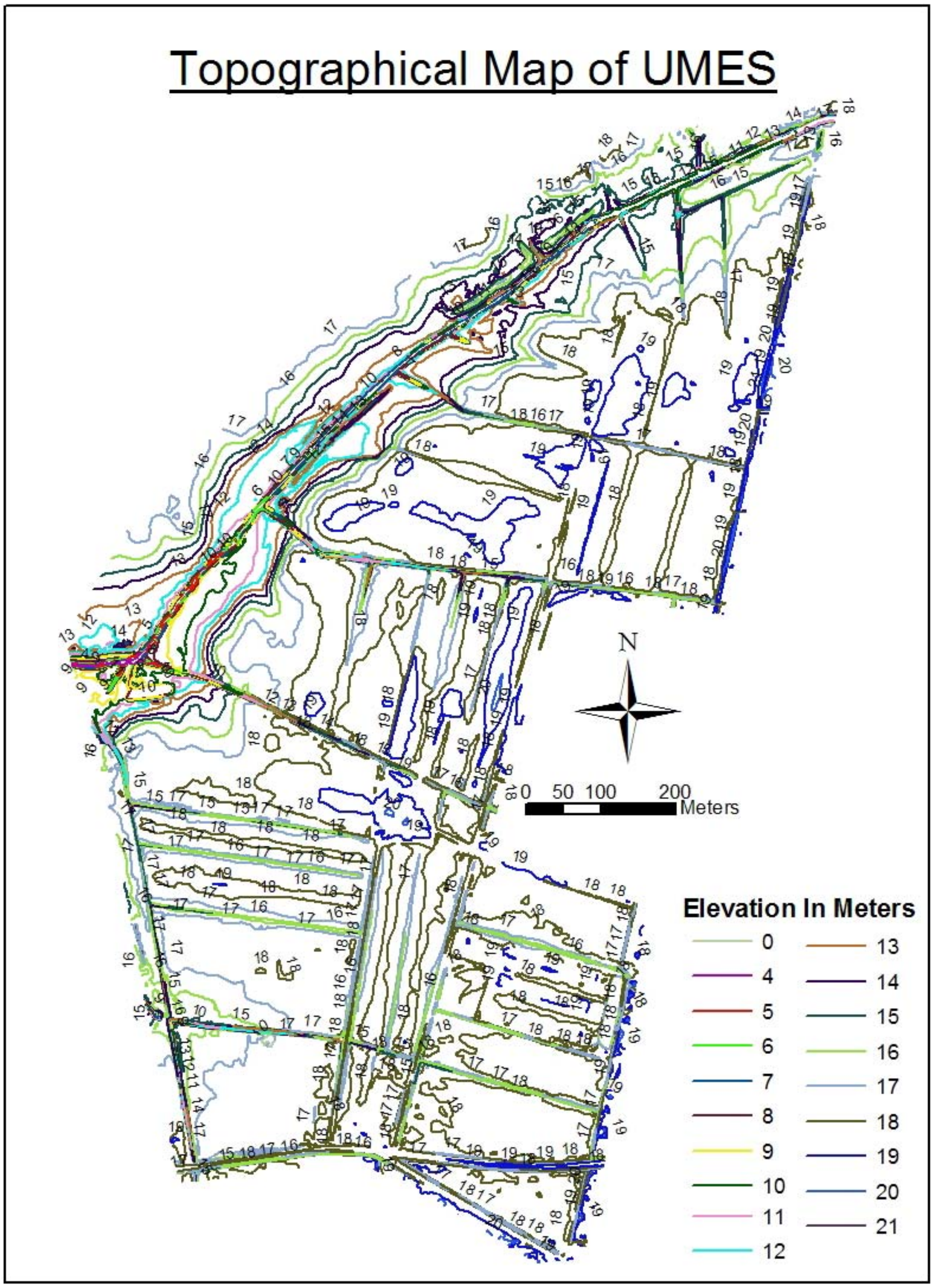


Figure 15. Topographical map of UMES Research farm generated from laser altimetry data. Contour interval is 0.3 m.

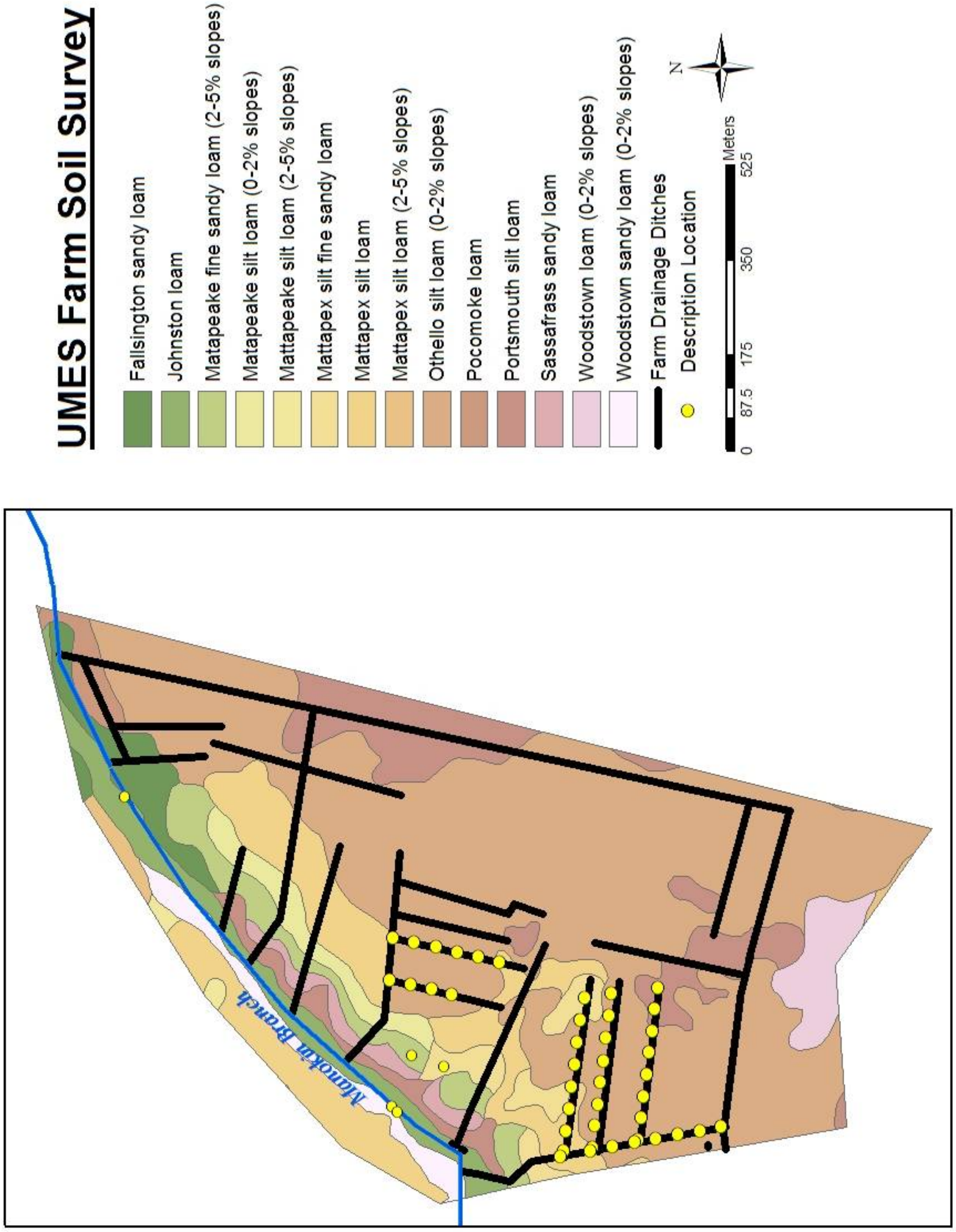


Figure 16. A digital copy of the 1966 Somerset County Soil Survey with drainage ditches and ditch soil survey locations (adapted from Matthews and Hall, 1966).

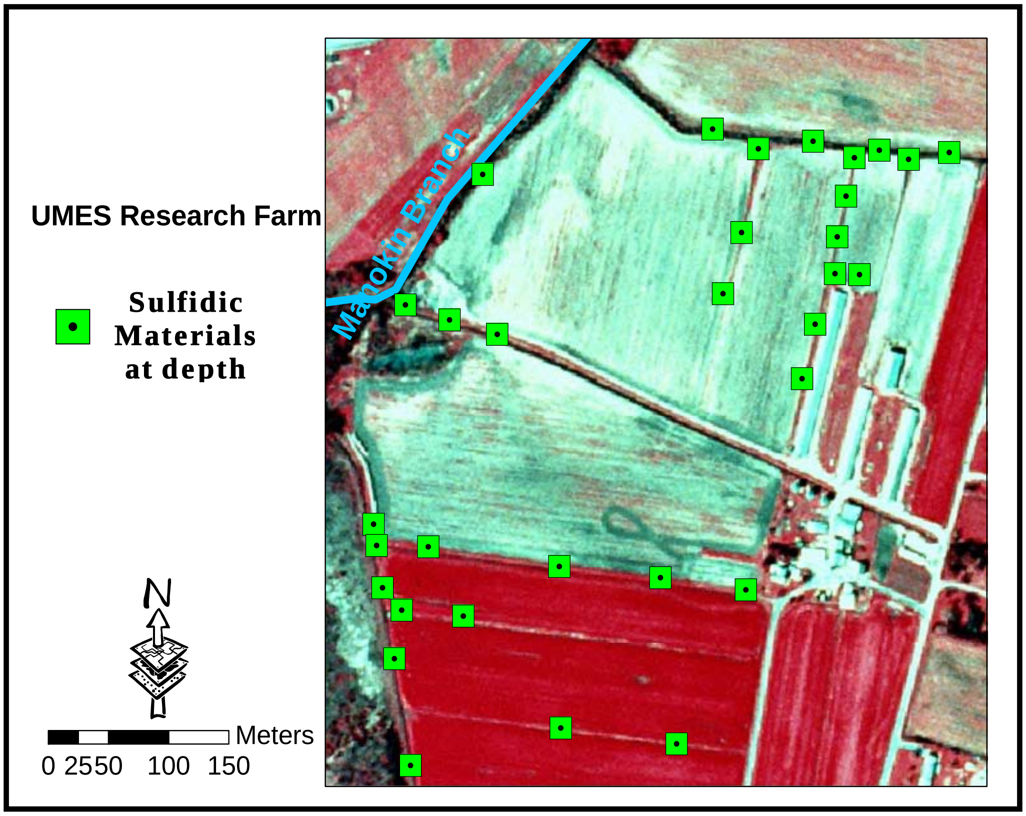


Figure 17. Map Shows area of UMES farm where sulfidic materials at depth have been identified.

Oxidized Horizon Transect

INTRODUCTION

During the summer of 2003 while performing soil profile descriptions of field soils near drainage ditches, a soil horizon that was bright in color (10YR 6/8) was found consistently at about 1 m from the soil surface. This horizon was found in nearly all profile descriptions performed on ditches DX1, DX2, and DX3. Early hypotheses regarding this horizon were that the ground water at this depth was oxygenated due to changes in ground water tables and the infiltrating rainwater as a result of ditching. An oxygenated ground water layer would provide the oxygen needed to oxidize ferrous Fe in the ground water. This theory was supported in the literature by Hayes and Vepraskas (2000). At the urging of the thesis committee, further evidence was needed to confirm this theory. This horizon is thought to contain high amounts of ferric Fe. If this horizon was found to be extensive around the farm, then it may have the potential to sorb large amounts of P and would be important to the overall understanding of P losses through subsurface drainage.

MATERIALS AND METHODS

A series of five transects were established in a perpendicular direction away from four randomly selected drainage ditches (Fig 19). Auger holes were dug using a 7.62-cm bucket auger down to a depth below this horizon. Auger holes were placed in the bottom the ditch, and at 1, 5, 10, 15, 20, 25, and 30 m away from the ditch. The top and bottom depths of the brightly colored horizon were recorded for each auger boring.

RESULTS

Results of transects are presented in Figures 20-24. The top and bottom depth where evident are recorded. Sample locations within drainage ditches had 100 cm added to reflect their positional difference from the field auger holes.

DISCUSSION

The oxidized horizon was very clear and evident near drainage ditches DX1, DX2, and DXXD2. Drainage ditch DX6 did not contain a uniform horizon away from the drainage ditch. At 10 and 20 m, a horizon containing many concentrations was noted, but at no other locations or depths was a horizon containing colors close to 10YR 6/8 found. DXXD2 contained no oxidized horizon within the drainage ditch, however, a horizon with bright colors was found in the field soils in the transect.

A second reason for this horizon is oxidation of sulfidic materials and the release of Fe. Sulfidic materials at depth below the entire UMES study site have been found. The lowering of ground water tables by ditching on the farm may be oxidizing these materials which would theoretically otherwise be stable in the reducing environment of the ground water. Iron is released from the oxidation of pyrite (FeS_2) in the ferrous form. Ferrous Fe would be easily oxidized in the ground water if exposed to oxygen. This could explain why the horizon is so bright and is at a depth in the field that corresponds to the bottom of the drainage ditch.

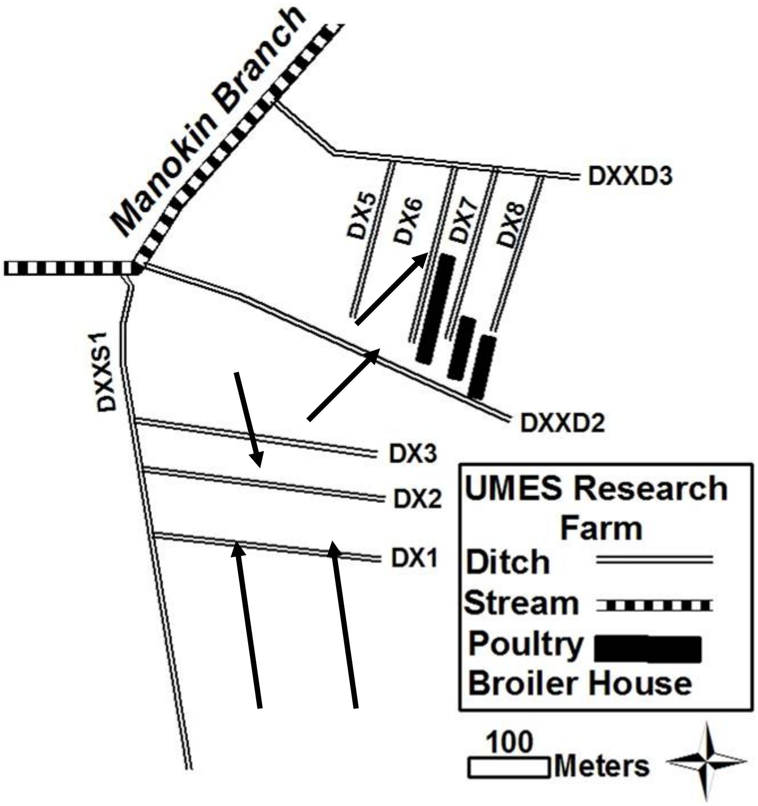


Figure 18. Map of UMES Research Farm and the locations of transects that were performed.

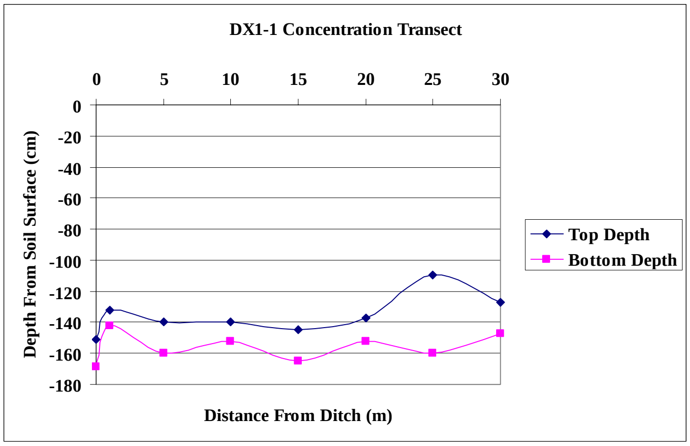


Figure 19. Graph shows the top and bottom depth (cm) of a soil horizon enriched with what is believed to be ferric iron along a transect away from the head of drainage ditch DX1. 100 cm have been added to the depth at 0 m to reflect the difference in position of the drainage ditch relative to the land surface.

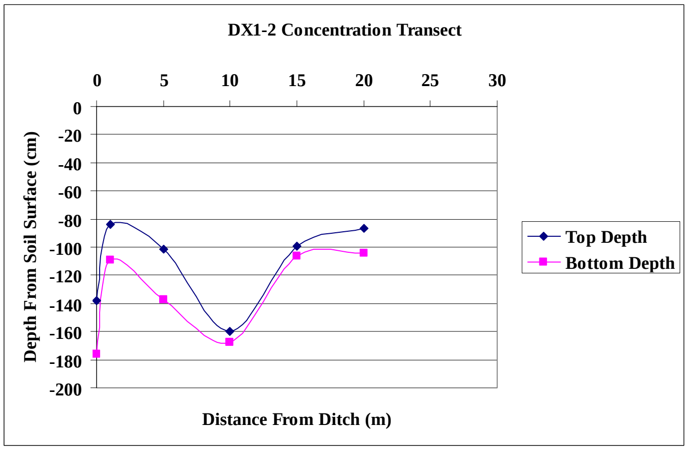


Figure 20. Graph shows the top and bottom depth (cm) of a soil horizon enriched with what is believed to be ferric iron along a transect away from the outlet of drainage ditch DX1. 100 cm have been added to the depth at 0 m to reflect the difference in position of the drainage ditch relative to the land surface.

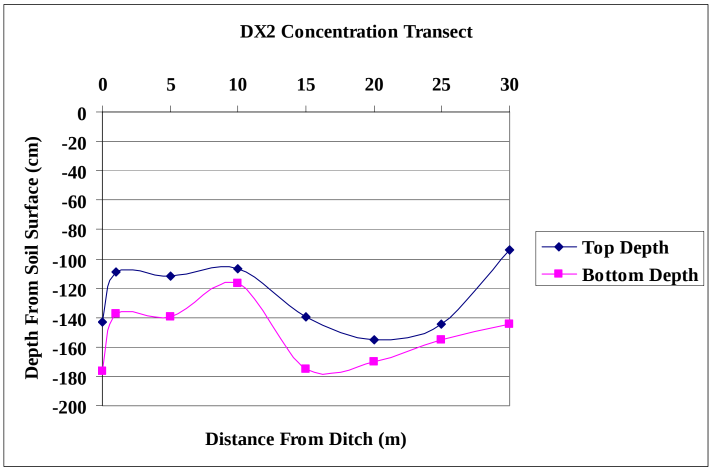


Figure 21. Graph shows the top and bottom depth (cm) of a soil horizon enriched with what is believed to be ferric iron along a transect away from the middle of drainage ditch DX2. 100 cm have been added to the depth at 0 m to reflect the difference in position of the drainage ditch relative to the land surface.

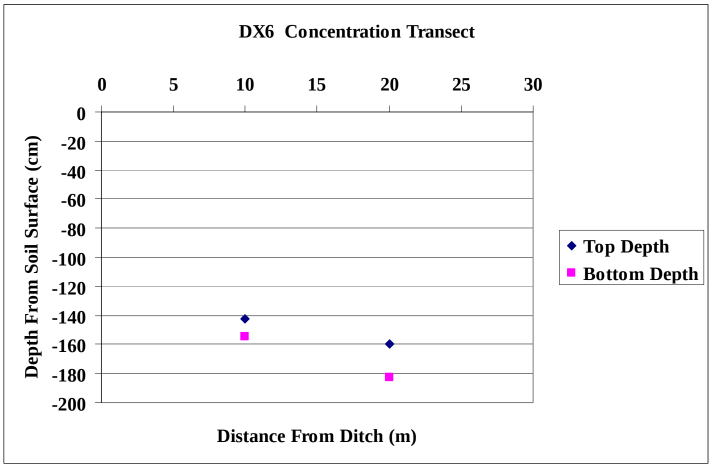


Figure 22. Graph shows the top and bottom depth (cm) of a soil horizon enriched with what is believed to be ferric iron along a transect away from the middle of drainage ditch DX6. Ferric iron was found only at 2 points along the transect.

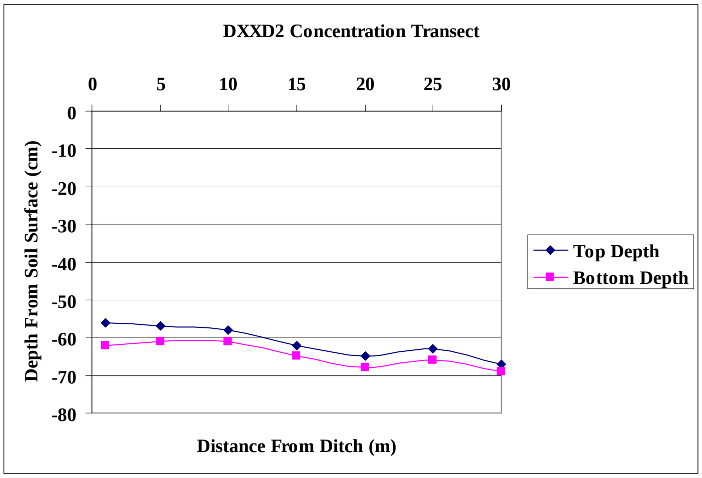


Figure 23. Graph shows the top and bottom depth (cm) of a soil horizon enriched with what is believed to be ferric iron along a transect away from the near the head of drainage ditch DXXD2. No horizon was identified within the ditch; however it was located outside of the ditch.

Figure 24. Picture shows the ditch soil horizon that has ferric forms of iron present.

Sulfidic Material Incubations

INTRODUCTION

While performing dry pH measurements on soil samples collected near drainage ditches for soil survey purposes, a dark soil was found to be very-acidic (pH=2). Additional samples were collected in the field, and this time they were kept on ice and transported back to the laboratory. Soil pH was performed on the moist samples upon arrival back at the lab. Soil pH was again found to be low, but around 4.0. It was suggested by others that based on the color (10YR 4/1), that these may be sulfidic materials. Upon addition of H_2O_2 to the soil sample in a beaker, a violent reaction occurred. Therefore, a study was begun to confirm the presence of sulfidic materials by established Soil Survey methods.

MATERIALS AND METHODS

Field Methods

Soil samples from various locations around the UMES Research Farm were collected while performing soil survey descriptions. In addition, descriptions of drainage ditch soils yielded additional soil samples that were suspected to contain pyrite (FeS_2). The suspicion of samples containing pyrite was based on a dark color (at least 10YR 4/1) color. Samples were collected using a soil bucket auger and a spade shovel in the spring of 2003.

Laboratory Methods

Incubations were performed using the methods of Soil Survey Staff (2003). Enough soil to fill a 12.5 x 12.5 cm tall plastic cup to a depth of 1 cm was used. Soils were kept in a moist field capacity state at room temperature throughout the experiment except for when pH measurements were being taken when soils would be moistened beyond field capacity. Once a week soil pH measurements were performed by using a pH electrode and adding enough water to permit a soil pH measurement. To qualify as sulfidic materials, mineral and organic soils must begin at a pH greater than 3.5, and after incubation for 8 weeks, drop 0.5 pH units to a value of 4 or less (Soil Survey Staff, 2003).

RESULTS

Incubations were performed for 3 straight weeks on all samples. Measurements were stopped for a period seven weeks after the third week because all samples had dropped at least 0.5 pH units and all were below 4.0. However, measurements were continued at 10 and 11 weeks after they had begun to confirm the pH drop. The results of the pH drop are presented in figures 26-28. Images of selected sulfidic materials are presented in Figures 28-32.

DISCUSSION

All samples analyzed met the definition of sulfidic materials by Soil Taxonomy. Confirmation of the fact that sulfidic materials are present at depth in every area of the farm is rather striking. As presented before, sulfidic materials have never been mapped in this area of Somerset County before. The presence of sulfidic materials in a region with intensive agriculture is certainly a concern. However, methods of water table control, or subirrigation may be the best method for preventing the oxidation of sulfidic materials in drainage ditches.

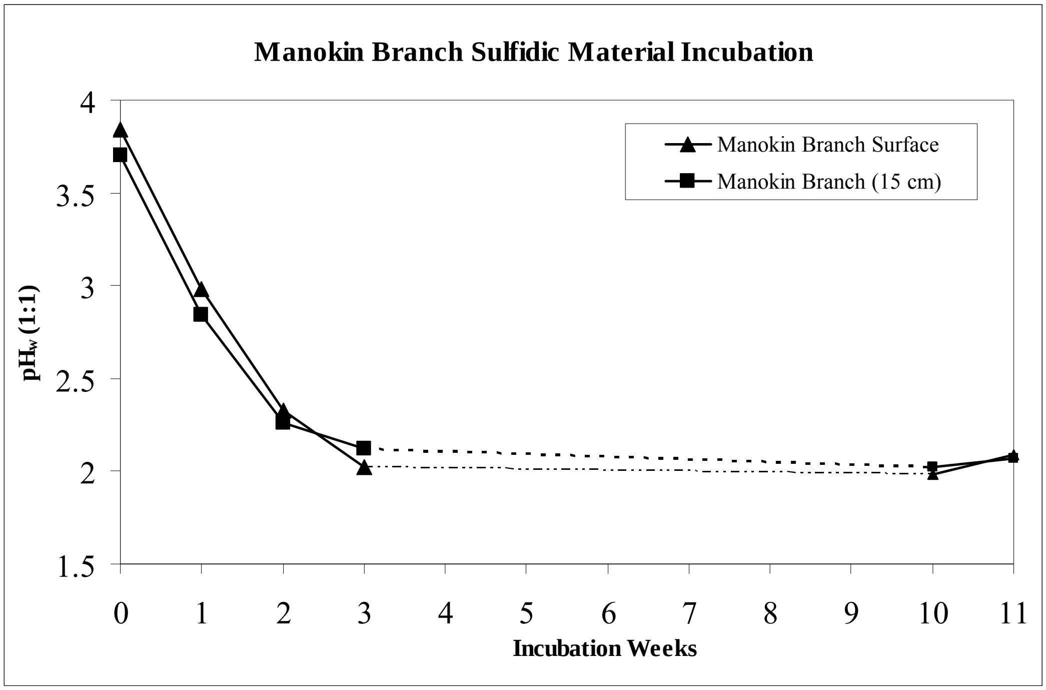


Figure 25. Graph shows a drop in pH after 3 weeks incubation time and at 10 and 11 weeks. Dotted lines represent a time when pH was not being measured. Soil samples were collected from the bank of the Manokin Branch (Manokin Branch Surface) and from 15 cm deep into the bank (Manokin Branch 15 cm) west of DX5 at UMES.

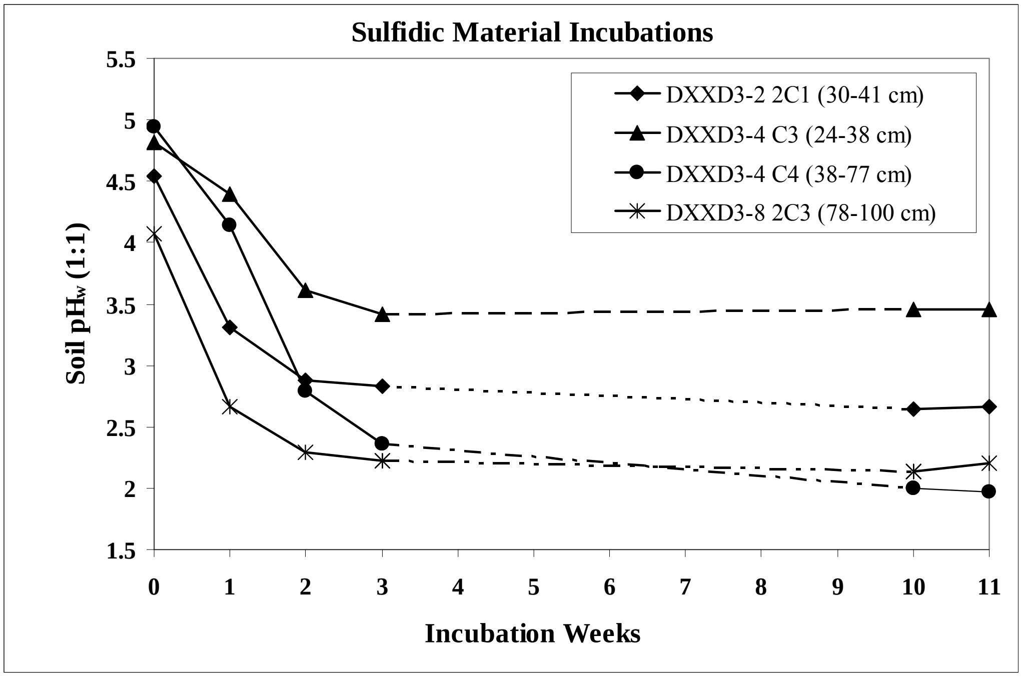


Figure 26. Graph shows a drop in pH after 3 weeks incubation time and at 10 and 11 weeks. Dotted lines represent a time when pH was not being measured. Change in pH over time during moist incubation of selected samples around the UMES farm.

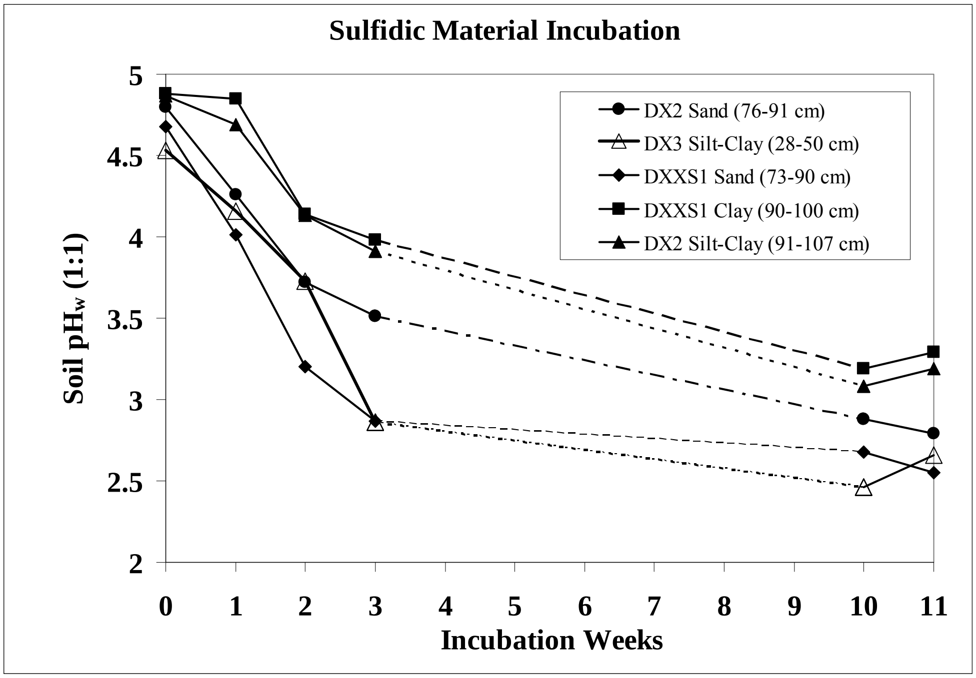


Figure 27. Graph shows a drop in pH after 3 weeks incubation time and at 10 and 11 weeks. Dotted lines represent a time when pH was not being measured. Change in pH over time during moist incubation of selected samples around the UMES farm.

Figure 28. Photo of sulfidic materials (dark grey) on the banks of the Manokin Branch west of DX5 at UMES.

Figure 29. Photo of sulfidic materials (dark grey) on the banks of the Manokin Branch west of DX5at UMES.

Figure 30. Photo of sulfidic materials under the Manokin Branch west of DX5 at UMES.

Figure 31. Photo of sulfidic materials (dark grey) in a soil profile located near the outlet of DX2. (Photo courtesy of Brian Needelman)

Figure 32. Close-up photo of sulfidic materials (dark grey) in a soil profile located near the outlet of DX2. (Photo courtesy of Brian Needelman)

Figure 33. Close-up photo of sulfidic materials (dark grey)in a soil profile located near the outlet of DX2. Notice the iron-rich groundwater in hole. (Photo courtesy of Brian Needelman

X-Ray Diffraction

Objective

The objective of this procedure was to identify different mineral species in the sand, silt, and clay fractions by x-ray diffraction.

Site and Sample Description

The soil samples that were used in this analysis are from agricultural drainage ditches from the University of Maryland Eastern Shore Research farm located in Princess Anne Maryland. Samples were collected from DX2-6 (A 0-4 cm) and DX1-2 (A3 8-20 cm).

Procedure

A full morphological profile description of the sample site was performed at the time of sampling in the fall of 2004 using a spade shovel, and standard soil survey techniques (Soil Survey Staff, 1993). The samples were placed into plastic bags and put on ice and brought back to the labororatoty for analysis. Samples were air-dried at 25^{o}C, ground, and passed through a 2-mm sieve.

The samples were then fractionated into separate sand, silt, and clay fractions. The size classes are as follows: sand (2.0-0.05 mm); silt (0.05-0.002 mm); and clay (<0.002 mm). The sand was separated using a wet sieving procedure. The silt and clay fractions were separated using centrifugation, and flocculation using salts to separate the two size fractions.

Once separated, the light sand fractions (<2.95 g cm^{-1}) were placed into individual cylinders with two carbide spheres and placed in an oscillating shaker at very high speeds to pulverize the sand grains for approximately 5 min. Once crushed, an adequate amount was placed into an aluminum powder mount and pressed with a piece of filter paper to ensure a rough surface. The sample was then ready to be scanned. The only sample with enough heavy sands to be run was sample DX1-2. This sample was ground and crushed by hand with a mortar and pestle. Once of sufficient consistency, the sample was sprinkled over a piece of double-sided tape and onto a glass slide and then it was ready to be run. The silt fraction was dried after separation and was crushed in a mortar and pestle and homogenized. A sufficient quantity was then placed in an aluminum holder and was then scanned. Clay samples for XRD analysis were placed into centrifuge tubes labeled Mg or K (0.15 g into 15 ml centrifuge tubes).

One had to be careful to make sure that the proper salt solution either MgCl or KCl was added to each treatment. If one accidentally messed up, they had to re-pipette more soil into a new test tube. Once placed on the glass slide, Mg samples were put into a desicator with ethylene glycol at 25°C. The K samples were left out to dry at 25°C. All Mg slides were run on the XRD at 25°C, while the K slides were run at 25°C, after heating to 300°C and after heating to 550°C.

The source of x-rays that were used was CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 4° 2-theta to 60° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 40 minutes. The aluminum sample holder or glass slide with double sided tape was placed into the x-ray machine and run.

RESULTS

Please see scan data below. There were problems associated with the aluminum holder that were discovered. The aluminum holder was showing a peak at 2.028 and 2.034 Ǻ. Therefore one must take care in interpreting peaks in this region on the sand and silt scan data. This problem was not present in the clay fraction because these samples were run using glass slides.

UMES DX2-6 (A 0-4 cm) Light Sand

Mineral	Relative Abundance
Quartz	XXXX
Albite	X
Microcline	tr

The light sand fraction of DX2-6 appears to be mostly quartz. The XRD scan below can be used as a reference. The most intense peaks are from quartz, which is also the most abundant. Albite is also present in this sample; however it is in a much smaller quantity than quartz.

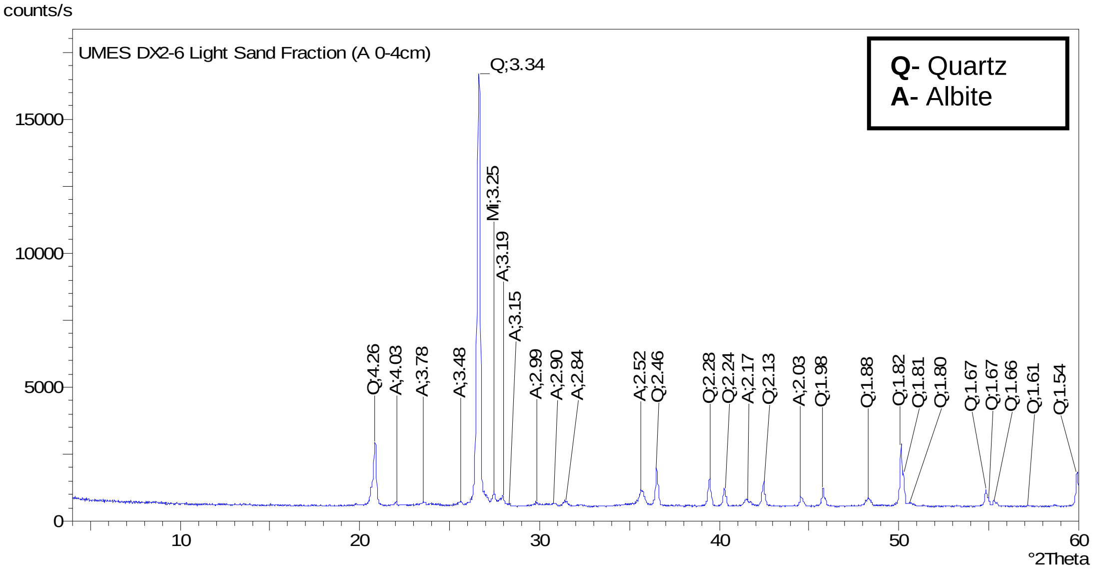


Figure 34. X-ray diffraction scan of DX2-6 (A 0-4 cm) light sand fraction. CuKα at 45 kilo-volts and 35 milli-amperes. A continuous scan was run from 4° 2-theta to 60° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 40 minutes.

UMES DX1-2 (A3 8-20 cm) Light Sand

Mineral	Relative Abundance
Quartz	XXX
Albite	XXX
Microcline	XX
Gibbsite	tr

Several mineral species were identified in this sample. Again, this sample was dominated by quartz but contained a few minerals that were not identified in the previous sample (DX2-6). This sample also contained appreciable amounts of albite, microcline, and muscovite. Gibbsite also appeared to be present; however, the intensity of the peak was extremely small and is almost nonexistent. Therefore, if it is present in the sample, it is in very trace amounts.

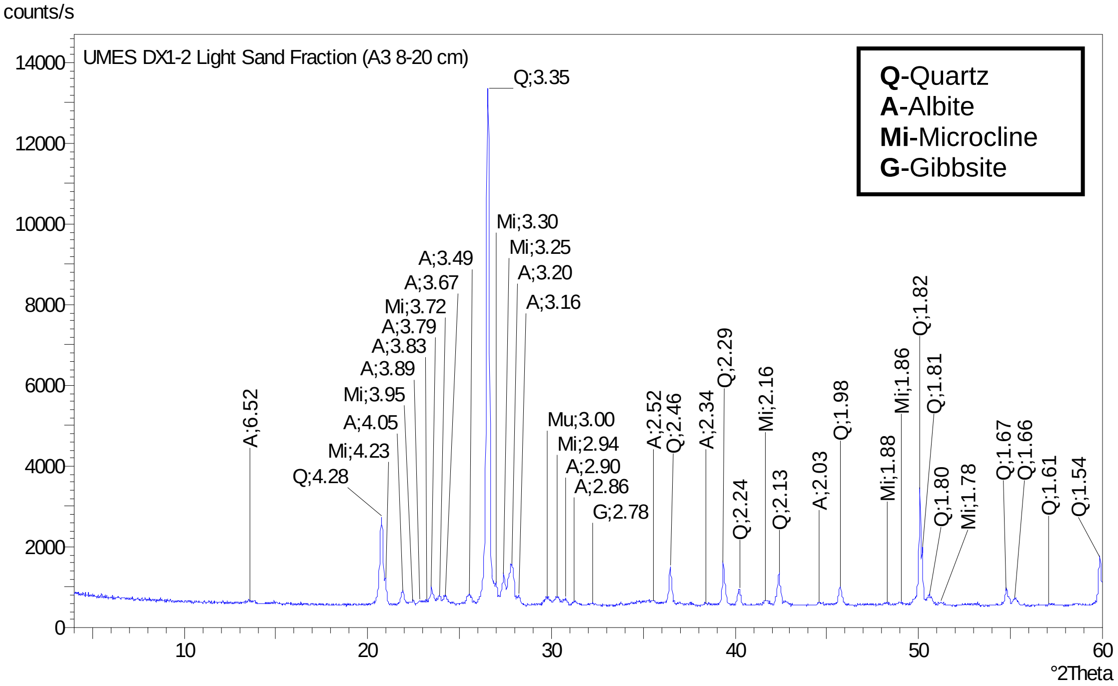


Figure 35. X-ray diffraction scan of DX1-2 (A3 8-20 cm) light sand fraction. CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 4° 2-theta to 60° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 40 minutes.

UMES DX1-2 (A3 8-20 cm) Heavy Sand

Mineral	Relative Abundance
Ilmenite	XXX
Zircon	X
Rutile	X
Muscovite	tr
Quartz	tr
Amphiboles	tr

The heavy sands were very difficult to interpret. The computer generated peak identification tended to yield minerals that were radioactive and other minerals that one could assume did not exist at the University of Maryland-Eastern Shore Research Farm. In addition, the noise in the scan, or rather all the small peaks made it difficult to pick out peaks that may be relevant. Many minerals were identified however. Ilmenite (iron titanium oxide) was the most abundant identified in this sample. The second most abundant mineral could not be identified. The peak at 3.99 Ǻ had the highest intensity, but I was unable to identify it. Several resources were checked for possible matches, however, none were successful. Zircon and rutile were also found, and zircon was identified in grain mounts. Amphiboles were identified, however I was unable to identify a specific species. Muscovite and quartz were also identified in trace amounts pointing to a little contamination of the sample in the separation procedures for the heavy and light sand fractions.

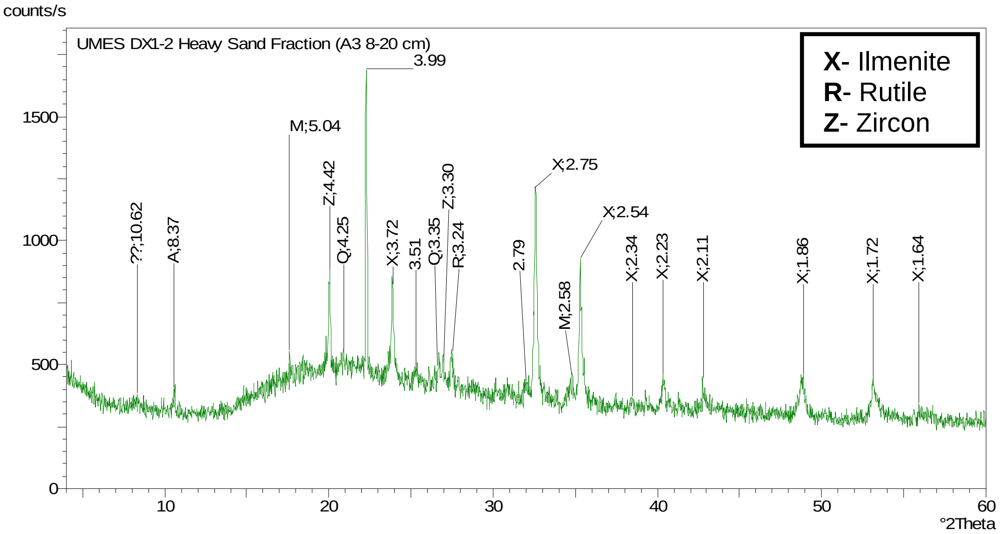


Figure 36. X-ray diffraction scan of DX1-2 (A3 8-20 cm) heavy sand fraction. CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 4° 2-theta to 60° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 40 minutes.

UMES DX2-6 (A 0-4 cm) Silt

Mineral	Relative Abundance
Quartz	XXXX
Albite	XX
Anatase	tr
Microcline	tr
Muscovite	tr
Kaolinite	tr

Quartz was the most dominant mineral in this sample. Albite was also present in a low quantity. Anatase, microcline, muscovite, and kaolinite were also present in trace quantities. This sample is relatively simple in terms of mineralogy.

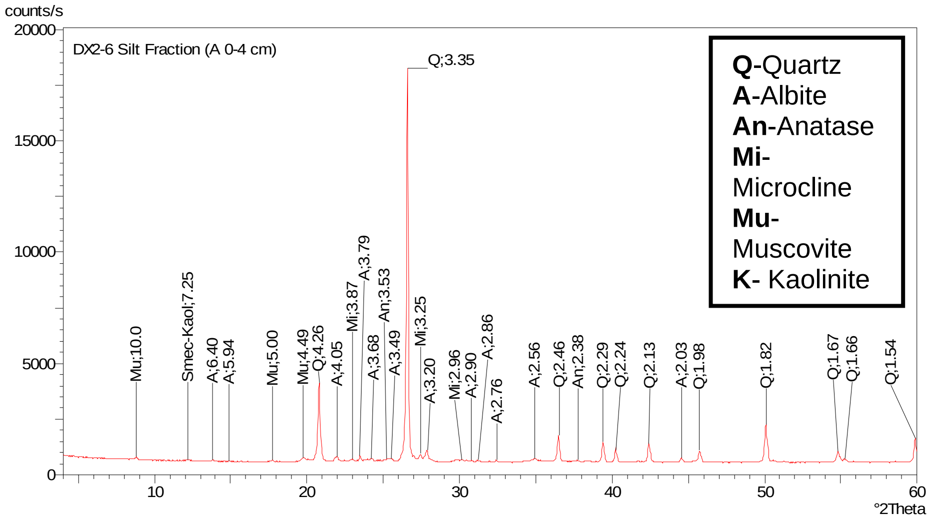


Figure 37. X-ray diffraction scan of DX2-6 (A 0-4 cm) silt fraction. CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 4° 2-theta to 60° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 40 minutes.

UMES DX1-2 (A3 8-20 cm) Silt

Mineral	Relative Abundance
Quartz	XXXX
Albite	XX
Microcline	tr
Muscovite	tr
Kaolinite	tr

Quartz was again the most dominant mineral in this silt sample. Albite was also present in a low quantity with the peaks for albite having a very low intensity compared to most of the quartz peaks. Microcline and muscovite were also present in trace quantities. Again, this sample is relatively simple in terms of mineralogy.

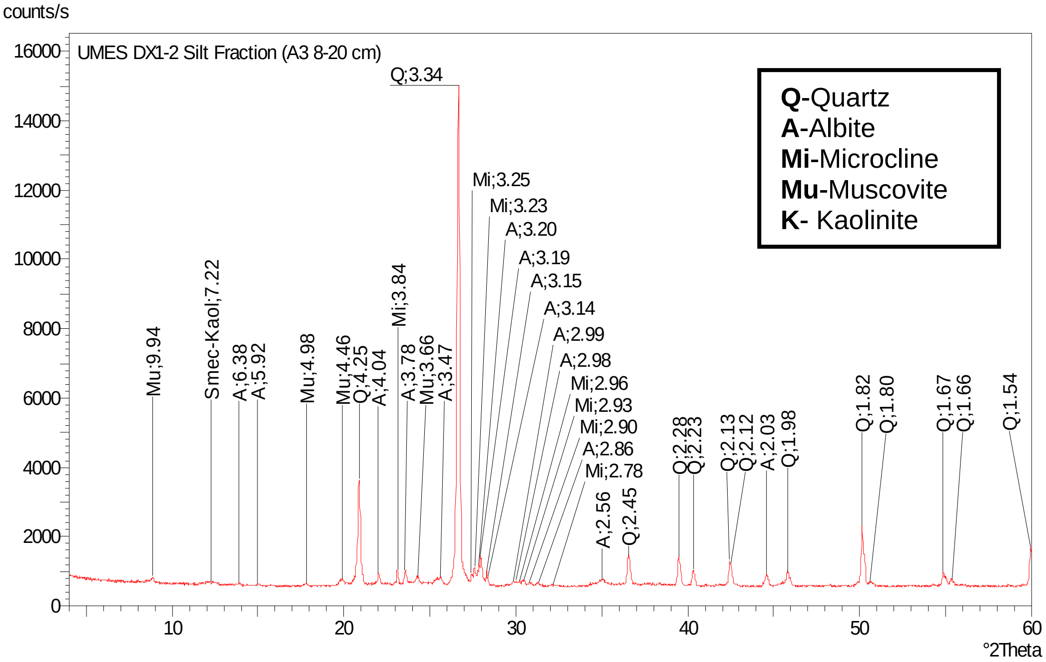


Figure 38. X-ray diffraction scan of DX1-2 (A3 8-20 cm) silt fraction. CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 4° 2-theta to 60° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 40 minutes.

UMES DX2-6 (A 0-4 cm) Clay

Mineral	Relative Abundance
Kaolinite	XXX
Quartz	XX
Muscovite	XX
Vermiculite	X
Anatase	tr
Feldspars	tr
Goethite	tr

This sample was found to be high in quartz and kaolinite. The loss of the kaolinite peaks upon heating to 550^{o}C is a fairly good indication that kaolinite was present. The strong peak at 5 nm indicates the presence of muscovite, while the peaks at 14.7 nm indicate the presence of vermiculite. Anatase, feldspars, and gibbsite were all present in trace amounts.

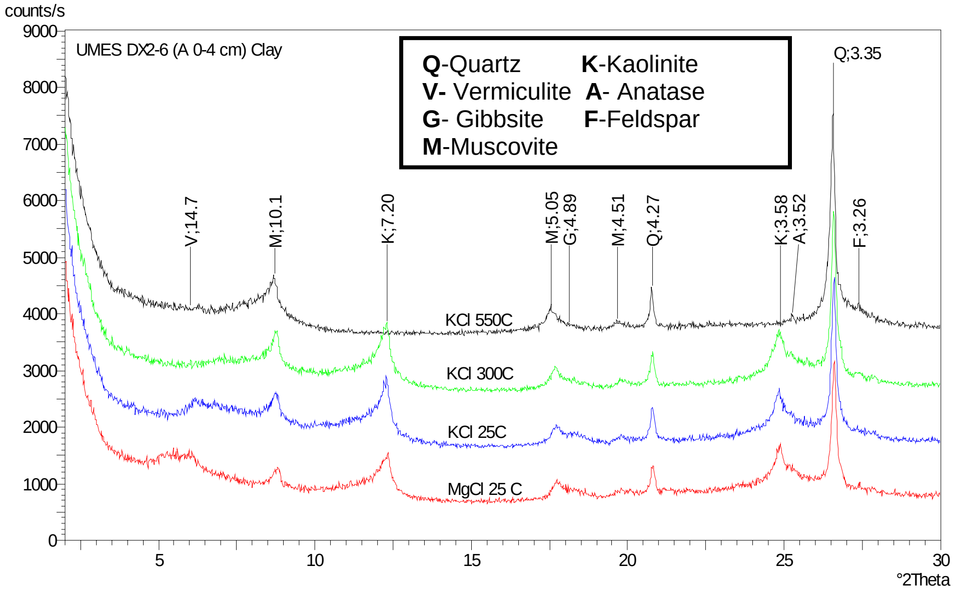


Figure 39. X-ray diffraction scan of DX2-6 (A 0-4 cm) clay fraction. CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 2° 2-theta to 30° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 15 minutes.

UMES DX1-2 (A3 8-20 cm) Clay

Mineral	Relative Abundance
Kaolinite	XXXX
Quartz	XX
Muscovite	XX
Vermiculite	X
Feldspars	tr
Gibbsite	tr

This sample was found to be high in kaolinite and quartz. The kaolinite peak in this scan was more intense than the previous scan (DX2-6) which might indicate a higher percentage of the sample was comprised of kaolinite. The loss of the kaolinite peaks upon heating to 550C is a fairly good indication of kaolinite. The strong peak at 5 nm again indicates the presence of muscovite, while the peaks at 14.1 nm indicate the presence of vermiculite. Feldspars and gibbsite were also present in very trace amounts.

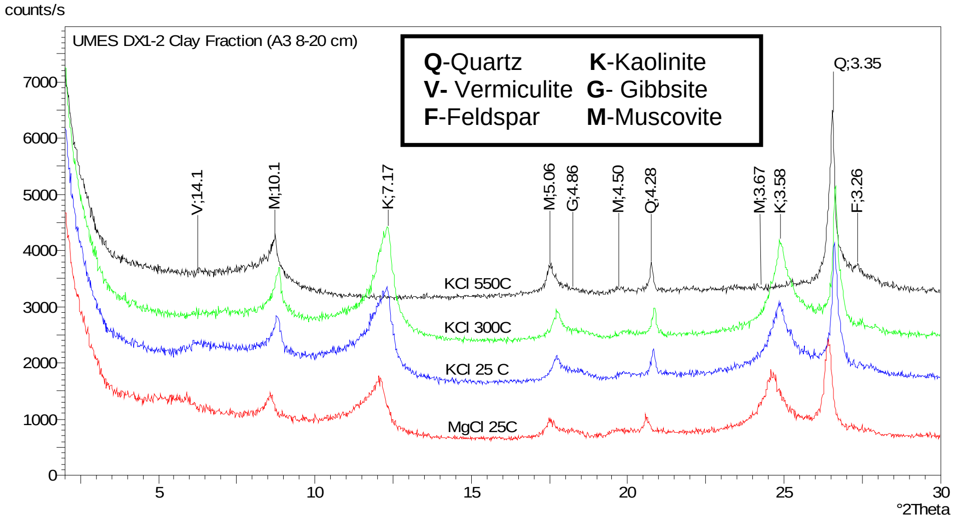


Figure 40. X-ray diffraction scan of DX1-2 (A3 8-20 cm) clay fractions. CuKα at 45 kilo-volts and 35 milli-ampheres. A continuous scan was run from 2° 2-theta to 30° 2-theta. The step size is 0.04° 2-theta per 2 seconds with a scan time around 15 minutes.

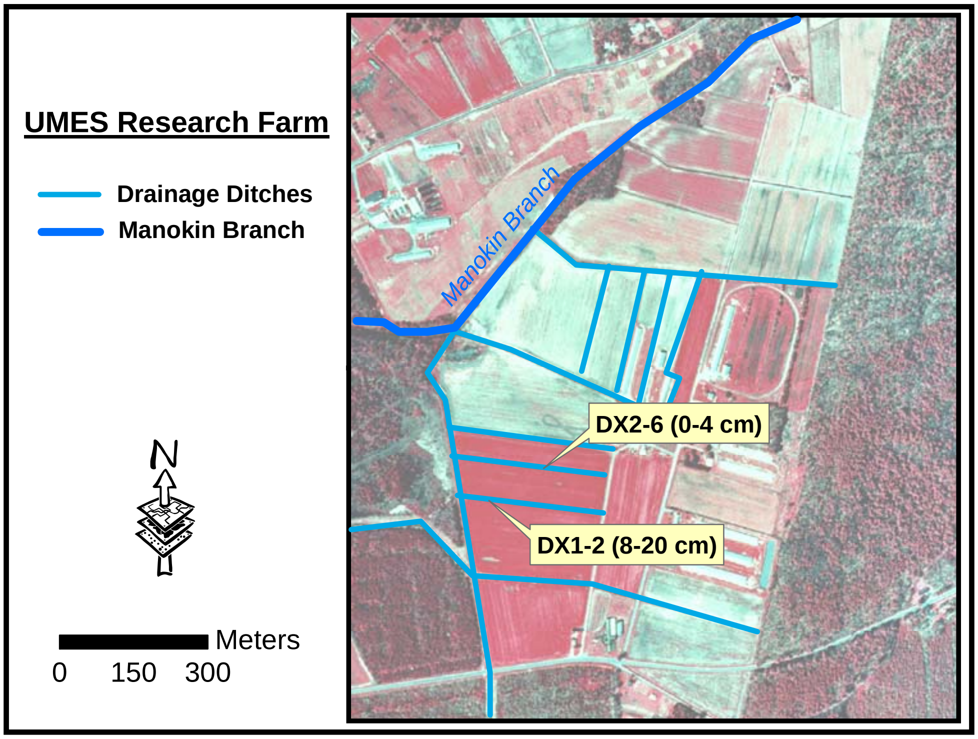


Figure 41. Map of XRD sample locations on the University of Maryland Research farm.

Thesis Conclusions

Materials found within agricultural drainage ditches at UMES can be and should be identified as soil. Pedogenesis has been shown to be occurring and layers within the material can be easily distinguished based on these mineral and organic alterations. Future investigations of the physical, chemical, and spatial properties of drainage ditch soils should classify and treat these materials as unique and distinguishable soil bodies.

Drainage ditch soils at UMES are high in P. Not only are the ditches high in P on the surface, but also in subsurface soils. Future studies should examine the mechanisms controlling P movement in, through, and out of soils in the drainage ditch network. The connection of drainage water quality measurements in connection to the nutrient status of ditch soils could yield insight into some of these mechanisms. The connection of P, redox cycles, sulfidic materials, and monosulfidic black oozes on the surface of drainage ditches is an important subject that also needs investigation.

The discovery of sulfidic materials at UMES is of particular concern, especially the fact that sulfidic materials can be found so close to the surface of drainage ditch soils. The original concept of the extent of the sulfidic materials at UMES did not go far beyond ditches DX1, DX2, and DX3. This was thought to be a minor inclusion in a large Othello map unit. However, as seen in this and previous chapters, sulfidic materials have been found everywhere on the farm. In addition, the formation of monosulfidic black oozes (MBO) on the surface of drainage ditches is also a concern.

The sulfidic materials at UMES can be found in many forms, which can be rather deceiving to the uninitiated. Near the outlet of DX3, sulfidic materials that are very dense, have a silty-clay texture and 10YR 4/1 colors are found within 20 cm of the ditch soil surface. In DXXD3, it is in the form of a sand, 1m in depth that is closer to 10YR 4/3 in color, but is equally as reactive to H_2O_2. Additionally, in DXXD2 and towards the head of DX1, pockets of a light greenish grey, 5G 8/1, silty clay can be found at less than 1 m, which are highly reactive to H_2O_2, and become very acidic after incubation. It is for this reason that future investigations of drainage ditches at UMES should evaluate every horizon with caution, and must be careful in assessing the potential for sulfidic materials at this site.

The Manokin Branch was straightened some years ago (~1930/40s?), and in doing so, sulfidic materials were exposed. The sulfidic materials are exposed on the stream bank and are below the water level even during the summer. The area where these have been identified is in a slight bend in the river at the bottom of the hill west of DX5. The potential for acidifying of stream waters, especially during stream bank erosional events during large storm events is great. This has the potential to be a major water quality problem, and must be addressed.

Ultimately, I believe that there is a need to do a full assessment of the forms, depth, extent, and distribution of sulfidic materials in the watershed. The UMES farm may simply be an anomaly in the fact that it has geologically deposited sulfidic materials. However, because the watershed is so heavily ditched, it is crucial to know the extent of the sulfidic materials prior to drainage ditch management decisions.

The formation of MBOs on drainage ditch soil surfaces are a concern to overlying water quality. They appear as a jet-black material with a soupy texture on the surface of drainage ditches. It can be mixed with coarse organic debris such as old grass and vegetation on the bottom of the drainage ditch. This material was originally thought to be decaying organic material. Indeed it is enriched with organics, but it contains monosulfides (FeS) that are very labile, can oxidize very quickly, and can strip essentially all oxygen out of the water when they do oxidize. This is not only a problem for living organisms, but may increase the losses of soluble P from the drainage network. The MBO appears to form after at least a week of saturation in the drainage ditches. It was not evident when drainage ditch descriptions were made because many of the drainage ditches were dry. The formation of MBOs in drainage ditches in the watershed should be examined more closely. The sources of sulfur, the mineralogy, and chemistry of the system also need to be studied.

Appendix A

Drainage Ditch Pedon Description

Ditch name and location: DX1-1	**Distance from outlet:** 0 m
Date: 7/15/04	**Depth of auger boring:** 100 cm
Time: 5:13 pm	**Water Depth in Ditch:** 0m

Notes: Surprised that alpha alpha is reactive on surface horizons and not lower. Guess is that GW is oxygenated, b/c of so many concentrations.

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**--**	**Type-%-Color-Size-Dist-Kind**
1	Ag1	5	10 YR	4	2	--	SiL	16	35	1 Gr	+	Conc,3%,10YR 4/4,fine,faint,pore linings Dep,2%,10YR 5/2,fine/med,faint,masses
2	A1	14	10 YR	4	3	--	SiL	16	35	1 Gr	+	Conc,3%,10YR 4/6,med,faint,masses
3	A’g2	23	10 YR	4	2	--	SL	9	70	1SBK	+	Conc,3%,10YR 4/6,med,faint,masses
4	2Cg1	50	2.5 Y	5	2	--	S	5	90	0 SGR	+	Conc,3%,10YR 5/6,med,dist,masses
5	2C1	65	2.5 Y	6	4	--	S	2	90	0 SGR	-	Conc,10%,7.5YR 5/8,coarse,prom,masses
6	2C2	90	10 YR	6	3	--	S	2	90	0 SGR	-	Conc,2%,10YR 6/6,med,dist,masses
7	2C4	100	10 YR	6	6	--	S	2	90	0 SGR	-	Conc,2%,10YR 5/8,med,dist.,masses

Drainage Ditch Pedon Description

Ditch name and location: DX1-2	**Distance from outlet:** 40 m
Date: 7/16/04	**Depth of auger boring:** 86 cm
Time: 3:30 pm	**Water Depth in Ditch:** 0 cm

Notes: Tested alpha alpha on shovel, was ok. Did not see any ground water when augering.

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**--**	**Type-%-Color-Size-Dist-Kind**
1	A1	2	10 YR	2	2	--	SiL	11	38	1 Gr	-	--
2	A2	8	10 YR	3	1	--	SiL	12	35	1 Gr	-	--
3	A3	20	10 YR	3	1	--	SiCL	29	19	0 SGR	-	Conc,2%,10YR 3/6,fine,dist,pore linings
4	2Cg1	36	2.5 Y	6	2	1% 1cm	LS	5	85	0 SGR	-	Conc,5%,10YR 6/8,med,prom ,masses
5	2C1	46	10 YR	6	6	--	S	2	95	0 SGR	-	Conc,2%,10YR 5/8,med,faint,masses Dep,2%,10YRS 7/2, fine/med,dist,masses
6	2C2	71	2.5 Y	7	3	--	S	2	95	0 SGR	-	Conc,3%,10YR 5/8,med,prom,masses Dep,3%,2.5Y 8/1,med,prom,masses
7	2Cg2	86	2.5 Y	6	2	--	S	2	95	0 SGR	-	Conc,1%,10YR 6/4,med,faint,masses

Drainage Ditch Pedon Description

Ditch name and location: DX1-3	**Distance from outlet:** 80 m
Date: 7/19/04	**Depth of auger boring:** 87 cm
Time: 4:20 pm	**Water Depth in Ditch:** 3 cm above surface
Easting:	**Notes:** Water deeper in some surrounding areas in ditch.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	A1	5	10 YR	3	2	2% <1cm	L	12	45	1 SBK	+	--
2	A2	15	10 YR	4	3	3% <1cm	S	3	95	0 SGR	+	--
3	Ag1	23	10 YR	4	2	5% <1cm	LS	6	85	0 SGR	+	--
4	A'3	30	10 YR	3	2	2% <1cm	SL	9	70	0 SGR	+	--
5	2Cg1	52	2.5 Y	5	1	1% <1cm	LS	5	80	0 SGR	+	Conc,1%,10YR 6/6,med,dist,masses
6	2Cg2	71	2.5 Y	5	2	1% <1cm	S	3	95	0 SGR	+	Conc,3%,7.5YR 5/8,med,prom,masses Conc,5%,10YR 6/6, med/coarse,prom,mass
7	2Cg3	87	2.5 Y	6	2	1% <1cm	S	3	95	0 SGR	+	Conc,2%,10YR 6/6,med,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX1-4	**Distance from outlet:** 120 m
Date: 7/19/04	**Depth of auger boring:** 68 cm
Time: 5:30 pm	**Water Depth in Ditch:** 2 cm above surface
Easting:	**Notes:** Auger sanded out; sand falling out of auger.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc- ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**--**	**Type-%-Color-Size-Dist-Kind**
1	A1	6	10 YR	3	2	--	SiL	13	35	1 GR	+	--
2	A2	18	10 YR	3	2	--	SCL	21	70	1 SBK	+	Conc,3%,10YR 4/6, fine,dist,pore linings Dep,3%,10YR 5/1,med,dist,masses
3	Ag1	24	10 YR	4	2	--	LS	5	83	1 SBK	+	Conc,3%,10YR 4/6, fine,dist,pore linings Dep,3%,10YR 5/1,med,dist,masses
4	A'3	32	10 YR	3	2	1% <1cm	SiL	16	30	1 SBK	+	Conc,3%,10YR 3/6,fine,dist,pore linings
5	2Cg1	38	10 YR	6	1	5% <1cm	S	3	94	0 SGR	+	--
6	2C1	68	10 YR	5	3	5% <1cm	COS	3	94	0 SGR	+	Conc,10%,10YR 5/8,med/coarse,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX1-5	**Distance from outlet:** 160 m
Date: 7/19/04	**Depth of auger boring:** 96 cm
Time: 5:55 pm	**Water Depth in Ditch:** 6 cm above surface
Easting:	**Notes:** Suspected acid sulfate soil at bottom of profile.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	Oi	3	10 YR	3	1	--	--	--	--	1 GR	+	--
2	Ag1	8	10 YR	4	2	--	SiL	10	38	1 GR	+	--
3	Ag2	12	10 YR	4	2	5% <1cm	SCL	22	70	1 GR	+	Conc,2%,7.5YR 3/4, very fine,dist,pore linings
4	Ag3	32	10 YR	4	1	3% <1cm	LS	8	85	0 SGR	+	Conc,3%,10YR 3/6,fine,dist,pore linings
5	2C1	43	10 YR	6	8	3% <1cm	S	3	95	0 SGR	+	--
6	2Cg1	57	10 YR	6	2	3% <1cm	COS	3	95	0 SGR	+	Conc,3%,10YR 4/6,med,dist,masses
7	2C’2	82	10 YR	6	3	--	S	3	95	0 SGR	+	Dep,1%,2.5Y 5/4,med,dist,masses
8	2C’3	96	10 YR	3	2	--	S	3	95	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX1-6	**Distance from outlet:** 200 m
Date: 7/19/04	**Depth of auger boring:** 93 cm
Time: 6:40 pm	**Water Depth in Ditch:** 7 cm above surface
Easting:	**Notes:** Did not hit any suspected acid sulfate soil.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	Oi	4	10 YR	3	1	--	--	--	--	--	+	--
2	Ag1	18	10 YR	4	2	--	SCL	22	60	1 GR	+	--
3	A1	27	10 YR	3	1	5% <1cm	SL	10	83	1 GR	+	Conc,2%,10 YR 6/2, med,dist,masses
4	2C1	46	10 YR	6 6	2 25% 6 55%	--	S	3	95	0 SGR	-	Conc,5%,10YR 5/8,med,dist,masses
5	2C2	64	10 YR	6	4	--	S	3	95	0 SGR	+	Conc,5%,7.5YR 5/8,med,dist,masses Dep,5%,10YR 5/2,med,dist,masses Conc,2%,10YR 4/6,med,dist,masses
6	2C3	78	10 YR	5	3	2% <1cm	S	3	95	0 SGR	+	
7	2Cg1	93	10 YR	5	2	2% <1cm	S	3	95	0 SGR	+	

Drainage Ditch Pedon Description

Ditch name and location: DX1-7	**Distance from outlet:** 240 m
Date: 7/19/04	**Depth of auger boring:** 103 cm
Time: 7:11 pm	**Water Depth in Ditch:** At surface
Easting:	**Notes:** 10% green clayey masses in last horizon, not sure if acid sulfate soil. No “zone of concentrations” or no horizon with many concentrations.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Structure**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	Ag1	5	10 YR	4	1	--	SiL	10	38	1 GR	-	--
2	Ag2	15	10 YR	4	2	--	SL	13	75	1 GR	+	--
3	Ag3	29	10 YR	4	1	3% <1cm	SCL	21	70	1 GR	+	Conc,2%,10 YR 6/2, med,dist,masses
4	2Cg1	44	10 YR	5	1	--	LS	7	87	0 SGR	+	Conc,5%,10YR 5/8,med,dist,masses
5	2Cg2	57	10 YR	5	2	--	LS	8	85	0 SGR	-	Conc,5%,7.5YR 5/8,med,dist,masses Dep,5%,10YR 5/2,med,dist,masses Conc,2%,10YR 4/6,med,dist,masses
6	2Cg3	70	10 YR	5	1	5% <1cm	SCL	26	70	0 MA	-	--
7	2Cg4	81	10 YR	4	1	--	SL	14	73	0 SGR	+	--
8	2C’1	89	10 YR	5	3	--	LS	8	87	--	+	--
9	2C’g 5	103	10 YR	4	2	--	LS	8	87	--	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX1-8	**Distance from outlet:** 280 m
Date: 7/20/04	**Depth of auger boring:** 104 cm
Time: 8:30 am	**Water Depth in Ditch:** No water subsurface flow between new and native soil interface
Easting:	**Notes:** Subsurface flow between new and native soil interface
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	A1	7	10 YR	2	2	1% <1cm	SiL	14	35	2 SBK	+	--
2	A2	16	10 YR	2	2	--	SiCL	28	18	1 GR	+	--
3	2Cg1	25	10 YR	6	2	1% <1cm	LS	5	80	0 SGR	+	--
4	2Cg2	45	2.5 Y	5	2	--	SCL	23	75	0 MA	+	Conc,3%,7.5YR 5/6,med,prom,masses Conc,2%,10YR 6/8,coarse,dist,masses
5	2Cg3	59	10 YR	6	1	--	SCL	28	65	0 MA	+	Conc,2%,10YR 6/8,med/coarse,dist,masses
6	2Cg4	78	2.5 Y	5	1	--	SCL	22	75	0 MA	+	Dep,3%,10YR 8/1,med,prom,masses
7	2C1	93	10 YR	5	3	--	S	8	89	0 SGR	+	Conc,5%,7.5YR 5/8,med,prom,masses Dep,5%,7.5YR 6/1,med,dist,masses
8	2C’g 5	104	10 YR	5	2	20% <1cm	S	3	90	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX2-1	**Distance from outlet:** 0 m
Date: 7/20/04	**Depth of auger boring:** 107 cm
Time: 9:20 am	**Water Depth in Ditch:** Water at soil surface
Easting:	**Notes:** Profile a little stretched out, ASS at <1 m
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Structure**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	A1	5	10 YR	3	2	--	SiL	12	35	1 GR	+	--
2	A2	15	10 YR	3	2	1% <1cm	SiL	14	55	1 GR	+	Conc,5%,7.5YR 3/4,fine,faint,porelinings
3	2C1	21	10 YR	5	3	1% <1cm	LS	6	85	1 SBK	-	Conc,5%,10YR 4/6,fine/med,prom,masses
4	2Cg1	35	2.5 Y	7	1	--	LS	5	85	0 SGR	+	Conc,5%,10YR 6/8,med,prom,masses Dep,1%,10YR 8/1,med,faint,masses
5	2Cg2	45	2.5 Y	5	2	--	S	3	90	0 SGR	+	Conc,3%,7.5YR 5/8,med,dist,masses
6	2Cg3	76	10 YR	6	2	--	S	3	90	0SGR	+	Conc,5%,7.5YR 5/8,med,prom,masses Conc,5%,10YR6/6,med/coarse, prom,masses
7	2Cg4	91	10 YR	4	1	--	S	5	80	0 SGR	+	ASS
8	2C'2	107	10 YR	3	1	--	C	45	35	0 MA	+	ASS

Drainage Ditch Pedon Description

Ditch name and location: DX2-2	**Distance from outlet:** 40 m
Date: 7/20/04	**Depth of auger boring:** 110 cm
Time: 10:40 am	**Water Depth in Ditch:** Water at 6 cm above soil surface
Easting:	**Notes:** ASS at bottom, C/A horizon 10YR 5/2 sand w/ krotovinas, of above material 10YR 3/2.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc- ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	5	10 YR	3	2	--	SiL	12	35	1 GR	+	Conc,2%,10YR 3/6,fine,faint,porelinings
2	A2	15	10 YR	3	2	--	SiL	15	35	2 GR	+	Conc,5%,5YR 3/4,med,dist,ped faces
3	Cg/A	23	10 YR 10 YR	3 5	2(45%) 2(55%)	--	SL	11	80	1 GR	+	Conc,5%,10YR 4/6,coarse,distinct,masses Conc,5%,7.5YR 3/4,med,distinct,masses
4	2Cg1	33	2.5 Y	6	2	--	S	3	92	0 SGR	+	Conc,10%,10YR 4/4,med/coarse,dist,masses Conc,5%,10YR 3/4,med,dist,masses
5	2Cg2	43	2.5 Y	6	1	--	S	3	92	0 SGR	+	Dep,5%,2.5YR 7/1,med,faint,masses
6	2Cg3	71	10 YR	6	2	--	S	3	92	0 SGR	+	Conc,10%,5YR 4/6,med,prom,masses Conc,5%,5YR5/8,med,prom,masses
7	2C1	98	10 YR	6	3	--	S	3	92	0 SGR	+	Conc,5%,5YR 4/6,med,prom,masses Conc,5%,10YR4/6,med,prom,masses
8	2C2	110	10 YR	3	1	--	S	5	92	0 SGR	+	ASS

Drainage Ditch Pedon Description

Ditch name and location: DX2-3	**Distance from outlet:** 80 m
Date: 7/20/04	**Depth of auger boring:** 87 cm
Time: 11:30 am	**Water Depth in Ditch:** Water at 6 cm above soil surface
Easting:	**Notes:**
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	7	10 YR	3	2	--	SiL	12	35	1 GR	+	Conc,2%,10YR 3/6,fine,faint,porelinings
2	A2	22	7.5 YR	3	2	1% 1cm	SiL	15	35	2 GR	+	Conc,5%,5YR 3/4,med,dist,ped faces
3	2C1	41	10 YR	6	3	--	SL	11	83	1 GR	+	Conc,5%,10YR 4/6,coarse,distinct,masses Conc,5%,7.5YR 3/4,med,distinct,masses
4	2C2	63	10 YR	6	3	--	S	3	92	0 SGR	+	Conc,10%,10YR 4/4,med/coarse,dist,masses Conc,5%,10YR 3/4,med,dist,masses
5	2C3	82	10 YR	5	8	--	S	3	92	0 SGR	+	Dep,5%,2.5YR 7/1,med,faint,masses
6	2Cg1	87	2.5 Y	5	1	--	S	3	92	0 SGR	+	Conc,10%,5YR 4/6,med,prom,masses Conc,5%,5YR5/8,med,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX2-4	**Distance from outlet:** 120 m
Date: 7/20/04	**Depth of auger boring:** 93 cm
Time: 12:00 pm	**Water Depth in Ditch:** Water at 2 cm above soil surface
Easting:	**Notes:**
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	4	10 YR	3	2	--	SiL	12	35	1 GR	+	--
2	A2	16	10 YR	3	2	--	SiCL	27	19	1 GR	+	Conc,2%,10YR 4/6,fine,dist,porelinings
3	2Cg1	29	10 YR	4	2	10% <1cm	S	5	90	0 SGR	+	Dep,2%,10YR 6/1,med,distinct,masses At base of horizon
4	2C1	55	10 YR	5	4	--	S	3	93	0 SGR	+	Conc,3%,10YR 4/6,med,dist,masses Conc,3%,10YR 5/8,med,faint,masses
5	2C’g2	69	10 YR	6	2	--	S	3	93	0 SGR	+	Dep,5%,5YR 5/8,med,prom,masses
6	2C’2	93	10 YR	6	3	--	S	3	93	0 SGR	+	Conc,5%,10YR 5/8,med,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX2-5	**Distance from outlet:** 160 m
Date: 7/20/04	**Depth of auger boring:** 90 cm
Time: 1:30 pm	**Water Depth in Ditch:** Water at 25 cm below soil surface
Easting:	**Notes:**
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	3	10 YR	3	2	--	mSiL	10	25	1 GR	+	--
2	A2	9	10 YR	3	2	1% <1cm	SL	8	65	1 SBK	+	--
3	Ag1	24	10 YR	4	2	5% <1cm	SL	8	65	1 SBK	+	--
4	2C1	58	10 YR	5	3	2% <1cm	LS	6	83	0 SGR	-	Conc,5%,10YR 6/4,coarse,prom,masses
5	2C2	80	10 YR	6	6	3% <1cm	LS	6	83	0 SGR	+	Conc,10%,10YR 5/8,med,prom,masses Conc,10%,10YR 6/6,coarse,prom,masses
6	2Cg1	90	10 YR	7	2	3% <1cm	S	2	95	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX2-6	**Distance from outlet:** 200 m
Date: 7/20/04	**Depth of auger boring:** 75 cm
Time: 2:00 pm	**Water Depth in Ditch:** Water at 6 cm below soil surface
Easting: **Northing:**	**Notes:**

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	4	10 YR	3	2	--	mSiL	10	25	1 GR	+	--
2	Ag2	18	10 YR	4	2	1% <1cm	SiCL	24	15	2 GR	+	Conc,2%,5YR 4/6,fine,dist,porelinings
3	Ag3	23	10 YR	4	2	1% <1cm	SiL	13	30	1 GR	+	--
4	2C1	49	10 YR	5	3	3% <1cm	S	3	87	0 SGR	+	Conc,2%,10YR 5/8,med,prom,masses
5	2Cg1	67	10 YR	5	2	--	LS	5	85	0 SGR	+	--
6	2C’2	75	10 YR	6	3	--	FS	3	90	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX2-7	**Distance from outlet:** 240 m
Date: 7/20/04	**Depth of auger boring:** 75 cm
Time: 2:45 pm	**Water Depth in Ditch:** Water at 6 cm below soil surface
Easting:	**Notes:** Surprised at lack of redox features in this profile. DX2-6 had little redox as well.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Structure**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	6	10 YR	3	2	--	SiL	15	35	1 GR	+	--
2	Ag1	12	10 YR	4	2	--	SiL	15	35	1 SBK	+	Conc,3%,5YR 3/4,fine,dist,porelinings
3	Ag2	18	10 YR	4	2	3% <1cm	L	10	47	1 SBK	+	Conc,3%,5YR 3/4,fine,dist,porelinings
4	2C1	29	10 YR	5	4	25% <1cm	S	2	95	0 SGR	+	--
5	2Cg1	40	2.5Y	4	1	5% <1cm	SL	8	77	0 SGR	-	--
6	2Cg2	53	2.5Y	5	1	--	SL	10	75	0 SGR	-	--
7	2Cg3	75	2.5Y	5	2	--	SL (sticky)	15	70	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX2-8	**Distance from outlet:** 280 m
Date: 7/20/04	**Depth of auger boring:** 75 cm
Time: 3:30 pm	**Water Depth in Ditch:** Water at 6 cm below soil surface
Easting:	**Notes:** Last description for DX2.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	Oi	3	10 YR	3	2	--	--	--	--	--	+	--
2	Ag1	7	10 YR	4	2	--	SiL	10	35	1 SBK	+	--
3	Ag2	18	10 YR	4	2	--	SiL	12	35	1 SBK	+	--
4	2Cg1	31	10 YR	5	2	5% <1cm	S	3	90	0 SGR	+	--
5	2Cg2	48	10 YR	6	1	3% <1cm	S	3	95	0 SGR	+	Conc,10%,7.5YR 5/8,med/coarse,prom,masses Dep,5%,2.5Y 5/1,med,dist,masses
6	2C1	60	10 YR	5	3	--	LS	5	83	0 SGR	+	Conc,15%,7.5YR 5/8,coarse,prom,masses
7	2C2	75	10 YR	6	3	--	S	3	95	0 SGR	+	Conc,10%,2.5Y 5/6,coarse,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX3-2

Distance from outlet: 40 m

Date: 7/21/04

Depth of auger boring: 83 cm

Time: 10:06 am

Water Depth in Ditch: 15 cm below soil surface

Easting:

Northing:

Notes: ASS soil very close to surface, @ 28 cm.

Horizon		Depth (cm)	Color			Texture				Struc- ture	Alpha Alpha	Redoximorphic Features
--	--	--	Hue	Value	Color	Rock	Class	Clay	Sand	--	+/-	Type-%-Color-Size-Dist-Kind
1	A1	9	10 YR	3	1	--	mSiL	8	30	1 GR	+	--
2	A2	17	10 YR	3	1	--	SiL	12	35	1 SBK	+	--
3	2Cg1	28	10 YR	6	1	--	SCL	24	55	0 MA	+	Conc,20%,10 YR 5/8,coarse,prom,masses Conc,5%,7.5YR 4/6,med,dist,masses
4	2Cg2	50	10 YR	4	1	--	C	45	30	0 MA	+	Conc,5%,5YR 4/6,coarse,prom,ped faces
5	2Cg3	70	10 YR	4	1	--	LS	7	83	0 SGR	+	Conc,10%,10YR 4/6,med,prom,masses Conc,5%,10YR 4/6,fine,dist,masses
6	2Cg4	83	10 YR	4	1	--	C	45	30	0 MA	+	Conc,1%,7.5 YR 3/3,fine/med,dist,ped faces

Drainage Ditch Pedon Description

Ditch name and location: DX3-3

Distance from outlet: 80 m

Date: 7/21/04

Depth of auger boring: 88 cm

Time: 10:45 am

Water Depth in Ditch: 15 cm below soil surface

Easting:

Northing:

Notes: No ASS present?

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	7	10 YR	3	1	--	SiL	9	35	1 GR	+	--
2	A2	18	10 YR	3	1	--	SiL	12	35	1 GR	+	Dep,3%,10YR 5/1,fine/med,dist,masses
3	2Cg1	40	10 YR	6	1	--	FS	10	90	0 SGR	+	--
4	2Cg2	62	10 YR	6	2	--	S	5	90	0 SGR	+	Conc,5%,7.5YR 4/6,med/coarse,prom,masses Conc,5%,10YR 5/8,med/coarse,prom,masses
5	2C1	87	10 YR	6	3	--	S	5	90	0 SGR	+	Conc,5%,10YR 6/6,coarse,prom,masses Conc,5%,10YR 4/6,med,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX3-4	**Distance from outlet:** 120 m
Date: 7/21/04	**Depth of auger boring:** 74 cm
Time: 11:15 am	**Water Depth in Ditch:** 15 cm below soil surface
Easting:	**Notes:** Profile short due to sands collapsing
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**+/-**	**Type-%-Color-Size-Dist-Kind**
1	A1	7	10 YR	3	2	--	mSiL	8	25	0 (10% co fibers)	+	--
2	A2	14	10 YR	3	1	--	L	10	45	1 GR	+	--
3	2Cg1	29	10 YR	5	1	--	S	4	90	0 SGR	+	--
4	2C1	65	2.5Y YR	6	4	--	S	4	90	0 SGR	+	Conc,35%,10YR 7/6,very coarse,prom,masses
5	2C2	74	10 YR	5	8	--	S	4	90	0 SGR	+	Dep,3%,10YR 6/3,coarse,prom,masses Horizon is one large concentration

Drainage Ditch Pedon Description

Ditch name and location: DX3-5

Distance from outlet: 160 m

Date: 7/21/04

Depth of auger boring: 101 cm

Time: 11:45 pm

Water Depth in Ditch: 0 cm **,** at surface

Easting:

Northing:

Notes: Mucky surface, very wet. Not as many roots, but no grass growing here, more bushes with thorns (don't know species) ASS soil Horizon #6

Horizon		**Depth (cm)**	**Color**			**Texture**				**Structure**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**--**	**Type-%-Color-Size-Dist-Kind**
1	A1	7	10 YR	2	1	--	mSiL	10	25	1 Gr	+	--
2	A2	18	10 YR	3	1	--	SiL	12	25	1 SBK	+	--
3	2C1	41	2.5 Y	6	6	5% 1cm	S	5	80	0 SGR	-	Conc,3%,10YR 4/6,med,dist,masses Whole horizon is oxidized
4	2Cg1	69	2.5 Y	6	2	5% 1cm	S	5	80	0 SGR	+	--
5	2Cg2	101	10 YR	4	1	--	S	5	80	0 SGR	+	Conc,3%,10YR 5/2,med,prom,masses Dep,2%,7.5YR 4/4, fine, prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX3-6	**Distance from outlet:** 200 m
Date: 7/21/04	**Depth of auger boring:** 97 cm
Time: 12:00 pm	**Water Depth in Ditch:** 30 cm below soil surface
Easting:	**Notes:** No oxidized horizon, Hypothesis is that ASS is a source of Fe to create horizon. Many fine/med roots in surface horizons due to grass growing in ditch.
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	**--**	**--**	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	**--**	**--**	**Type-%-Color-Size-Dist-Kind**
1	Oi	4	10 YR	3	2	--	--	--	--	--	+	--
2	A1	11	10 YR	3	1	--	SiL	10	30	1 GR	+	--
3	Ag2	24	10 YR	4	1	--	SiL	11	35	1 SBK	+	Conc,10%,10YR 3/6,fine,dist,porelinings
4	2Cg1	42	10 YR	5	1	5% 1cm	S	4	90	0 SGR	+	Conc,15%,10YR 4/6,med/coarse,prom,masses
5	2Cg2	58	10 YR	6	1	5% 1cm	S	4	90	0 SGR	+	Conc,2%,10YR 4/6,med,prom,masses
6	2Cg3	83	10 YR	5	2	--	S	4	90	0 SGR	+	Dep,3%,10YR 7/1,med,dist,masses Conc,2%,10YR 5/8,fine/med,prom,masses
7	2Cg4	97	10 YR	5	2	--	S	4	90	0 SGR	+	Conc,5%,10YR 6/6,med,prom,masses

Drainage Ditch Pedon Description

Ditch name and location: DX3-7	**Distance from outlet:** 240 m
Date: 7/21/04	**Depth of auger boring:** 91 cm
Time: 1:30 pm	**Water Depth in Ditch:** 30 cm below soil surface
Easting:	**Notes:** ASS soil horizons # 5 and 6
Northing:	

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	A1	5	10 YR	3	1	--	SiL	10	35	1 GR	+	--
2	Ag2	13	10 YR	4	1	--	SiL	12	35	1 GR	+	--
3	2C1	30	10 YR	5	3	--	S	4	90	0 SGR	+	Conc,3%,2.5Y 6/8,med,prom,masses Dep,3%,2.5Y 7/1,med,prom,masses
4	2C2	60	10 YR	5	4	--	S	4	90	0 SGR	-	Conc,5%,10YR 5/8,med/coarse,prom,masses Conc,5%,10YR 6/6,med,prom,masses
5	2Cg1	76	10 YR	4	1	2% 1cm	S	4	90	0 SGR	+	Dep,2%,10YR 5/2,med,faint,masses
6	2Cg2	91	10 YR	4	1	5% 1cm	S	4	90	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX3-8	**Distance from outlet:** 280 m
Date: 7/21/04	**Depth of auger boring:** 87 cm
Time: 2:00 pm	**Water Depth in Ditch:** 6 cm below soil surface
Easting: **Northing:**	**Notes:** Some horizons were not reacting to alpha-alpha Horizon #9 is a potential ASS

Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	A1	6	10 YR	2	1	--	L	10	45	1 GR	+	--
2	A2	11	10 YR	3	1	--	SiL	12	35	1 SBK	+	--
3	Ag3	18	10 YR	4	2	--	SiL	12	35	1 SBK	-	Conc,3%,7.5YR 4/6,fine,dist,masses
4	2Cg1	41	2.5 Y	6	1	--	S	4	90	0 SGR	-	Conc,10%,10YR 4/6,coarse,prom,masses Conc,2%,10YR 2/1,med,prom,organic? Conc,2%,10YR 6/6,med,prom,masses
5	2Cg2	47	10 YR	5	1	5% 1cm	S	4	90	0 SGR	-	Conc,5%,2.5Y 7/6,med,prom,masses Dep,2%,2.5Y 7/1,med,prom,masses
6	2Cg3	55	2.5 Y	6	2	5% 1cm	S	4	90	0 SGR	-	--
7	2C1	67	2.5 Y	6	3	3% 1cm	S	3	90	0 SGR	+	Conc,7%.5YR 4/6,med/coarse,prom,masses Conc,5%,5YR 5/6,med/coarse,prom,masses
8	2C'g 4	76	10 YR	5	1	3% 1cm	S	3	90	0 SGR	--	Conc,2%,7.5YR 5/6,med,prom,masses
9	2C'g 5	87	10 YR	4	1	2% 1cm	S	3	90	0 SGR	--	Conc,3%,5YR 4/6,med,prom,masses ASS

Drainage Ditch Pedon Description

Ditch name and location: DX5-1	**Distance from outlet:** 0 m
Date: 7/21/04	**Depth of auger boring:** 103 cm
Time: 3:04 pm	**Water Depth in Ditch:** 30 cm below soil surface
Easting:	**Notes:** ASS looking material at 63 cm
Northing:	

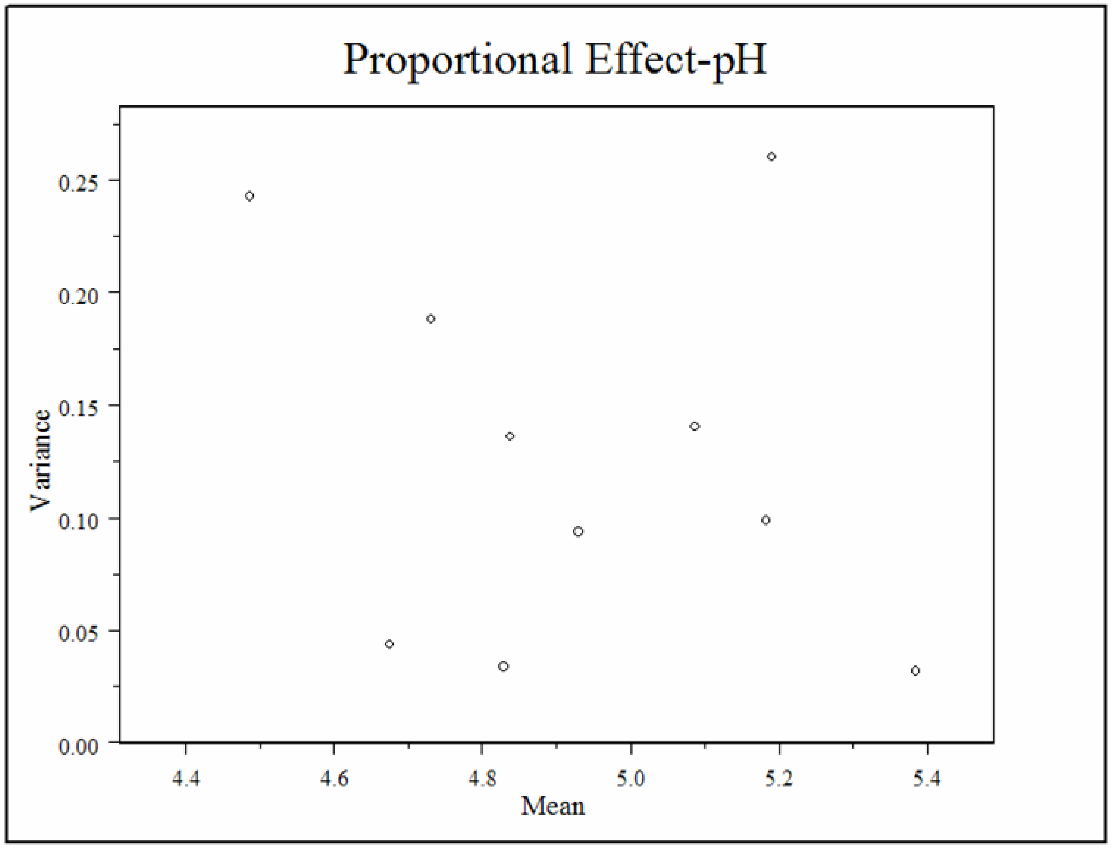
Horizon		**Depth (cm)**	**Color**			**Texture**				**Struc-ture**	**Alpha Alpha**	**Redoximorphic Features**
--	--	--	**Hue**	**Value**	**Color**	**Rock**	**Class**	**Clay**	**Sand**	--	--	**Type-%-Color-Size-Dist-Kind**
1	A1	7	10 YR	2	1	--	mSiL	8	35	1 SBK	-	--
2	A2	12	10 YR	2	1	--	SiL	10	35	1 SBK	+	--
3	Ag3	22	10 YR	4	2	3% 1cm	SiL	15	35	1 SBK	+	Conc,2%,7.5YR 4/6,fine,faint,porelinings and ped faces
4	Ag4	31	10 YR	4	2	10% 1cm	SL	13	60	1 SBK	+	Conc,4%,7.5YR4/4,fine/med,dist,porelinigs and ped faces
5	A'3	41	10 YR	4	3	2% 1cm	SL	9	65	0 SGR	+	Conc,3%,7.5YR 5/6,fine/med,dist,masses
6	2C1	48	2.5 Y	5	3	2% 1cm	FS	5	90	0 SGR	+	Conc,5%,10YR 5/6,med,prom,masses
7	2Cg1	63	10 YR	4	1	--	S	3	90	0 SGR	+	Conc,5%.10YR 5/6,med,prom,masses Conc,5%,10YR 4/6,med,prom,masses
8	2Cg2	103	2.5 Y	4	1	--	S	3	90	0 SGR	+	--

Drainage Ditch Pedon Description

Ditch name and location: DX5-3	**Distance from outlet:** 80 m
Date: 8/22/03	**Depth of auger boring:** 166 cm
Time: 7:45 pm	**Water Depth in Ditch:** 19 cm below soil surface after 1 hour
Easting:	**Notes:** Roots A1-15% fine; A2- 10% fine, no alpha-alpha so could not test profile.
Northing:	

Horizon		Depth (cm)	Color			Texture				Struc-ture	Alpha Alpha	Redoximorphic Features
--	--	--	Hue	Value	Color	Rock	Class	Clay	Sand	--	--	Type-%-Color-Size-Dist-Kind
1	Ag1	5	10 YR	4	2	--	SiL	18	35	1 SBK	-	--
2	Ag2	18	10 YR 2.5Y	4 4	3 2	--	SiL	23	35	1 SBK	-	Conc,1%,10YR 4/6,fine,dist,porelinings
3	BC1	30	10 YR	4	3	--	SiL	24	35	1 SBK	-	Conc,5%,10YR 3/6,fine,faint,porelinings
4	BC2	44	2.5 Y	4	3	2% 1cm	SiL	25	35	1 SBK	-	Conc,5%,10YR 4/6,fine/med,dist,porelinings Dep,3%,10YR 5/1,med,dist,masses
5	2C1	74	2.5 Y	5	4	3% 1cm	SL	13	60	0 SGR	-	Conc,5%,10YR 5/6,med,prom,masses Conc,5%,10YR 4/6,med,prom,masses Dep,10%,10YR 6/2,med/coarse,dist,masses
6	2Cg1	114	2.5Y	6	1	5% 1cm	SL	12	70	0 SGR	-	Conc,5%,10YR 6/6,fine,prom,masses Conc,8%,2.5Y 6/6,med,faint,masses
7	2C’2	135	10 YR	4	4	35% 1cm	FSL	11	65	0 SGR	--	Conc,5%,10YR 5/6,med,prom,masses Conc,5%,10YR 5/6,med,prom,masses
8	2C’3	151	10 YR	5	3	13% 1cm	FS	5	90	0 SGR	--	Conc,10%,10YR 5/6,coarse,prom,masses
9	2C’g2	166	10 YR	4	2	--	FS	3	90	0 SGR	--	Conc,3%,10YR 5/6,coarse,prom,masses

Appendix B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Variance
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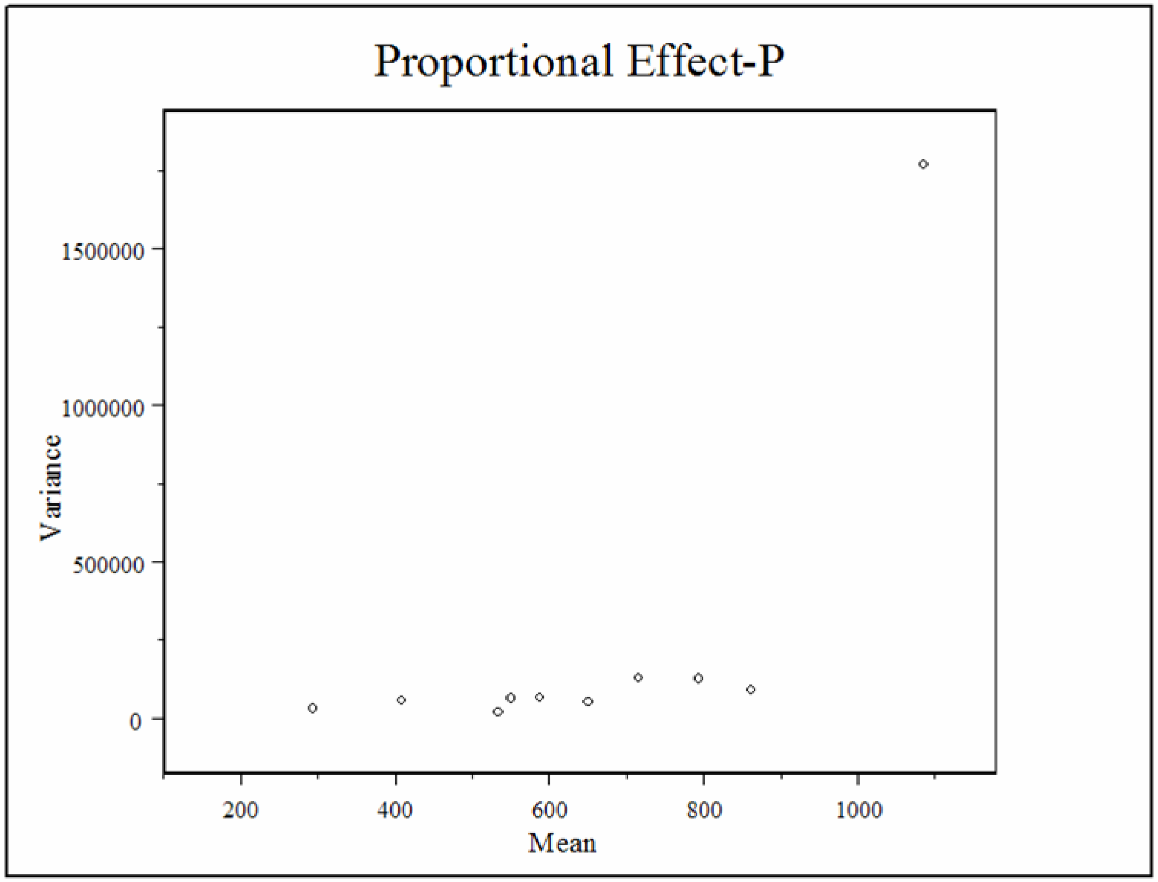
0.05
0.10
0.15
0.20
0.25
Mean
4.4
4.6
4.8
5.0
5.2
5.4


Proportional Effect-P
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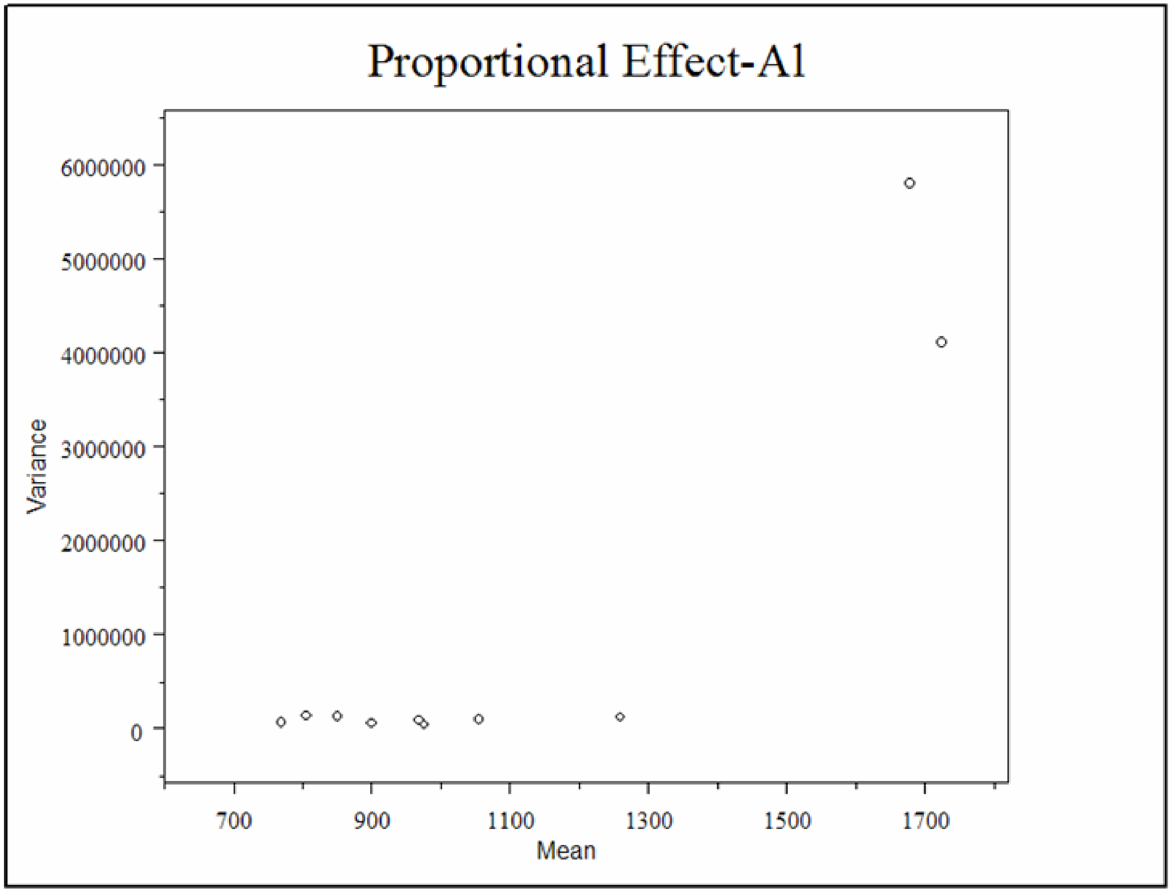
1000000
500000
0
Variance
200
400
600
800
1000
Mean


Proportional Effect-A1
Variance
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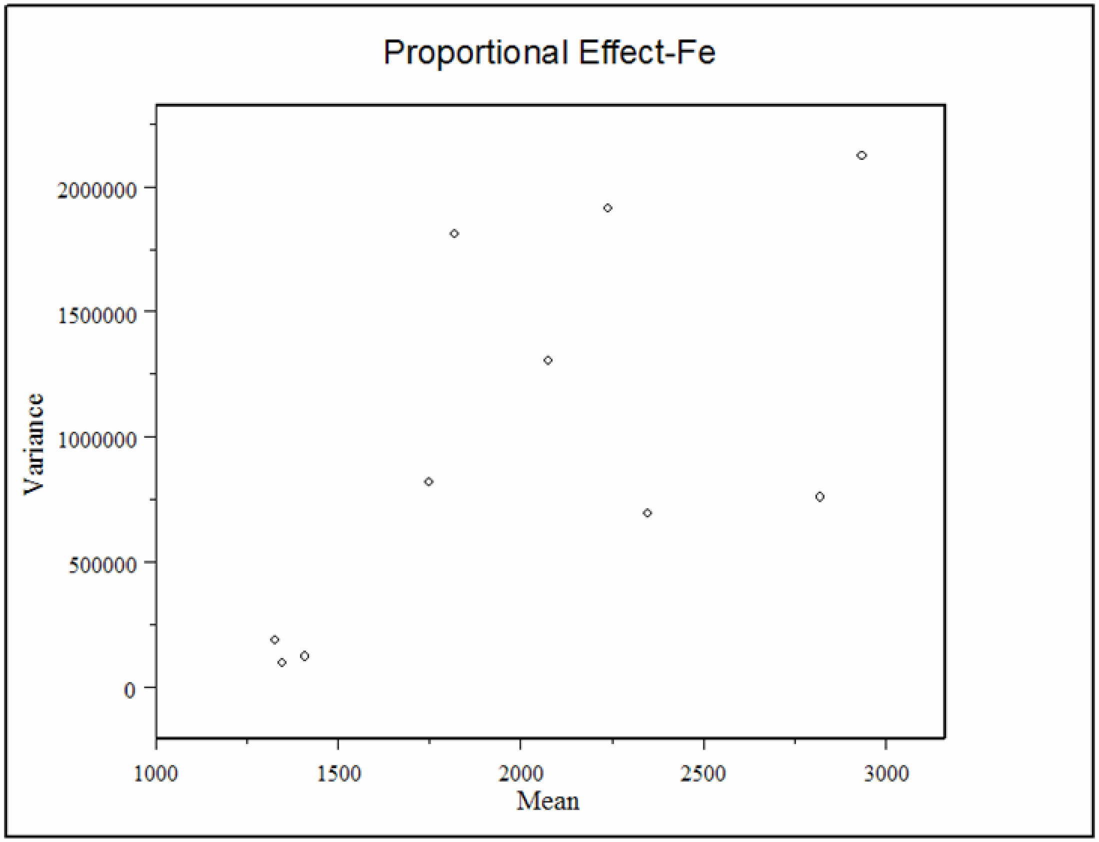
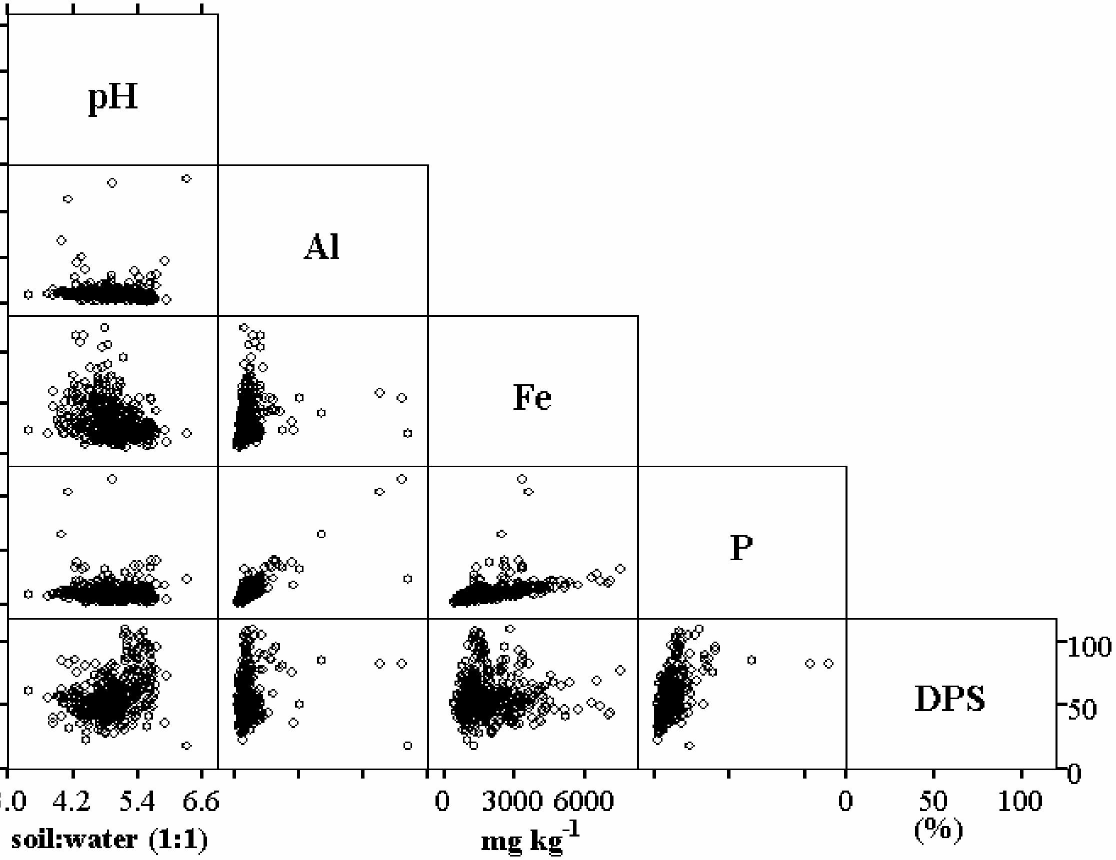
5000000
4000000
3000000
2000000
1000000
0
700
900
1100
1300
1500
1700
Mean


Proportional Effect-Fe
Variance
2000000
1500000
1000000
500000
0
1000
1500
2000
2500
3000
Mean


pH
Al
Fe
P
DPS
mg kg-1
15000
10000
5000
0
mg kg-1
6000
3000
0
100
50
0
3.0
4.2
5.4
6.6
soil:water (1:1)
0
3000
6000
mg kg-1
0
50
100
(%)

Appendix C

Algae in a drainage ditch at UMES

Algae in a drainage ditch at UMES

Evidence of bioturbation; crawfish hole in ditch at UMES.

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