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The science and policy that compels the wetland mitigation of phosphate-mined lands

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The Science and Policy that Compels the Wetland Mitigation
of Phosphate-Mined Lands

by

Nejma Danielle Piagentini

A thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science
Department of Biology
College of Arts and Sciences
University of South Florida

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List of Acronyms

BMR	Florida Department of Environmental Protection Bureau of Mine Reclamation
CARL	Florida's Conservation and Recreation Lands Program
CDA	Coordinated Developed Area
CE	United States Corps of Engineers
CWA	FWS Clean Water Act
EMAS	European Union Eco-Management and Audit Scheme
EPA	United States Environmental Protection Agency
ERP	Environmental Resource Permit
EU	European Union
EWRA	The Emergency Wetlands Resources Act
FDEP	Florida Department of Environmental Protection
FWS	United States Fish and Wildlife Service
GATT	General Agreement of Tariffs and Trade
HGM	Hydrogeomorphic Approach
IFA	International Fertilizer Industry
IHN	The Integrated Habitat Network
IIED	International Institute for Environmental and Development
IWWR	Interagency Workgroup on Wetland Restoration
MMPA	Marine Mammal Protection Act
MOA	Memoranda of Agreement
NAFTA	North American Free Trade Agreement
NMFS	United States National Marine Fisheries Service
NRC	National Research Council
NRCS	Natural Resource Conservation Service
RPMC	Resource Planning and Management Committees
U.S.	United States
UNEP	United Nations Environment Program
VNIRS	Visible-near infrared reflectance spectroscopy
WET	Wetland Evaluation Technique
WRP	Wetlands Reserve Program
WWF	The World Wide Fund for Nature

The Science and Policy that Compel the Wetland Mitigation of Phosphate-Mined Lands

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ABSTRACT

The State of Florida ranks fifth in the world's production of phosphate. The phosphate industry relies on surface mining to withdraw the phosphate ore, and this process can devastate the natural environment. One of the most impacted natural resources is wetlands. Federal laws permit the legal destruction of wetlands providing their loss is compensated by the mitigation (i.e., the restoration, creation, or enhancement) of other wetlands, but the complexity of wetland ecosystems makes the mitigation process difficult. One of the goals of this thesis is to review the established Federal, State and non-regulatory guidelines for the development and maintenance of mitigated wetlands, evaluate their efficacy and present some underlying reasons for successful versus unsuccessful mitigation projects.

The environmental repercussions of phosphate mining are not only pertinent to Florida or the United States. Wetland mitigation has become a global issue. Laws and programs that facilitate specific countries do not benefit wetland ecosystems on a landscape level. It is important to remain cognizant of the ramifications of wetland destruction and avoid piecemeal solutions to a wide-spread problem. Thus, my second objective is to investigate the progress and status of international wetland preservation. I will examine how different countries and international organizations are addressing the environmental impacts of mining, and underscore the relevant methods and protocols.

I will also supplement this review by proposing the use of soil microbial communities as bioindicators of wetland development and sustainability. I will describe the laboratory and field procedures necessary to evaluate the various biological and physical aspects of mitigated wetlands, thereby offering managers an effective monitoring

technique. My intention is to confirm that microorganism development and preservation are critical to wetland health and longevity.

My final objective is to document the relevant literature on environmental policy, and provide current scientific and policy review for researchers, managers and legislators. This thesis will synthesize the diverse and often contradictory theories, and suggest possible methodologies to bridge the science-policy gap.

Overall, I intend to supply researchers, managers, and government agencies with a source of publications that can assist in evaluating, managing and monitoring wetland mitigation projects.

Chapter 1: Wetlands - Definitions, Impacts and Legislation

Introduction

Wetlands are among the most productive ecosystems in the world, comparable to rain forests and coral reefs. A variety of species of microorganisms, plants, insects, amphibians, reptiles, birds, fish, and mammals comprise wetland ecosystems (Figure 1.1). Physical and chemical features such as climate, landscape shape (topology), geology, and the movement and abundance of water help to determine the flora and fauna that inhabit each wetland (U.S. Environmental Protection Agency [EPA], 2006). The functions of a wetland and the values of these functions to human society depend on a complex set of relationships between the wetland and the other ecosystems in the watershed.

Wetlands cover almost 30 percent of the State of Florida and account for just over 10 percent of the remaining wetland area in the lower 48 United States. Over the past 200 years, Florida has lost an estimated 10 million acres of wetland, about half of the total area thought to exist in the 1780's (Clark, 2004). Some of these remaining wetlands are well known, like the Florida Everglades, while others may be small and unassuming. All play a vital role in flood protection, water quality and wildlife habitat.

This first chapter will discuss wetlands, their delineation characteristics, classification schemes, and importance. In addition, Federal, State and non-regulatory wetland regulations for natural and mitigated wetlands will be provided, as well as case studies of successful and unsuccessful mitigation projects.

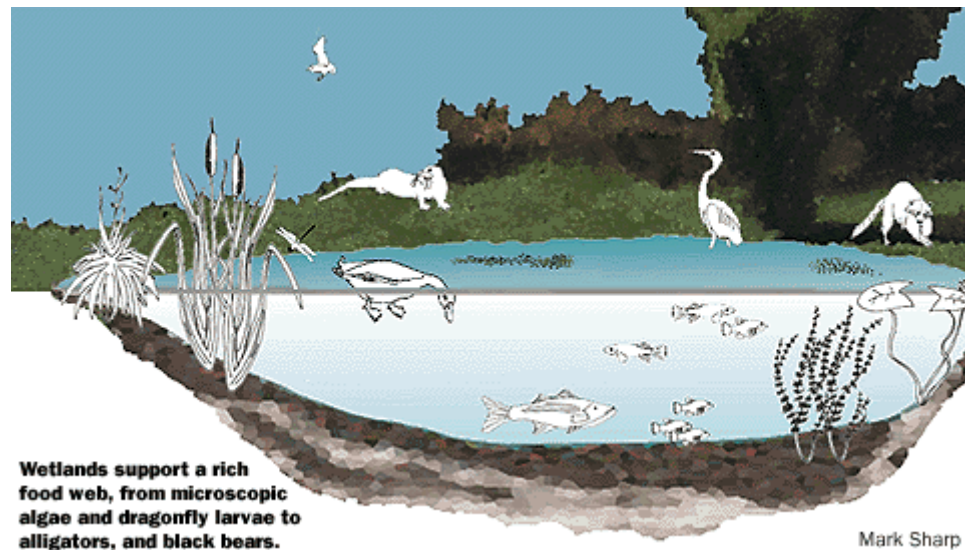


Figure 1.1: Wetland Biodiversity (U.S. Environmental Protection Agency [EPA], 2006).

Wetland Terminology

Today there are more than 50 definitions of the term “wetland.” Each definition is based on the spatial, temporal, political, economical, and social differences among states and federal agencies. In 1979, Cowardin, Carter, Golet and Laroe proposed the following definition, which was later adopted by the United States (U.S.) Fish and Wildlife Services (FWS):

Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For purposes of this classification, wetlands must have one or more of the following attributes: (1) at least periodically, the land supports predominately hydrophytes, (2) the substrate is undrained, hydric soil, and (3) the substrate is non-soil and is saturated with water or covered by shallow water at some time during the growing season of each year (Cowardin et al., 1979, p.3).

Today, the U.S. Environmental Protection Agency (EPA), U.S. Army Corps of Engineers (CE), and the U.S. Department of Agriculture, National Resources Conservation Service (NRCS) collectively refer to wetlands as:

Those areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas (Environmental Laboratory, 1987, p. 9).

States often incorporate federal language into their definitions. For example, the State of Florida defines wetlands in Section 373.019 (17) of the Florida Statutes, and Section 62-340.200 (19) of the Florida Administrative Code, as follows:

(Wetlands are)...areas that are inundated or saturated by surface water or groundwater at a frequency and a duration sufficient to support, and under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soils. Soils present in wetlands generally are classified as hydric or alluvial, or possess characteristics that are associated with reducing soil conditions. The prevalent vegetation in wetlands generally consists of facultative or obligate hydrophytic macrophytes that are typically adapted to areas having soil conditions described above. These species, due to morphological, physiological, or reproductive adaptations, have the ability to grow, reproduce, or persist in aquatic environments or anaerobic soil conditions. Florida wetlands generally include swamps, marshes, bayheads, bogs, cypress domes and strands, sloughs, wet prairies, riverine swamps and marshes, hydric seepage slopes, tidal marshes, mangrove swamps and other similar areas. Florida wetlands generally do not include longleaf or slash pine flatwoods with an understory dominated by saw palmetto.

All of these definitions express three primary delineation, or diagnostic, characteristics used to determine wetland presence and viability.

Characteristics of Wetlands

Although wetland types and locales vary, they all possess environmental characteristics that distinguish them from upland or other aquatic ecosystems. These characteristics are used as guidelines for the Army Corps of Engineers (CE) 1987 Wetland Delineation Manual, which is still used today. Wetland delineation involves establishing the boundary between wetlands and uplands (or non-wetland areas).

Wetlands are characterized by unique hydrologic, soil (substrate), and biotic conditions. The CE describes these characteristics in the following manner:

- **Wetland Hydrology** constitutes the areas inundated either permanently or periodically, or the soil is saturated to the surface at some time during the growing season of the prevalent vegetation.
- **Wetland Soils** are classified as hydric; possess characteristics that are associated with reducing soil conditions.
- **Wetland Vegetation** includes macrophytes that are typically adapted to areas having the hydrologic and soil conditions described above (Environmental Laboratory, 1987).

The Delineation Manual provides guidance for identifying jurisdictional wetlands. States have traditionally relied on vegetation for wetland identification, but recently have either adopted the Federal approach, or developed tiered methods of analysis consisting of first considering vegetation and, when necessary, considering the additional diagnostic characteristics of hydric soil properties and hydrologic indicators. The Wetland Delineation Manual has been updated through a series of guidance documents memoranda from the CE Wetland Research Program, and an electronic edition is available online at <http://el.erdc.usace.army.mil/wetlands/pdfs/wlman87.pdf>.

Positive indicators of hydrophytic vegetation, hydric soils, or wetland hydrology must be present during some portion of the growing season for an area to be considered a natural wetland. Exceptions to the delineation rules do exist, however, because wetland systems are often subject to alteration. Naturally occurring events can promote the creation or alteration of wetland systems. For example, beaver dams can impound water, thereby shifting the hydrology and vegetation of a wetland system. A wetland may be considered “atypical” when one or more of the field indicators have been obscured by some recent change. If the change took place after 1977 it may be determined to be a wetland and subject to the Clean Water Act.

Problem area wetlands are wetlands that are inherently difficult to identify because field indicators of one or more wetland parameters may be absent or misleading, at least at certain times of the year. For example, wetlands that have been purposely or

incidentally created by human activities may not possess the appropriate wetlands indicators compared to local, natural wetlands. The substrate or hydrology may be different due to the resources available to the wetland developers. In addition, during the dry season, highly seasonal wetlands often lack the hydrology and vegetative indicators typical of non-seasonal wetlands (Environmental Laboratory, 1987).

Of the three delineation characteristics, wetland hydrology is considered the most important variable for determining the development and maintenance of wetlands on a landscape level, because it is deemed the process that drives the other ecological elements of the system. Furthermore, understanding the hydrologic aspects of wetlands helps explain their landscape diversity (Winter, 1988, 1992). Bedford (1996) contends that replacement wetlands cannot be considered equivalent to natural wetlands unless their hydrologic regimes are comparable. The surface water components of wetlands, such as the hydroperiod (the period of time during which the wetlands is covered by water) and water depth gradients, have been the subject of extensive research (Menges & Waller, 1983; Lieffers, 1984).

The hydrologic regime of the ecosystem is determined by the flow, duration, amount, and frequency of water on a site (Interagency Workshop on Wetlands Restoration [IWWR], 2003). Brinson (1993) describes three basic wetlands flow types: (1) vertical fluctuations of the water table that result from evapotranspiration and subsequent replacement by precipitation or groundwater discharge into the wetland, (2) unidirectional flows that range from strong channel-contained currents to sluggish sheet flow across a floodplain, and (3) bidirectional, surface or near-surface flows resulting from tides or seiches.

Snodgrass, Komoroski, Bryan and Burger (2000) studied hydroperiod duration and evaluated models of lentic communities to determine the relationship between hydroperiod length and species sustainability. Their results indicated that short-hydroperiod wetlands support a unique group of species. Snodgrass et al. (2000) concluded that hydroperiod length should be a primary criterion for the development of wetland regulations, and advocated the conservation of a diversity of wetlands that represent the entire hydroperiod gradient (termed the “Landscape Approach”).

The hydrologic complexities of wetlands have prompted the establishment of field indicators, which are used to identify the hydrological characteristics of natural wetland systems. Examples of these indicators include:

- Standing or flowing water observed on the area during the growing season;
- Waterlogged soil during the growing season;
- Watermarks on trees or other erect objects. Such marks indicate that water periodically covers the area to the depth shown on the objects;
- Drift lines, which are small piles of debris oriented in the direction of water movement through an area. These lines often occur along contours and represent the approximate extent of flooding in an area;
- Debris lodged in trees or piled against other objects by water; and
- Thin layers of sediment deposited on leaves or other objects. Sometimes these become consolidated with small plant parts to form discernible crust on the soil surface (Environmental Laboratory, 1987).

The readily available water in wetlands supports soils (called hydric soils) that remain saturated for part or all of the year. The upper part of hydric soils (6-12 inches) becomes anaerobic as water stimulates the growth of microorganisms, which use up the oxygen in the spaces between soil particles (IWWR, 2003). Anaerobic soils have unique chemical attributes. For example, the decomposition of organic matter is slowed down as a result of the anaerobic nature of wetlands. As such, wetlands trap carbon as soil organic matter instead of releasing it into the atmosphere (as carbon dioxide) and are considered carbon sinks (Sims, 1990).

The thickness and permeability of wetland soils are essential to wetland formation (Winter, 1988). The low permeability of organic soils, as a result of their underlying silts and clays, generally restrict the vertical movement of water to or from the peat layer, yet allow local groundwater discharges and recharges to occur. Thus water can persist in wetland systems for longer periods of time (Bedford, 1996).

The NRCS has listed approximately 2,000 types of soils in the United States that may occur in wetlands. This list, as well as a list of the Field Indicators of Hydric Soils,

is available on their website (<http://soils.usda.gov>). Field indicators offer on-site means to confirm or reject the presence of hydric soil. Some of the indicators include:

- An odor of rotten eggs (sulfuric smell),
- Mottled soils, which are grayish subsoils with orange or reddish mottles, reflecting a fluctuating water table but prolonged saturation; and
- Predominantly sandy soil that has dark stains or dark streaks of organic material in the upper layer below the soil surface. These streaks are decomposed plant material attached to the soil particles. When soil from these streaks is rubbed between the fingers, a dark stain is left on the fingers (Environmental Laboratory, 1987).

With the permission of the Soil Survey staff, the Florida Department of Environmental Protection (FDEP) generated a document on their website that outlines the hydric soils in Florida, and offers soil criteria and field indicators (<http://www.dep.state.fl.us/mainpage/default.htm>). In addition, a detailed description of the hydric soil indicators developed for Florida is provided in Soil and Water Relationships of Florida's Ecological Communities .

Plants that can grow, compete and reproduce amidst the waterlogged, anaerobic conditions of wetlands are facultative or obligate hydrophytic macrophytes, or hydrophytes. Hydrophytes have developed morphological and physical adaptations and distinct colonization strategies that allow alternate metabolic pathways for successful oxygen exchange in otherwise toxic environments (Elliot, 2004). Wetland plants can be grouped based on their expected frequency of occurrence in the wetland system.

- Obligate wetland plants (OBL) must live and grow in wetlands and deep water habitats in order to survive, and occur in wetlands 99 percent of the time;
- Facultative wetland plant species (FACW) usually occur in wetlands 67 to 99 percent of the time, but are occasionally found in non-wetlands;
- Facultative plant species (FAC) are equally likely to occur in wetlands or non-wetlands (34 to 66 percent of the time);
- Facultative upland plants (FACU) exist predominantly in non-wetland habitats (occurring 1 to 33 percent of the time), but can survive in wetlands;

- Obligate upland plants (UPL) almost occur more than 99 percent of the time in uplands (U.S. Fish and Wildlife Service [FWS], 2006);

A positive (+) or negative (-) sign, when used with indicators, more specifically defines the frequency of occurrence in wetlands. The positive sign designates, “slightly more frequently found in wetlands” and the negative sign indicates, “slightly less frequently found in wetlands.” (FWS, 2005).

Of all the vascular plants that grow in the United States, only one-third can tolerate the unique conditions associated with most wetlands. Of those that do occur in wetlands, only 26 percent are obligate hydrophytes (FWS, 2005). Typical wetland vegetation includes emergent plants (those with leaves that grow through the water column, such as cattails, sedges, and rushes), submerged plants (pondweeds, eelgrass), floating-leaved plants (such as water lilies and duckweed), trees (cypress, red maple and swamp oak), shrubs (such as willows and bayberry), moss, and many other vegetation types. The most common wetland plants are cattails, bulrushes, cord grass, sphagnum moss, bald cypress, willows, mangroves, sedges, rushes, arrowheads and water plantains. Some field indicators of wetland plants are:

- Trees having shallow root systems and/or swollen trunks (e.g. bald cypress, tupelo gum); and
- Roots found growing from the plant stem or trunk above the soil surface (Environmental Laboratory, 1987).

Several studies have shown that the diversity of plant communities within and among wetland ecosystems is related to patterns of groundwater flow and groundwater chemistry, which result from the hydrogeologic setting of the wetland, the interplay and abundance of water and nutrients in the soil and factors such as the salinity, water depth and water level fluctuation, type of soil, temperature, local elevation, and landscape (Siegel, 1983; Siegel & Glaser, 1987; Glaser, Janssens & Seigel, 1990).

The interplay of wetland hydrology, hydric soil and hydrophytes allow for a complex ecosystem. Wetlands improve the quality of surface water by filtering, absorbing and settling out soil particles, organic matter and some nutrients such as phosphorus (Mitsch & Gosselink, 1993). These removed materials provide an excellent

environment for algae and bacteria, which actively degrade organic pollutants, such as pesticides, and decompose chemicals arriving from upstream natural and human sources (Mitsch & Gosselink, 1993). Microbial communities (on plant roots and in the soil) convert organic nitrogen into ammonium nitrogen and perform denitrification (the conversion of nitrates in the soil to less toxic atmospheric nitrogen). The dense wetland vegetation also traps pollutants that attach to the soil particles, and controls erosion by reducing wave and current energy, binding and stabilizing the soil, and helping the soil recover from flood damage (Maier, Pepper & Gerba, 2000). Figure 1.2 depicts the various components and interactions within wetland systems.

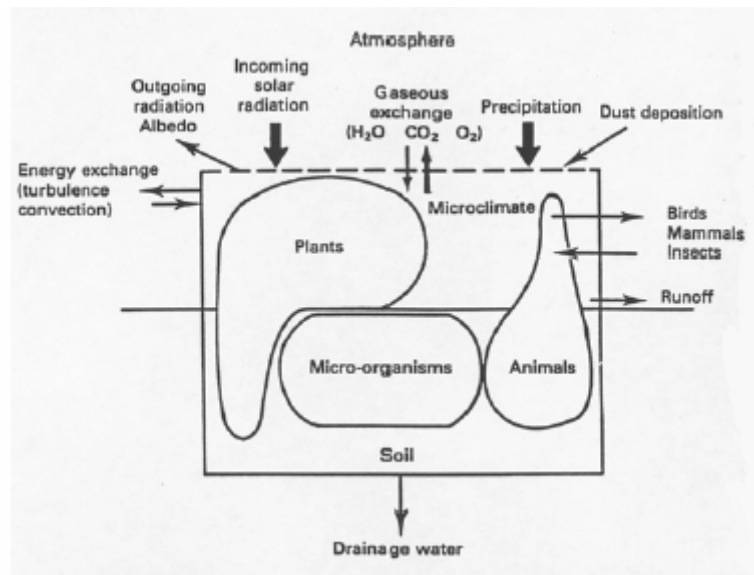


Figure 1.2: A Schematic of a Wetland System (Maser, 2006).

Wetlands serve as transitional areas between terrestrial and aquatic systems, and support a variety of micro- and macrofauna. More than one-third of the threatened and endangered species in the United States live primarily in wetlands, and nearly half use wetlands at some point in their lives (EPA, 1995). Numerous species of invertebrates, fish, and reptiles depend on wetland water cycles for reproduction and survival. Approximately 75 percent of all commercial marine fish species depend on estuaries, which in turn rely on nearby wetlands to maintain these productive ecosystems (IWWR, 2003). Nearly all amphibians and at least 50 percent of migratory birds use wetlands

regularly. The breadth of species that depend on wetlands for survival, reproduction and breeding habitat is given on the EPA website (www.epa.gov) and is available in the 1995 EPA resource entitled “America’s Wetlands: Our Vital Link Between Land and Water.”

In addition to serving as habitats for various wetland species and filtering surface water, wetlands have a multitude of other benefits, such as providing:

- Healthy fisheries;
- High Biological productivity;
- Biodiversity protection;
- Erosion control;
- Flood damage reduction and sediment control;
- Aesthetic and recreational opportunities; and
- Water quality improvement (FWS, 2005; EPA, 1995).

Wetland Classifications and Settings

Wetlands occur in a wide variety of settings due to differences in geological characteristics, such as surface relief and land surface slope, the thickness and permeability of the soils, and the composition, stratigraphy and hydraulic properties of the underlying geological materials (Bedford, 1996). Winter (1992) outlined eight physiographic settings in which wetlands develop: (1) terraces and scarps within coastal lowlands, (2) terraces within riverine valleys, (3) steep slopes (mountains) adjacent to narrow lowlands, (4) depressions in large extensive lowlands (playas), (5) morainal depressions, (6) dune fields, (7) sinkholes and (8) permafrost. As all physiographic settings do not occur in all climatic regimes, the number of *ideal* locales is limited. Further wetland occurrence is restricted by the intricacies of local surface and groundwater flows (hydrodynamics).

Cowardin et al. (1979) identified five major wetland systems – Marine, Estuarine, Riverine, Lacustrine and Palustrine. These systems are distinguished by a variety of hydrologic, geomorphic, chemical, and biological characteristics such as:

- Marine - open ocean and its associated shoreline

- Estuarine - estuaries where mixing of salt and fresh water occur
- Riverine - channels of rivers and streams
- Lacustrine - lakes, reservoirs and deep ponds
- Palustrine - mostly freshwater wetlands and shallow ponds plus inland saline wetlands (see Appendix A).

The FWS have developed a wetland classification system based on the determinations of Cowardin and his collaborators. The FWS identified 55 different classes of wetland and deepwater habitats used to map wetlands throughout the United States. Wetlands and deepwater habitats are first divided into five ecological systems (listed above), then grouped by subsystems and classes. Most of the nation's wetlands are Palustrine wetlands. At the class level, wetlands are separated by vegetation (where more than 30 percent of the area is vegetated) or by the predominant substrate for other wetlands. Vegetated classes include aquatic bed, emergent wetland, scrub-shrub wetland, forested wetland, and moss-lichen wetland. Non-vegetated classes include rock shore, unconsolidated shore (e.g. mudflat and sandy beach), streambed, rock bottom, and unconsolidated bottom. Wetlands may further be described by various modifiers including water regime (hydrology), water chemistry (pH and salinity/halinity), soil (organic or mineral), and other special modifiers (e.g. partly drained, diked/impounded, excavated and artificial) (FWS, 2005).

The National Research Council's 1995 report entitled "Wetlands: Characteristics and Boundaries" outlines a second classification scheme. This report lists several major classes of wetlands and typical associated vegetation:

- Freshwater Marsh - grasses, sedges herbs;
- Tidal Salt and Brackish - salt tolerant grasses, rushes;
- Prairie Potholes - grasses, sedges, herbs;
- Fens - sedges, grasses, shrubs;
- Bogs - sphagnum moss, shrubs trees;
- Swamp Bottomland - cypress, gum, red maple; and

- Mangrove Forest - black, red white mangroves (National Research Council [NRC], 1995).

Brinson (1993) defines wetland classification in terms of the Hydrogeomorphic Approach (HGM) to assessing wetland functions. The HGM Approach assesses wetlands based on three fundamental factors that influence how wetlands function: (1) hydrodynamics – the direction, flow and strength of water movement within the wetland, (2) geomorphic setting – the topographic location of the wetland within the surrounding landscape, (3) hydrology – precipitation, surface flow and groundwater discharge (i.e. that water source.).

The HGM Approach first classifies wetlands based on their differences in functioning, defines the functions that each class of wetlands performs, and uses reference wetlands to establish the range of functioning of the wetland. Regional assessment models are then developed based on the functional profile that describes the physical, biological and chemical characteristics of a regional wetland subclass (Brinson, 1993).

A supplementary classification of wetlands is based on the ecosystem's hydrogeologic setting, which is the position of the wetland in the landscape with respect to the flows of surface water and ground water. Several authors have summarized this classification method (Winter, 1992; Winter & Woo, 1990). For the complete national wetland classification standards, please visit: <http://wetlands.fws.gov/PubsReports/pubs>.

Wetlands types include tidal marshes, prairie potholes, seagrass beds, forested wetlands and seasonally ponded sites, such as vernal pools. Some of these wetland types, such as seasonal wetlands that are dry most of the year, may not always appear to be wetlands. Each system is unique and offers distinct structure, ecological function, aesthetic use and recreational benefits. For example, smaller wetlands near flood-prone residential areas may offer greater flood protection than a larger wetland more suitable for extensive wetland habitat (Bingham, Clark, Haygood & Leslie, 1990).

Legislation for Wetland Protection and Mitigation

Federal Wetland Regulations

Federal, State and Local wetland regulations were developed for the protection of wetlands through permitting requirements, which control discharges and hazardous waste. These regulations also control use and/or alteration of wetlands as they pertain to other resources (e.g., wildlife) that depend on proper functioning wetlands (Dennison & Berry, 1993). The advent of legislation created to protect the nation's waterways began in 1899 with the Rivers and Harbors Act prohibiting the creation of obstructions to the navigable capacity of any of the waters of the United States without specific approval of the Chief Engineer of the CE.

In 1972, Congress passed the Federal Water Pollution Control Act Amendments, also known as the Clean Water Act, to restore and maintain the chemical, physical, and biological integrity of the Nation's waters (Lewis, 2001). The Act defined "navigable waters" as "waters of the United States." In 1977, the CE redefined navigable waters to include "isolated wetlands and lakes, intermittent streams, prairie potholes, and other waters that are not part of a tributary system to interstate waters or to navigable waters of the United States, the degradation or destruction of which could affect interstate commerce." This definition remains in effect today (Lewis, 2001).

Provisions in the Food Security Act (FSA) of 1985 lent support to the wetland preservation fight. Section 3821, Subchapter III (Wetland Conservation) of Chapter 58 (Erodible Land and Wetland Conservation and Reserve Program) of the Act states that individuals draining wetlands after December 1985 would not be eligible for many of the farm and loan programs of the U.S. Department of Agriculture (USDA). This is the called the Swampbuster provision of the 1985 Act. Swampbuster discourages the conversion of wetlands to cropland use. Section 3822 of the Act deals with the delineation of wetlands, resulting wetland delineation maps, and other issues such as the appeal process if a land owner does not agree that features delineated as wetlands on their farms are indeed wetlands (Dennison & Berry, 1993).

In 1988, as part of his election campaign, then Vice-President George Bush pushed for a “no net loss” of our nation’s wetlands as a national goal. He endorsed the recommendations of the National Wetland Policy Forum, an intergovernmental, public and private coalition convened at the recommendation of the EPA (Dennison & Berry, 1993). After the 1988 election, the Bush Administration upheld their decrees and the FWS released the National Wetlands Priority Conservation Plan in 1989, which implemented the Federal Emergency Wetlands Resources Act of 1986 (FWS, 1989). The Bush campaign recognized and acknowledged wetland losses across the United States and developed procedures for the acquisition of wetlands by Federal, State and local governments.

Unfortunately, over time, all good intentions began to wane as legislation was being passed that contradicted previous efforts. The most noted examples were the Memorandums of Agreement (MOA) in 1989 and 1990 between the CE and EPA regarding wetland mitigation requirements. This agreement authorized a less than one-to-one replacement of wetland acreage for degraded or lost wetlands. Many environmentalists believed that this was a major step in the overall weakening of wetland protection (Dennison & Berry, 1993). “No net loss” remained federal policy under the Clinton Administration, but today seems to have been relegated to nothing more than a catch phrase.

Of all of the past wetland protection legislation, the most controversial is Section 404 of the Clean Water Act (CWA). State, federal and private entries are required to obtain a permit from the CE before depositing dredged or fill materials into the waters of the United States, which includes wetlands. Under Section 404, wetlands may legally be destroyed, but some form of reclamation or mitigation must compensate their loss. Several sections of the CWA pertain to regulating impacts to wetlands:

- Section 101 specifies the objectives of this Act which are implemented largely through Title III (Standards and Enforcement).
- Section 301 (Prohibitions) specifies that the discharge of dredged or fill material into waters of the United States is subject to permitting specified under Title IV

(Permits and Licenses) of this Act and specifically under Section 404 (Discharges of Dredge or fill Material) of the Act.

- Section 401 (Certification) specifies additional requirements for permit review particularly at the state level.
- Section 309(a) allows the EPA to issue administrative compliance orders requiring a violator to stop any ongoing illegal discharge activity and, where appropriate, to remove the illegal discharge and otherwise restore the site.
- Section 309(g) stipulates that the EPA and the CE can assess administrative civil penalties of up to, but not exceeding, \$125,000 per action.
- Sections 309(b) and (d) and 404(s) give the EPA and CE the authority to take civil actions, seek restoration and other types of injunctive relief, as well as civil penalties. The agencies also have authority under Section 309(c) to bring criminal judicial enforcement actions for knowingly or negligently violating Section 404 (Lewis, 2001; Dennison & Berry, 1993).

Administration of the CWA falls to several agencies each with specific tasks: the CE issues permits; the EPA reviews permit applications, prepares guidelines for disposal sites, denying or restricting the use of any defined area as a disposal site, general enforcement; the FWS manages fish and wildlife game species and protects the threatened and endangered species within wetlands; and the U.S. National Marine Fisheries Service (NMFS) provide consultation and assistance under Fish and Wildlife Coordination Act (Dennison & Berry, 1993).

In addition to jointly implementing the CWA Section 404 program, the EPA and CE share Section 404 enforcement authority. Section 404 violations stem from failure to comply with the terms or conditions of a Section 404 permit, or discharging dredged or fill material to waters of the United States without a permit. Although EPA's Section 404 enforcement program relies on voluntary compliance, enforcement tools are provided under Sections 309 and 404 of the CWA (Dennison & Berry, 1993).

In 1989, the EPA and CE entered into a MOA on enforcement to ensure efficient and effective implementation of their shared authority. Under the MOA, the CE has the lead on Corp-issued permit violation cases. For unpermitted discharges, the EPA and the

CE determine the appropriate lead agency, based on criteria in the MOA. Decisions regarding enforcement consider many factors when determining the need and type of enforcement actions, such as the amount of fill, the size of the water body (acres of wetlands filled and the environmental significance), the discharger's previous experience with Section 404 requirements, and the discharger's compliance history (EPA, 1990).

Though the provisions of Section 404 are the most discussed, no single piece of legislation entirely expresses the federal goals for wetland protection and management. In reality, a series of goals that relate directly or indirectly to wetland management and protection are embedded in several pieces of federal legislation.

Executive Order 11990 (Protecting of Wetlands) sets goals for federal agency action, and Executive Order 11988 (Flood Plain Management) requires federal agencies to avoid direct or indirect support for floodplain development wherever there is a practical alternative (Dennison & Berry, 1993). In 1973, the EPA adopted a "Statement of Policy on Protection of the Nation's Wetlands" which lists the values of wetlands and describes the Agency's decision-making processes. The EPA has also developed an implementation plan through its Wetlands Strategic Planning Initiative, which lists five goals for wetland management. In addition, the Endangered Species Act authorizes the FWS to prepare and implement a Habitat Conservation plan in certain situations to protect endangered species (Dennison & Berry, 1993).

Wetland language is also found in directives such as the Emergency Wetlands Resources Act of 1986, which states that existing federal, state, and private cooperation should be strengthened in order to minimize further losses and to assure their management for future generations (Dennison & Berry, 1993). In 1987, the National Wetlands Policy Forum was convened at the request of the EPA to address major policy concerns on protection and management of wetland resources. In November 1988 the Forum released "Protecting America's Wetlands: An Action Agenda" which contains over 100 recommendations for improving wetlands conservation. Recently, the EPA and the CE announced the release of a comprehensive, interagency National Wetlands Mitigation Action Plan, as well as an improved Wetlands Mitigation Regulatory

Guidance Letter, to further the achievement of the goal of “no net loss” of wetlands (Dennison & Berry, 1993).

Additional federal environmental laws that may impose additional permitting and regulatory compliance requirements on activities undertaken in wetland areas include:

- The Coastal Zone Management Act;
- The Endangered Species Act;
- The National Historic Preservation Act;
- The Preservation of Historical and Archaeological Data Act;
- The Marine Mammal Protection Act;
- The National Environmental Policy Act;
- The Clean Air Act;
- The Coastal Barriers Resources Act; and
- The Marine Sanctuaries Act.

State and Local Legislation

The “State Wetland Protection Programs”, prepared in 1986 for the EPA, identified 14 states with specific inland wetland laws (Dennison & Berry, 1993). Local governments may have ordinances that specifically address wetland management, particularly in regions where states delegate some wetlands regulatory responsibilities to local governments. Goals for wetland management may also be found in planning documents of local governments or regional planning bodies.

Some states have consistency requirements that apply to inland as well as coastal areas. Florida has a number of innovative programs for coordinating state, regional and local planning, and assigning authority to the most appropriate level of government. For example, Resource Planning and Management Committees (RPMC), comprised of interested parties and land managers, identify resource issues and develop management plans that are later incorporated into local comprehensive plans and land use regulations. The RPMC process promotes advanced planning and provides a vehicle for introducing state goals and policies to the local level (Dennison & Berry, 1993).

The Florida Department of Environmental Protection (FDEP) and the five water management districts (South Florida, Suwannee River, St. John's River, Southwest Florida and Northwest Florida) share the responsibility for protecting wetlands in Florida. Wetlands are regulated by Chapter 373 of the Florida Statutes; Part IV of the Statutes is the State's surface water regulatory program, which is jointly implemented by the State and the water management districts using rules adopted by the FDEP for each specific water management district (Morgan Lewis, 2001).

Impacts to wetlands (including dredging or filling) require an Environmental Resource Permit (ERP) from the FDEP or one of the water management districts. The ERP program and Sovereign Submerged Lands Program regulate activities throughout Florida, except for the Florida Panhandle. The Wetlands Resource Permit program covers the Panhandle area and regulates any alterations of water, and wetlands that are connected (either naturally or artificially) to "named water" such as Gulf of Mexico, bays, sounds, estuaries, lakes and streams (FDEP, 2006a).

Active mining throughout Florida has sparked several legislative changes. Florida has mandated that all land mined for phosphate after July 1, 1975 must be reclaimed (defined in the Wetland Mitigation Terminology section) (Chapter 211, Florida Statutes; Chapter 378, Florida Statutes). Prior to 1975, more than 149,000 acres of land were mined and disturbed by phosphate mining operations. Taxes collected on each ton of phosphate rock mined today are used to help pay for the reclamation areas mined prior to 1975, most of which are now in private ownership. These taxes also help pay for preserving environmentally sensitive lands through the Florida's Conservation and Recreation Lands (CARL) Program.

Currently, about 15 percent of the Bone Valley district of Polk and eastern Hillsborough and Manatee counties are overlain by wetlands. Native wetlands include both the herbaceous ecosystems (wet prairies and marshes), and forested or shrub-dominated ecosystems (cypress domes, cypress swamps, mixed cypress-hardwood swamps, bay forests and swap thickets). The phosphate industry has been required by law to reclaim wetlands since the adoption of the FDEP rules (Appendix B), which state

that the wetlands affected by mining operations are to be restored to at least pre-mining surface areas, acre-for-acre and type-for-type (FDEP, 2006b).

The FDEP Bureau of Mine Reclamation (BMR) was developed in the late 1980s to preserve, manage and protect state lands another than parks, recreational and wilderness areas. The Integrated Habitat Network (IHN) and Coordinated Developed Area (CDA) plans, prepared by the BMR, provide guidelines for t reclamation and permitting throughout the central Florida phosphate mining districts (Florida Department of Environmental Protection Bureau of Mine Reclamation [FDEP-BMR], 2002a). Before any mining proceeds, the land reclamation process begins with the submittal of a reclamation plan to the BMR and other local, state and federal agencies. The BMR reviews the plans for consistency with adjacent mines and land uses, and with its regional conceptual plan. For a reclamation plan to be approved, it must provide detailed information on the identified land (including vegetative cover, animal species, topography, watersheds, land use and potential archaeology sites) both before the mining process and after reclamation (FDEP-BMR, 2002a).

The reclamation projects are actively managed by the phosphate mining companies, but sites are also field inspected quarterly by FDEP staff to ensure that permit conditions are achieved. Evaluations are considered on a case-by-case basis, according to the conditions mentioned in the permit, because no standardized, quantitative assessment of phosphate mine reclamation exists. Once reclamation had been satisfactorily completed in accordance with permit requirements, the mining company can be “released” from further obligation to perform mitigation, and the BMR no longer has jurisdiction over the area (FDEP, 2006b).

According to the December 2002 BMR “Rate of Reclamation Report,” 175,752 acres of land in Florida land have been mined since the mandatory reclamation law passed in July 1975. Of those acres, 53,821 acres have been reclaimed and released, and another 58,138 acres have been reclaimed through vegetation for industrial use, making a total of 111,959 reclaimed sites. This amount is 64 percent of what has been mined from 1975-2004 (FDEP-BMR, 2002b). These statistics sound impressive, but we must recognize that many factors are necessary for successful reclamation.

Reclamation is difficult to define because it depends on the reclamation goals. For example, the Strategic Regional Policy Plan for South Florida specifies several measures of success including: (1) improving the quality and quantity of the wetlands in the region, (2) increasing the percent of the restored/enhanced wetlands, and (3) reducing the loss of wetlands (Southwest Florida Regional Planning Council, 2004). Success also depends on improvements being made to the existing reclamation process and policies that emphasize the importance of ecosystem functionality. The numerous challenges in wetlands restoration (e.g., creating and maintaining the appropriate hydrologic regimes, ensuring water quality, controlling invasive or opportunistic plant species, soil development, and native vegetation establishment) seem to limit our advancements. Additional research is needed to further address the ecological query of wetland sustainability. As a result, the Florida Institute of Phosphate Research (FIPR) was developed to conduct and fund research aimed at addressing the environmental, economic and health impacts of Florida's phosphate mining and fertilizer industry.

FIPR is financed with funds from the state severance tax on phosphate rock (Florida Institute of Phosphate Research [FIPR], 2004a). Studies include those designed to develop better techniques for reclaiming land, evaluate the chemical processing of phosphate rock, the beneficiation process, public health issues, and emergent mining and processing technologies (FIPR, 2004a). Technical advisory committees, made up of stakeholders from government, industry, environmental groups and academics, review projects.

Recently, research has focused on the potential of clay settling areas as the setting for wetlands creation. In 1984, FIPR funded a 5-year restoration research project that centered on integrated landscape restoration (Brown & Tighe, 1991). The project was designed to provide the phosphate industry with information about the structure and organization of watersheds (sizes, slopes, upland/wetland ratios, spatial organization, etc.) and ecosystems (dominant species in all strata, soils, topography, and hydrologic regimes). The project team also studied mined lands to characterize abiotic conditions in order to cross reference mined lands with appropriate ecosystem types (Brown & Tighe, 1991). The authors considered the work a restoration cookbook, as they outlined the

design principles for watersheds, the characteristics and design guidelines for floodplains, and the structural characteristics of natural wetlands.

As the phosphate ore of the northern portions of the southern district is mined out, the industry is moving southward. Local opposition arises from the persistence of noise and air pollution, the alterations of the regional hydrology, and the clay settling basins, which are estimated to occupy as much as 40 to 60 percent of the landscape after mining (Brown, 2005). Current research funded by FIPR is addressing these topics, as well as modeling landscape scenarios of future impacts.

Non-regulatory Approaches to Wetland Regulation

Federal, state and local governments have also adopted several non-regulatory programs for wetland protection. The principal owners and managers of wetlands at the federal level are the U.S. Department of the Interior, including the FWS, the National Park Service and the Bureau of Land Management; the U.S. Department of Commerce, specifically the National Oceanic and Atmospheric Administration; and the U.S. Department of Agriculture, mainly the U.S. Forest Service. Other Federal agencies, such as the Bureau of Reclamation in the Department of the Interior, the CE and the Department of Defense, also manage wetlands (Sunding & Zilberman, 2003).

Some non-regulatory programs involve the government acquiring full or partial title of the land, while others provide financial inducement, tax credits or government payments to land owners for protecting and managing the wetlands in a desired manner. Many of the programs, however, have broader goals and the protection of the wetland is only part of that broader purpose (Goldfield & Clark, 1990). Overall, most programs encourage protecting existing wetlands, and promoting restoration and enhancement efforts.

Land acquisition has proven to be an effective program for wetland preservation. The Emergency Wetlands Resources Act of 1986 (EWRA) contains a provision for establishing a National Wetlands Priority Conservation Plan, which guides acquisition efforts on all levels. The FWS has designed a 10-year acquisition plan which aims to protect more than 2.6 million acres of waterfowl habitat through the purchase of land or

conservation easements (FWS, 1988). For land acquisition, funds can be generated through appropriations from general revenues (usually based on property tax receipts), special purpose bonds, and intergovernmental transfers (Sunding & Zilberman, 2003). The EWRA also authorizes entrance fees at certain national wildlife refuges with 70 percent of the proceeds going to the Migratory Bird Conservation Fund. Other funding sources include bonds, general revenue (from state taxes), stamp taxes, severance taxes, and tourist impact taxes, with additional funds coming from other taxes and revenues, as from state forests and hunting permits (Sunding & Zilberman, 2003).

The majority of the nation's wetlands are privately owned, thus private landowners are encouraged to manage/protect these resources through economic incentives such as subsidies and tax breaks.

Wetland Mitigation

Mitigation Terminology

The terms *mitigation* and *reclamation* are often interchanged. Wetland mitigation refers to the restoration, creation, or enhancement of wetlands as compensation for permitted wetland losses under Section 404 of the Clean Water Act (Lewis, 1989). The term mitigation is synonymous with "compensatory mitigation" because this process reduces environmental damage by avoiding, minimizing, and compensating for activities that damage protected resources. Mitigated wetlands can be reclaimed, restored, enhanced or created (constructed). Reclamation can be used in a number of ways, including describing: 1) the conversion of mined or other disturbed lands into economically productive properties, such as grazing lands or orchards; 2) the filling in of wetlands or shallow coastal waters to create land, usually for housing or urban infrastructure, but also for agriculture in some parts of the world; and 3) the conversion of disturbed lands to natural or semi-natural habitat (Streever & Peck, 2002). Both terms are used throughout this document.

The National Research Council, in its 1992 report "Restoration of Aquatic Ecosystems", defined restoration as the return of an ecosystem to a close approximation of its condition prior to disturbance, and the reestablishment of pre-disturbance aquatic

functions and associated physical, chemical and biological characteristics. Restoration encompasses a plethora of terms applied to projects undertaken to replace or repair the ecosystem function. This process can be achieved by rehabilitation or reestablishment processes. Rehabilitation is used to describe aesthetic improvements made to an existing, natural resource. Reestablishment is the restoration of a former wetland, which now may be degraded (NRC, 1992).

Site restoration requires the recontouring of spoils, the replacement of native habitat and plant communities, and the creation of a landscape that includes areas of low topographic relief where surface and groundwater can develop appropriate hydrologic regimes (Brown, 2005). Two techniques have been widely used to introduce wetland vegetation to created wetland areas: mulching and hand-planting. In many projects, organic mulch from wetlands permitted for mining is hauled to the restoration site complete with the roots, sprouts and seeds of wetland plant species intact. The mulch is then spread over the subsoil to form an organic layer that may vary from a few centimeters to tens of centimeters in thickness. As the water table returns to pre-mining levels, germination of the seed material in the mulch begins the revegetation process, and later, trees and other species are planted to complete the suite of plant species required by the restoration permit (IWWR, 2003).

Another mitigation option is enhancement. This technique involves increasing one or more of the functions performed by an existing wetland beyond what currently or previously existed in the wetland (Lewis, 1989). Often there is an accompanying decrease in other functions. For example, adding more water to a wetland may create better habitat for fish, but it will decrease the ability of the wetland to hold flood waters. This trade-off is particularly true for enhancement in relatively undisturbed wetlands (Lewis, 1989).

Mitigation through preservation involves the protection of wetlands through appropriate legal and physical mechanisms, such as conservation easements and title transfers. Preservation can include protection of upland areas adjacent to the wetland, or enhancement of the existing system (Lewis, 1989).

Some mitigation requires the creation (or construction) of a new wetland system that is designed to replace the existing acreage, structure or function of a pre-existing wetland. Constructed wetlands are created by excavating upland soils to elevations that can support wetland species and the appropriate wetland hydrology. Hammer (1989) described constructed wetlands as man-made complexes of saturated substrates, emergent and submergent vegetation, animal life and water that simulate natural wetlands for human use and benefits. Architecturally, constructed wetlands are a series of rectangular plots filled with soil or gravel and lined to prevent waste from leaching into the groundwater (Figure 1.3).

Constructed wetlands have different basic designs, and thus have different flow characteristics. Kadlec (1987) described the different flows as: (1) horizontal flow systems with the wastewater level above the soil surface, (2) horizontal subsurface flow systems with the wastewater level below the soil surface, (3) and vertical flow systems with upstream or downstream characteristics, and continuous or intermittent loading.

The soil is the main supporting material for plant growth and microbial films, and promotes the physiochemical and biological process that occur between the plants, microorganisms, and pollutants of the wetland ecosystem. The vegetation grown in these plots offer a root mass for filtration, and provide oxygen and carbon for water treatment. The roots also offer attachment sites for microorganisms, which consume the available oxygen in the process of breaking down pollutants (Stottmeister et al., 2003).

Constructed wetlands can serve as wastewater treatment plants. The operation and maintenance costs are lower than conventional treatment plants, and less energy and supplies are needed. These facilities can survive with periodic on-site labor, rather than the continuous, full-time attention required by conventional plants (EPA, 2004). In addition, constructed wetlands can also serve as wildlife sites, designed to attract various animals and provide needed wetland habitat.

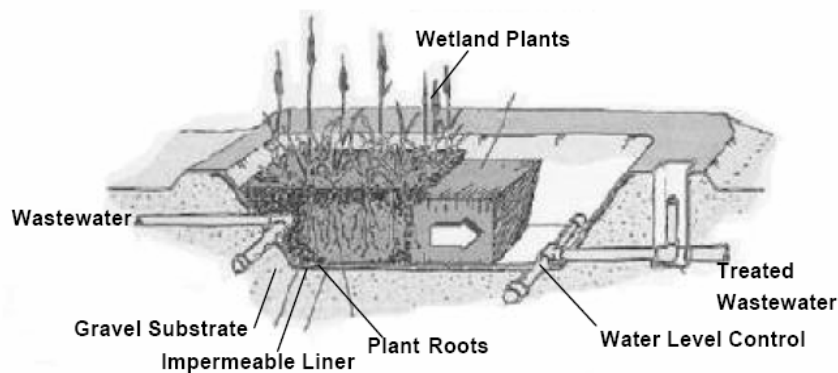


Figure 1.3: Schematic Diagram of a Constructed Wetland (EPA, 2004)

Constructed wetlands can be used to treat domestic, agricultural and mine drainage wastewater, as well as leachate, contaminated groundwater and industrial effluents (EPA, 1988). With the increased use of constructed wetlands, government agencies are devising appropriate regulations to protect public health and safety without unduly burdening constructed wetland designers and operators (EPA, 2000). A constructed wetland bibliography, compiled by the United States Department of Agriculture's Natural Resources and Conservation Service (formerly the Soil Conservation Service) and the Water Quality Information Center at the National Agricultural Library, consists of more than 600 citations outlining the use, maintenance and legislation of constructed wetlands. One hundred and sixty one of these citations have abstracts. A second edition, *Constructed Wetlands and Water Quality Improvement (II)*, created in June 2000, is now available. Both of these files are accessible online at <http://www.nal.usda.gov/wqic/Constructedwetlandsall>.

Case Studies of Successful Wetland Mitigation Projects

Land managers often focus on ways to identify functional wetlands in the field. Some of the documented indicators involve plants, soils and wildlife species. Though much has been written about wetland indicators, our ability to successfully mitigate these resources remains unacceptable. Glubiak, Nowka and Mitsch (1986) predicted that while protection of wetlands has been offered through the Section 404 permitting process, this

legislation is not able to prevent wetlands losses as effectively as it should. However, mitigation success stories do exist. The State of Florida and Federal government are conducting large-scale ecosystem restorations, such as the South Florida Everglades Ecosystem Restoration. This restoration project was developed to address the excess nutrient loading in the Florida Everglades and re-establish the natural hydrology of the area (Chimney & Goforth, 2001).

The Everglades dominate the landscape of south Florida and are considered an extremely important natural ecosystem. As a result of agricultural and urban development, eutrophication resulting from stormwater runoff, changes in hydrology and the invasion of exotic species, the biotic integrity of the Everglades is now threatened. To protect this valuable resource, approximately 17,000 hectares of treatment wetlands, referred to as Stormwater Treatment Areas, are being built to treat surface runoff before it is discharged into the Everglades (Chimney & Goforth, 2001).

Overall, only a very limited number of wetland community types have been created in reclaimed phosphate mining areas. Of the 30 classified types of wetlands occurring in Florida, 11 types were observed in the pre-mining landscape within the Florida phosphate mining districts. Of the numerous plant communities present in wetlands pre-mining, only about 6 types have been generally recreated in mitigation projects (Erwin, Doherty, Brown & Best, 1997).

Increasing uncertainty exists about the appropriate time frames and best criteria for evaluating wetland reclamation. Many techniques can be used to monitor the progress of mitigated wetlands; observing vegetation growth is considered the easiest and most common method (Wentworth, Johnson & Kologiski, 1988; Jarman, Dobertein, Windmiller & Lelito, 1991; Atkinson, Perry, Smith & Carins, 1993). Vegetative cover may be an easy measure of success but is often a poor indicator of function. Magurie (1985) used area, vegetative cover and implementation of permit conditions to estimate mitigation success in Virginia and found that only 50 percent of the 23 mitigated wetlands were successful.

An additional study by Reimold and Cobler (1985) conducted for the EPA gave similar results. Odum (1989) investigated the “self organization” capacity of nature to

both recruit species on its own and to make choices from those species introduced by humans. The author introduced as many species as possible, under the assumption that the system would assist in its own design by choosing those most appropriate. This approach emphasized the importance of natural colonization of species in ecosystem. Reinartz and Warne (1993) compared 11 created wetlands in southeastern Wisconsin that were naturally colonized with 5 wetlands in the same region where 22 species were introduced by seeding. Results indicated that success was achieved primarily through multiple seeding, multiple transplanting and establishment of hydrologically open systems, which allowed nature to participate in the wetland design. A large numbers of mitigation projects were undertaken in the 1970s and 1980s (reviewed and summarized in Kentula, Brooks, Gwin, Holland and Sherman, 1992) and analysis of these projects revealed that establishment of wetland vegetation was primarily used as a measure of wetland success.

As it is so difficult to monitor and manage every aspect of a wetland ecosystem, the Indicator Species Concept was proposed whereby a single species would be monitored and/or protected. Debate over this initiative arose because there is no consensus on what the indicator should indicate, and which species provide the best indicators (Simberloff, 1998). Other monitoring techniques include managing Umbrella Species (a species that needs such large tracts of habitat that saving it will automatically save many other species) and Flagship Species (charismatic large vertebrate used to anchor conservation campaigns since they can arouse public interest and sympathy). Both of these concepts have been used for monitoring, but remain controversial.

A review of the methods used to measure success in previous restoration efforts by McCoy and Mushinsky (2002) specified the value of multi-metric methods for measuring the success of wildlife community restoration when the restoration covers a broad geographic area. Using multi-metrics, species were ranked according to the magnitude of the difference between distributions at restored and reference sties. McCoy and Mushinsky applied this species ranking method to data from 30 restored sites (phosphate-mined land) and 30 reference sites in central Florida, and results indicated that ranking enables well-founded decision-making about which species can be excluded,

if necessary, without seriously compromising the restoration goal. This technique also overcame the problem that a single species may inordinately influence the result (McCoy & Mushinsky, 2002).

The concept of ecosystem management also addresses the problems of single-species management. According to this viewpoint, if a healthy ecosystem is maintained, the component species will all thrive. Some conservationists see ecosystem management as a “Trojan Horse” that would allow continued environmental destruction in the name of modern resource management (Simberloff, 1998).

The amount of time required for successful mitigation is also a consideration. Petranka and colleagues (2003) used amphibians to assess the ecosystem function of compensatory wetlands created at a mitigation site in western North Carolina. Petranka, Murray and Kennedy (2003) studied pond colonization and long-term community dynamics in ponds to examine whether landscape variables influenced the initial colonization of 22 constructed ponds (ten constructed and ten reference ponds) over seven breeding seasons. Results indicated that the constructed ponds supported significantly more species than reference ponds. In addition, the authors concluded that post-restoration monitoring for 2-3 years may be sufficient to characterize species and communities that will utilize ponds for the first decade or so after pond creation.

While soil properties have not been traditionally used to measure wetland restoration success, Whited et al. (1999) recently suggested that, in addition to return of hydrology, plant communities, and wildlife use, some soil properties could be used as criteria for assessing wetland restoration. Nair, Graetz, Reddy and Olila followed in 2001 with a study designed to compare created wetlands to native wetlands in phosphate-mined areas from central and north Florida using soil development analyses. The criteria selected for evaluation of soil samples from the 184 sites included soil compaction, bulk density, organic matter (carbon) and nitrogen content, the carbon–nitrogen ratio, and available and total nutrient contents. Based on the above-mentioned parameters, results indicated that the reclaimed wetlands were slowly developing into “typical” wetlands.

Nair et al. (2001) mentioned that the rate of development could possibly be increased by minimizing soil compaction, incorporating organic matter, or by

fertilization. Some of the substrates used in constructing a wetland in phosphate-mined areas include overburden, sand tailings, and/or clay, which provide both mechanical support and growth media for plants (Nair et al., 2001). It is also important to stress that the characteristics of soils in created wetlands can influence the levels of available resources, which in turn will affect the composition and diversity of above-ground vegetation and soil microflora and fauna (Chambers, Brown & Williams, 1994).

The fact of the matter is, it is time to do something concrete, monumental and decisive. If when a wetland is reclaimed certain species survive while others do not, we still need to consider the successful facets of the endeavor. We have to look at the value of mitigating wetlands and not get distracted by the faltering aspects which propagate the need for new, unproved innovations.

Case Studies of Unsuccessful Wetland Mitigation Projects

The reasons for reclamation shortcomings are numerous. Primarily, wetlands are hard to define. The physical and biological characteristics of wetlands are dynamic – plant boundaries can change in a season, water levels in hours. This lends to inconsistencies in terminology and interpretations. For example, reclamation success can have different connotations. Compliance success is determined by evaluating permitting requirements, whereas functional success is determined by assessing the restoration of ecological function to an altered ecosystem (Kentula, 2000). Historically, success was defined by individual projects, but ecologists are now considering the success of restoration on a landscape scale. Landscape success is a measure of how restoration (or management in general) contributes to the ecological integrity of the region or landscape and the maintenance of biodiversity (Kentula, 2000).

One of the greatest challenges is wetland creation, especially if it involves putting a wetland where one did not exist before. The primary quandaries in creation projects are bringing water to a site where it does not naturally occur, and establishing vegetations on soils that are not hydric. While creation is possible, it typically requires significantly more planning and effort than other restoration projects, and the outcome is difficult to predict. Developers often attempt to convert uplands to wetlands, but their efforts result

in ecosystems that are distinct from natural wetlands in the area and provide limited wetland functions. Creating wetlands from open water is less difficult with respect to establishing a water source, but it often requires placing dirt or other fill into existing aquatic habitats, which means destroying one kind of aquatic habitat to create another. While this trade-off sometimes can be justified ecologically, the engineering and regulatory challenges of these projects are so complicated that professional expertise and oversight are almost always required.

The outcome of a creation and enhancement project is often difficult to predict because these projects essentially try to produce a new ecosystem. With restoration projects, results are more predictable, although there may still be uncertainty depending on the type of wetland, extent of degradation, and many other factors. Under certain circumstances, creation or enhancement may be the best option but for the most part, restoration is more likely to have a positive outcome in terms of improving wetland resources.

Another mitigation dilemma is the functional equivalency of mitigated wetlands to natural reference wetlands. Mitigation changes the spatial distribution of wetland ecosystems, therefore Bedford (1996) advocates taking a landscape approach to defining hydrologic equivalence, so that potential cumulative effects of wetland diversity can be identified. Mitigation alters natural configuration and spatial distribution of wetland systems not only in immediate areas, but also over larger, geographic scales. Decisions regarding which sites to permit are often made in isolation, without consideration of other projects or the potential changes to the wetland system.

The majority of the nation's wetlands are privately owned, and landowners may manage their property in ways contradictory to wetland preservation. Thus mitigation legislation should continue to encourage private landowners to manage and protect their valuable wetland resource.

Additional problems stem from the ideas that wetlands are "lost." In reality, the land itself remains although there has been a change in the complex nature of the functions and benefits the original wetland provided. Also, in academic arenas, knowledge and expertise about the resources are scattered and conflicting. There are

differences in the reporting of the rate at which wetlands currently are being altered, the causes of these losses, ways to measure wetland function, and the types of existing wetlands.

Statistics of unsuccessful wetland mitigating focus on the acreage loss rather than the function omitted as a result of the loss. A measure of the damaged ecological functions would allow for a better insight into the wetland's importance in a particular area. Some wetland functions can be mimicked with engineered structure, but engineered methods typically do not provide the maximum ecological benefit. For example, instead of re-establishing native vegetation on wetland edges to control erosion, a cement wall could be used to armor the bank. A cement wall could limit the erosion for a time, but it does not provide the other ecosystem benefits of wetlands, such as filtering pollutants and providing fish habitat (IWWR, 2003).

Developing guidelines for wetland restoration, creation, and enhancement is difficult because the goals of people undertaking wetland projects vary widely and these goals influence the kind of activities best suited to a particular site (IWWR, 2003). Erwin (1991) found that of the 40 mitigation projects in South Florida involving wetland creation and restoration, only about half of the required 430 hectares of wetland had been constructed, and that 24 of the 40 projects (60 percent) were judged to be incomplete or failures. Two of the contributing factors were improper water levels and incorrect hydroperiod.

Some may consider the presence of non-native species in restored landscapes as unauthentic restoration. However, because the soils of mined lands have been altered and in most cases do not resemble those characteristics of native ecological communities, the likelihood of developing ecosystems with the same suite of species found in the native communities is diminished. Additionally because today there are more introduced species in the Florida landscape than in the past, it is probable that restored phosphate mined lands will contain many non-natives. For these reasons, if restored lands are not highly maintained, the presence of non-natives may be impossible to control (Brown, 2005). Petranka et al., (2003) conducted an 8-yr study to examine the demographic responses of the wood frog (*Rana sylvatica*) and spotted salamander (*Ambystoma*

maculatum) to wetland creation at a mitigated bank in western North Carolina. Drought and outbreaks of a pathogen (Ranavirus) were the primary causes of low juvenile production from 1998 to 2002, but despite site perturbations, the breeding population of *A. maculatum* remained relatively stable. This study shows the need to be prepared for non-success as a result of natural phenomena, and emphasizes that even natural events can impact populations.

Additional factors that can limit the success of a mitigation project include:

- The need for consistent definitions;
- The high spatial and temporal variability of project sites;
- The limited number of reference wetlands in the area available for comparison. In addition, comparisons are 1:1 regardless of similarities or lack thereof between reference sites;
- The limits of power of any statistical comparisons due to sampling inconsistencies;
- Insufficient replication;
- Land managers looking for an “endpoint” and not acknowledging that endpoints can vary;
- No pre-alteration comparisons of site; and
- Natural disturbances or stress levels that may preclude functional equivalency (as with Petranks et al., 2003).

Though we strive to discontinue anthropogenic alterations of our nation’s wetlands, unacceptable, natural losses in many parts of the country may persist. For example, Louisiana’s coastal wetlands continue to dwindle because of flood control systems built along the Mississippi River during past decades. Effective wetland management policies of the future will have to address potential activities, as well as the activities of the past which have attributed to continual wetland loss.

In Florida, phosphate mining has affected both the quantity and quality of local wetlands. The following chapter will investigate the mining process and reveal how our need for phosphate is conflicting with our desire to preserve environmental health.

Chapter 2: Phosphate Mining and Wetland Alteration

Introduction

Phosphate mining occurs in Idaho, Louisiana, Mississippi, North Carolina, Texas, Wyoming and Florida, but singularly the State of Florida provides approximately 75 percent of the nation's supply of phosphate fertilizer and about 25 percent of the world supply (FIPR, 2004b). Most of the phosphate mines in Florida are located in the northern and central portion of the state, which include an area called "Bone Valley" that stretches from Polk County southward through portions of Hillsborough, Manatee, Hardee, Sarasota and DeSoto counties (FIPR, 2004b). About 95 percent of the phosphate rock recovered in Florida is used in agriculture (90 percent is used to produce chemical fertilizer and 5 percent is used as feed supplements for livestock). The remainder is added to a variety of consumer products including soft drinks, toothpaste, bone china, film, light bulbs, vitamins, flame-resistant fabrics, optical glass and shaving cream (FIPR, 2004b).

Chapter 2 will describe the phosphate mining process and the subsequent environmental repercussions. This chapter will also illustrate how the mining process affects wetland ecosystems, and how the phosphate companies are participating reclamation.

The Phosphate Mining Process

The local geology plays a major role in mine location; mines are typically situated in areas where the commodity is shallow enough to economically extract (FDEP, 2005). There are four types of surface mining – open-pit mining, strip mining, mountaintop mining and dredging. Open-pit mining involves extracting rock or minerals from the earth from an open pit or borrows. In Florida, phosphate is mined using this technique. The following map (Figure 4) illustrates the distribution of mines in Florida. The colored

dots represent different mineral commodities, and each dot may represent more than one mine.

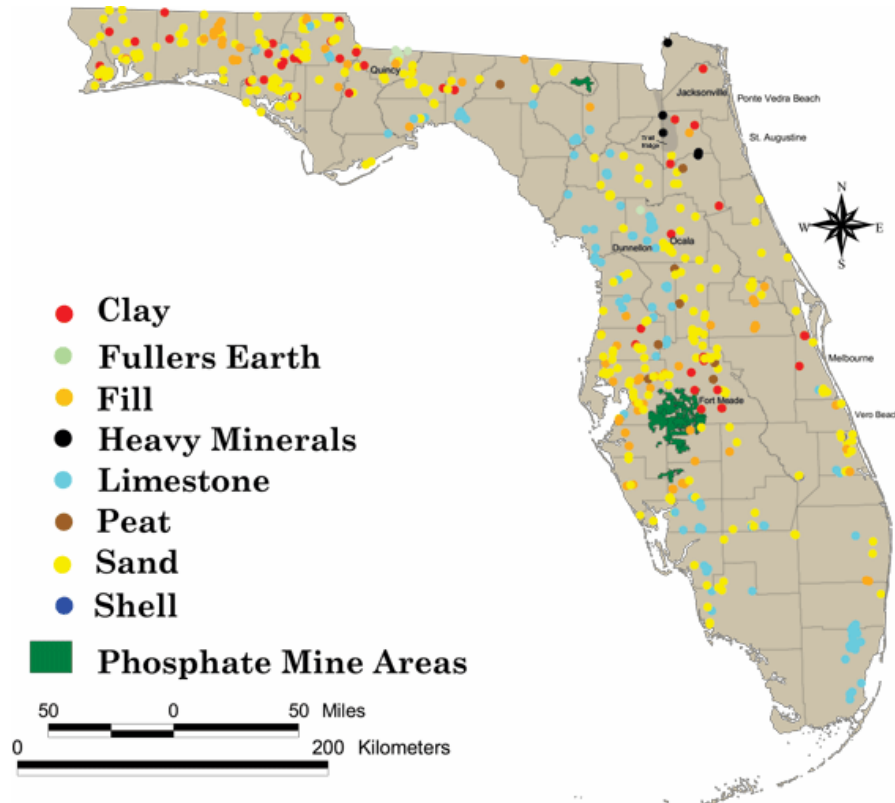


Figure 2.1: Distribution of Mines in Florida (FDEP, 2006c)

Mine development involves: (1) prospecting and exploring to locate and delineate the ore resource; (2) economic, environmental and technical feasibility assessment of the ore body; (3) planning and designing the mine layout, site infrastructure and the mining sequence; (4) obtaining relevant government permits and approvals; and (5) constructing and commissioning the operation. Phosphate ore is found 15 to 50 feet beneath the ground in a mixture of phosphate pebbles, sand and clay known as the phosphate matrix. Dragline excavators (draglines) remove approximately 15 to 20 feet of overburden (soil on top of the matrix), pile it in nearby spoil piles, and transport the underlying phosphate matrix to shallow containment areas (FIPR, 2004b).

Next, high-pressured water is used to make a slurry, which is sent through pipes to the beneficiation plant. The beneficiation (or concentration) process removes contaminants and barren material prior to further processing. The naturally occurring impurities contained in phosphate ore depend heavily of the type of deposit (sedimentary or igneous), associated minerals, and the extent of weathering. Impurities can include organic matter, clay and other fines, siliceous material, carbonates, and iron bearing minerals (International Fertilizer Industry Association and the United Nations Environment Program [IFA-UNEP], 2001). Iron minerals may be present in the form of magnetite, hematite and goethite, and are typically removed through scrubbing, size classification, or magnetic separation. The concentration of these characteristics influences the beneficiation processes used (IFA-UNEP, 2001).

At the beneficiation plant, the slurry is moved through a series of vibrating screens that physically separate the clay, sand and pebble-sized particles. Other beneficiation processes include crushing/grinding, water washing, and hydrocyclones. The clay particles are then pumped through pipelines into storage ponds (clay settling ponds) where they sink to the bottom. The sand and phosphate concentrate are moved to a flotation plant where reagents are added to separate the two. The sand-tailings are used to back-fill the mined areas, and the phosphate concentrate is sent to dewatering tanks (FIPR, 2004b). Following beneficiation, the concentrated phosphate rock is converted to phosphate fertilizer in chemical plants, producing a gypsum by-product that is stacked near the plant. After the mining process has been completed, the final land uses include reclaimed land (about 50 - 60 percent of the landscape), clay settling areas and chemical plants (40 percent of landscape), and transportation lines and gypsum stacks (about 10 percent of the land). The mining process is illustrated below (Brown, 2005).

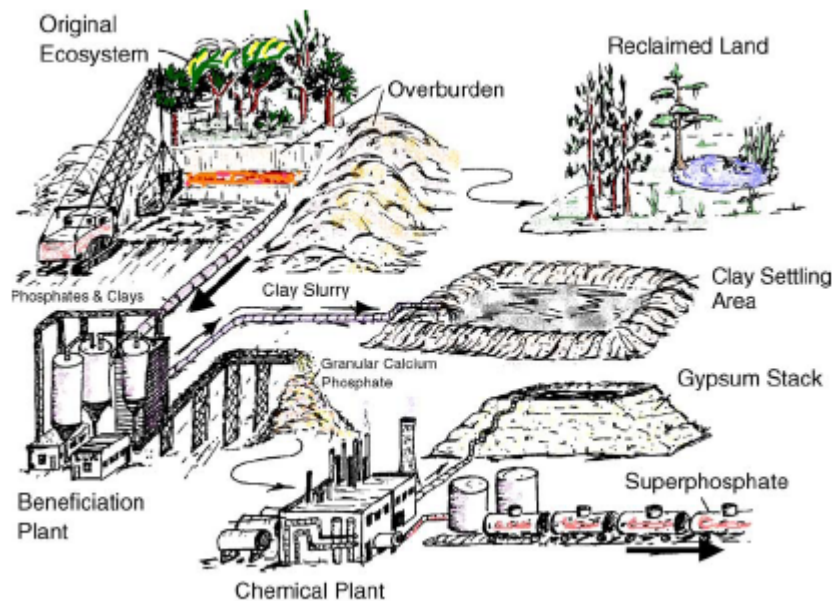


Figure 2.2: Schematic Diagram of Phosphate Mining Process (Brown, 2005).

The Environmental Repercussions of Mining

Almost every facet of phosphate mining affects the native environment. Extraction and beneficiation change the natural landscape by removing topsoil and vegetation, excavating and depositing overburden, disposing processing wastes, and underground mining-induced surface subsidence. Surface and groundwater may be adversely affected by the release of processing water, the erosion of sediment, the leaching of toxic chemicals from overburden and processing wastes, and by dewatering operations. Intensified local air pollution may result from the release of emissions, such as dust and exhaust gases (IFA-UNEP 2001). In their 2001 report, “Environmental Aspects of Phosphate and Potash Mining,” the International Fertilizer Industry and United Nations Environment Program (IFA-UNEP) outlined the following additional impacts:

- Changes in air quality by emissions of:
 - Dust;
 - Exhaust particulates and exhaust gases such as carbon dioxide (CO₂); carbon monoxide (CO), nitrogen oxides (NO_x), and sulfur oxides (SO_x);

- Volatile organic compounds (VOCs) from fueling and workshop activities; and
 - Methane released from some geological strata.
- Changes in water quality by:
 - The release of slurry brines and contaminants into process water;
 - The erosion of fines from disturbed ground such as open-cut workings; overburden dumps and spoil piles and waste disposal facilities;
 - The release or leakage of brines; and
 - The weathering of overburden contaminants, which may then be leached.
- Changes in social goods and intangible values (such as community lifestyles, land values and the quality of the ecosystem in the vicinity of the mine site) by factors such as:
 - Modification of the landscape;
 - Noise and vibration from activities such as blasting and the operation of equipment; and
 - Changes in wildlife habitat (IFA-UNEP, 2001).

In Florida, phosphate mining began in the mid 1960s. In 1990, there were 11 phosphate companies operating in Florida; by 2004, as a result of the changes in ownership and corporate buyouts, there were three (FIPR, 2004b). Each company has numerous mines, ranging in size from 4,500 acres to about 21,000 acres with the average of about 10,000 acres. Currently, there are two active mining areas in Florida known as the Northern and Southern Phosphate Districts, where about 5,000 acres of land are mined each year (Brown, 2005).

The dominant producer in today's Bone Valley is the Mosaic Company, created in 2004 when IMC Global combined with Cargill's crop-nutrition business (Hartley, 2005). Cargill had been a relative newcomer to Florida, buying several large phosphate operations in Bone Valley and the Tamp Bay area in the 1990s. Mosaic also continues to mine former IMC properties at Hopewell and Kingsford. Other than Mosaic, Bone Valley's contending producer is CF Industries (Hartley, 2005).

As the scale and extension of operations grow, concern continues. The raw phosphate is essential for sustaining the world's agricultural output. As there is no synthetic form of phosphate, the only supply is from natural deposits located beneath the earth's surface in certain parts of the world. In general, the phosphate companies in Florida have a good record of reclaiming their strip-mined lands. IMC-Agrico (now under The Mosaic Company) has won several environmental awards for their reclamation projects. However, a series of catastrophic incidents involving several companies has raised doubts about the phosphate industry's environmental image.

In June 1994, a 15-story-deep sinkhole opened in an 80 million-ton pile of phosphogypsum waste (known as a gypsum stack) at IMC Agrico's New Wales plant (Satchell, 1995). The cave-in dumped 4 to 6 million cubic feet of toxic and radioactive gypsum and wastewater into the Florida aquifer, which provides 90 percent of the State's drinking water. The company has voluntarily spent \$6.8 million to plug the sinkhole and control the spread of contaminants in the groundwater (Satchell, 1995).

Since 1990, at least 6 dams in Florida have failed, three of them since October 1994 (Satchell, 1995). Billions of gallons of effluent have inundated nearby land, polluting streams and killing fish and other aquatic life. The biggest spill occurred in October 1985, when an IMC-Agrico dam burst and released 1.8 billion gallons of water. A 482 million-gallon torrent engulfed Hillsborough County, and two people nearly drowned (Satchell, 1995).

In December 1997, a breach of a phosphogypsum stack owned by the Mulberry Cooperation resulted in the release of 50 to 60 million-gallons of toxic wastewater into the Alafia River in Polk County, causing massive aquatic deaths (Roth, Novey & Regalado, 2005). The IFA-UNEP maintains that the mining industry has improved over the decades, although they acknowledge that challenges remain.

Odum et al. (1983) documented the interactions between the phosphate industry and wetlands on several sites in and around the Northern and Central Phosphate Districts. The authors evaluated landforms created by mining, the use of detritus as a seed source, tree planting on waste clay sites and the use of tree cores from cypress trees to monitor stress in wetlands. The first section of this report demonstrates how data concerning tree

growth can be used to design reclaimed landscapes and reveals that the major factor controlling vegetational succession appears to be proximity to natural seed sources (Odum et al., 1983).

Rushton (1988) followed with a study focusing on the wetland establishment on waste clay sites and documented the following results: a pattern of arrested succession emerged that seemed to be related to the distance to seed source; the soils characteristic of mined lands tended to have higher concentration of clay than native soils, and thus had lower air and water movement; and soils and water in phosphate mined area were higher in phosphorus and tended to naturally favor low diversity ecosystem types characterized by early successional colonizing species.

Wetland Alterations from Phosphate Mining

Wetland ecosystems are defined by two aspects; (1) abiotic conditions (temperature, soil type, terrain, disturbance regime, etc.) and resources (water, nutrients, sunlight) and (2) the trophic structure or feeding relationships among the species that shape the flow of energy through the ecosystem (Brown, 2005). An alteration to a wetland system may disrupt the abiotic components (soil structure might be changed, as with mining) as well as the trophic structure (the loss of primary producers, as when land is cleared).

Several authors have suggested that in the past 200 years between 30 – 50 percent of wetlands within the conterminous United States have been lost, and many of the remaining wetlands are degraded (Bingham et al., 1990; Dahl & Johnson, 1991; Kentula, 1992). Wetland alterations result from both natural and anthropogenic phenomena. Indirect impacts from pollutants, urban runoff, and invasion by non-native species can degrade wetlands as effectively as the direct impact of an excavation. Alterations can occur as discrete events (i.e. excavation activities, drainages, spills, fire and clearing) or transpire gradually (changes in hydrology, water level, sediment loads, concentration of chemicals, or the composition of biological communities). These alterations may induce subtle changes to the ecosystem or may devastate the landscape.

Wetlands altered by chemical contamination or biological changes may appear functional because the alteration may not affect the wetland's physical appearance, but in reality the system has been affected. Though alterations to a wetland are localized, the cumulative effects of small-scale wetland alterations can have regional or national effects (Bingham et al., 1990).

The entire surface mining process ravages the natural environment both directly and indirectly. Direct effects include changes in water quality and sediment deposition, in addition to destruction of wetland habitat. Indirect alterations can consist of alteration to the natural hydrology or the release of sediment and particle-laden waters into the wetlands and waterways (Kusler & Kentula, 1990). Mining may disconnect wetlands from natural water sources or from outfall structures which discharge excess water out of the wetland (Bedford, 1996). Increased turbidity within the water column due to discharge or spills during mining may affect the amount of light reaching vegetation below the surface. Increases in turbidity may result in mortality of the aquatic plants which provide habitat and food for other organisms (Kusler & Kentula, 1990).

The natural landscape suffered as a result of years of mining, until laws were passed to rectify the situation. Beginning in 1899 and culminating in the 1970s, federal, state and local laws came into effect which required the reclamation of impacted wetlands. Although the land reclamation process has been outlined and detailed, the increasing amounts of reclamation literature suggest that these guidelines are outdated and lack enforcement.

Wetland restoration and creation are relatively new fields; few engineers are trained in ecology and few ecologists have experience in engineering methods (Mitsch & Wilson, 1996). Failures are attributed to lack of understanding basic yet dynamic principles of wetland ecology. A need for a landscape point of view has prompted my investigation of international mining and how different countries are addressing wetland reclamation.

Chapter 3: International Mining and Wetland Preservation

Introduction

The environmental repercussions of phosphate mining and wetland reclamation are not only pertinent to central Florida. Of the numerous countries producing phosphate, the United States produces 44,600 metric tons (FIPR, 2004b). Extensive transportation and industrial infrastructures, which enhance the production and exportation of the product, allow the United States to generate this enormous amount. Morocco and China rank a close second and third, respectively. Morocco's phosphate reserves are estimated to be nearly six times that of the United States, which may indicate a potential shift in production power in the future. Additional active phosphate mining occurs in Russia, Tunisia, Jordan, Brazil, Israel, South Africa, Syria, Togo, Senegal, Australia and Canada (FIPR, 2004b).

All of the phosphate mining countries suffer the environmental ramifications of surface mining and wetland degradation. Laws and programs aimed at specific areas and in specific parts of the country do not benefit wetland ecosystems on a landscape level. In order to avoid piecemeal solutions to a wide-spread problem, my second objective is to investigate international phosphate mining to examine how different countries and international organizations are addressing the environmental implications. Chapter 3 will investigate the international environmental and mining communities and will offer some international solutions to wetland preservation.

The International Environmental Community

Wetlands are as varied internationally as locally. Table 3.1 outlines the range of natural habitats within different temperate and tropical wetland ecosystems.

Table 3.1: Range of Natural Habitats within Different Temperate and Tropical Wetland Ecosystems (Bacon, 1997).

Climate (Representative Country)	Wetland Ecosystem Present
Temperate Climate (such as The United States) ¹	Rock bottom Unconsolidated bottom Aquatic bed Rocky shore Unconsolidated shore Emergent marsh wetland Forested wetland
Tropical Climate (such as Trinidad and Tobago) ²	Saturated forested wetland Tidally saturated forested wetland Permanently flooded emergent herbaceous wetland Seasonally flooded emergent herbaceous wetland Intermittently exposed unconsolidated shore Semi-permanently flooded aquatic bed Channels and pools Marginal terra firma
Oceanic/Coastal Systems (such as Surinam) ³	Stagnant brackish and hypersaline pools Drying up lagoons Tidal lagoons Soft tidal mudflats High Fiddler-crab zone of tidal mudflats Firm and tough clay banks Lower foreshore sandy beach Back slope sandy beach

¹Cowardin et al., 1979; ²Bacon, 1988; ³Swennen & Spaans, 1985

Wetlands variability stems from seasonal fluctuations in rainfall or inundation patterns. The United States National List of Plant Species includes 6,728 species. Under more extreme conditions (such as the Arctic Tundra, high mountain peat bogs and hypersaline salt marshes in the dry tropics) this diversity is lower, even though a range of highly specialized plants will often be present (Bacon, 1997).

Wetland ecosystems are highly productive. The availability of water, which transports nutrients and removes waste products, and the interaction between plant roots and microscopic organisms able to use nitrogen, allow wetland plants to grow rapidly and produce large quantities of organic matter. With tropical wetland plants, such as

mangroves, primary production can go on all year and reach levels comparable to the most intensively mechanized agricultural production (Bacon, 1997). Table 2 lists the productivity of various ecosystems.

Table 3.2: Productivity of Selected Wetland Ecosystems (Bacon, 1997).

Wetland type	Location	Annual production, tons per hectare per year (above ground only)
Estuarine mangrove	Sri Lanka	12
Tidal salt marsh	Louisiana, USA	14
Riparian forest	Louisiana, USA	14
Freshwater (reed) marsh	Denmark	14
Freshwater (Papyrus) marsh	Kenya	30
Freshwater (reed) marsh	Wisconsin, USA	34
Tropical seagrass bed	Caribbean	70

The species diversity and high production levels of wetland plants support very diverse animal communities. Wetlands of the Ebro Delta in Spain support 48 resident species of fish, 29 mammals and 46 resident or migratory birds (MAPA, 1991). Among the 104 species recorded in the Black River Morass, Jamaica, were 11 seabirds, 36 waterfowl, 7 birds of prey, a kingfish and 49 forest birds (Bacon, 1987). Approximately 251 species have been found in the Cache River Basin in Illinois, USA (FWS, 1994). According to Weigers (1990), 30 species of birds commonly breed in the somewhat restricted wetland forests in Western Europe.

The livelihood and culture of large numbers of people, in almost every country of the world, depend on wetland resources. A major portion of fisheries production, most hunting, much forest production and a significant part of ecotourism, as well as elements of heritage and environmental quality have evolved from nearby wetland ecosystems (Bacon, 1997). Twenty percent of commercial fish in Australia are caught in mangrove resources, while 35 percent are food for commercial marine species (Robertson & Duke, 1987). Other commercial and economic uses of wetlands include:

- Fuel such as firewood, alcohol, charcoal, wood (curing fish, smoking rubber and firing bricks), peat;

- Medicines (from fruit, sap, bark, leaves) such as diuretics, purgatives, astringents, vitamins (mainly b group), treatments for: arthritis, leprosy, catarrh, rheumatism, skin rashes, hemorrhoids, snake bite, tuberculosis;
- Agricultural, horticultural and aquaculture products such as fodder, fish feeds, green manure, peat/compost/fiber landscape plants, planting for coastal protection, ornamental pond plants, insect repellent (Bacon, 1997).

It is important to stress, however, that it is not sufficient just to protect the populations of plants and animals that are directly exploited – their health, survival, and sustainability depend on maintaining the whole complex of biodiversity that characterizes wetland ecosystems. The protection of such ecosystems is one of the goals of international environmental organizations such as the Ramsar Convention.

The Convention on Wetlands of International Importance Especially as Waterfowl Habitat, signed in Ramsar, Iran in 1971 (referred to as the Ramsar Convention) is among the oldest global multilateral environmental agreements. Since its inception, its 138 Contracting Parties (encompassing at least 4 countries, including the United States) are committed to the conservation and wise use of all wetlands, to the designation and management of wetlands of international importance (Ramsar sites), and to international cooperation in conservation implementation (Rodriguez, 2004). A variety of inland, coastal and near-shore marine wetland ecosystems are considered in these proceedings.

The Convention owes its origins to concerns in the 1960's over increasing destruction and degradation of wetlands, and the impact to migratory waterfowl (Hails, 1997). As such, approximately half of the current Ramsar sites have been designated for the bird populations. The Convention works closely with non-governmental environmental organizations, scientific expert networks, water bird ecologist and general researchers. The four non-government organizations that work closely with the Ramsar Convention, which are recognized as the “partner organizations,” are Birdlife International, the World Conservation Union, Wetlands International and the World Wide Fund for Nature (Pritchard, 1996).

Agencies such as the World Bank also play an important role in conservation efforts, and there are publications from the Organization for Economic Cooperation and Development which contributes to the wealth of information being shared (Pritchard, 1996). In 1997, the International Institute for Environmental and Development produced a valuable directory of 150 sets of national, international and sectoral guidelines, and at least twice this number is known to exist elsewhere (Roe, Dalal-Clayton & Hughes, 1995).

Literature searches reveal that although the Ramsar Convention requires regular monitoring of wetland sites to detect changes in ecological character, the examples of successful long-term monitoring, especially in tropical wetlands, are limited. Bennun (2001) remarked that monitoring schemes run into three kinds of difficulties - conceptual, logistical and political. The author emphasized that the sole goal on any monitoring scheme should be to assess deviation from the determined reclamation recommendation, so that appropriate corrective action can be taken. This assumes that adequate baseline information exists, and Bennun suggested that the global community does not have such data.

Amezaga, Santamaria and Green (2002) maintained that international wetland conservation approaches are based on the preservation of isolated sites considered to be of special importance, thereby creating a “fragmented” world view. The distribution of wetlands is critical to determining wetland connectedness even in the absence of direct hydrologic links. The authors hypothesized that the effect of wetland loss on regional diversity might be much larger than previously expected, and thus call for more research linking local species richness. In a similar paper, Amezaga and Santamaria (2000) concluded that the shortcoming in the current management of European wetlands stem from the contradiction of creating highly protected natural reserves surrounded by unprotected non-natural territory. Thus, as is suggested for the United States, a regional and landscape approach to wetland management (within a continental context) was recommended.

Some countries have attempted to use trade agreements to force less environmentally-advanced countries to develop wetland protection policies, especially

when they are close enough to have a large impact on one another. An example is the North American Free Trade Agreement (NAFTA) between Canada, United States and Mexico (Marshall, 2000). While the three participating parties were negotiating NAFTA, they also addressed environmental concerns that they all shared. Prior to this agreement, all three countries had developed other environmental agreements focusing on border resources that the parties shared. Under NAFTA, economic and environmental joint decisions were achieved. Another example of the success of trade agreements is the General Agreement of Tariffs and Trade (GATT). The intent of GATT is to promote fair trade between the members' parties (Marshall, 2000). GATT's ability to limit impacts on the environment was tested when the United States tried to impose trade sanctions on Mexico in an attempt to enforce the Marine Mammal Protection Act (MMPA). Mexico was killing dolphins in their tuna nets; the issue resulted in two cases that were brought before the courts (Marshall, 2000).

Relatively few large-scale inventories of the world's wetlands exist because of the difficulties of spatial scale, associated cost, and multiple objectives that drive classification. Kingsford et al. (2004) evaluated the extent of wetlands across New South Wales in Australia. Results indicated approximately 5.6 percent (4.5 million hectare) of the area is wetland. Currently, conservation reserves only protect 3 percent of this area. Several books have been written about the various wetlands all over the world. The authors discuss the wetlands in isolation, however, with each referencing only their particular areas. Unlike these authors, the editors of *Wetlands of the World* bring together most of the basic information present on many types of wetlands in various locales.

The first volume of this work covers wetlands in Africa, Australia, Canada and Greenland, the United States, the Mediterranean, Mexico, Papua New Guinea, South Asia and tropical South America. The second and third volumes include Central America, Western, Northern and Central Europe, Northern and Western Asia, the Middle East, and Indonesia, the Far East, and New Zealand (Whigham, Dykyjova & Hejny, 1993). Each chapter describes and discusses the geologic, geomorphic and hydrologic controls on wetland formation within the region; the distribution of wetlands; their flora, fauna and

ecological characteristics; human impacts; and recommendations for conservation and management of these wetlands.

Some of the reviews are more complete than others, and some of the factors are discussed more than others (Whigham et al., 1993). For example, the chapters on South Asia and Australia cover geology, geomorphology and hydrology only in sparse detail, while the chapters on the United States focus almost exclusively on hydrological factors and ignore the geologic and geomorphic conditions that produce the regional hydrology. The differences in emphasis reflect the differences in political structure and wetland classification. Rather than attempt a single, unified classification, the authors subscribed to the wetland schemes and definitions put forth by each region, thereby making interregional comparison difficult. All of the authors did agree on a series of recommendations:

- Universal adoption of the Ramsar convention for international conservation;
- More complete national wetlands inventories, with emphasis on remote sensing;
- Detailed assessments of human impacts and introduced species; and
- The development of regional management and conservation plans that focus on wetland preservation and limiting their destruction (Whigham et al., 1993).

International wetland organizations are also addressing the lack of cohesive wetland literature. The World Wide Fund for Nature (WWF) Australia is conducting annual reviews of the quality of public environmental reports released by companies' signatory to the Australian Mineral Industry Code for Environmental Management (IFA-UNEP, 2001). The Fund recognizes the importance of environmental reporting in communicating environmental and social performance to stakeholders. The reports are assessed using nine performance indicators, which were selected to assist stakeholders using environmental reports as tools for assessing the environmental and social performance of the company. The stakeholders believe that environmental reports should reflect the company's performance, define goals, address stakeholder concerns arising from operations and provide independent verification of the contents (IFA-UNEP, 2001). The WWF emphasizes the importance of external third party verification to assess company performance. Other international organizations participating in wetland

research include Wetlands International, which was formed in 1995 when three organizations joined to create a wetland alliance.

Nations that previously did not embrace wetland conservation are now addressing wetland degradation socially and politically. Within the recent adoption of The Federal Policy on Wetland Conservation, the Government of Canada has decreed to achieve the following goals:

- Maintain the benefits derived from wetlands throughout Canada;
- Achieve “no net loss” of wetland functions on federal lands and waters;
- Enhance and rehabilitate wetlands in areas where the continuing loss or degradation of wetlands or their functions have reached critical levels;
- Recognize wetland functions in resource planning, management, and economic decision-making with regard to all federal programs, policies, and activities;
- Secure wetlands of significance to Canadians; and
- Recognize sound, sustainable management practices in sectors such as forestry and agriculture that make a positive contribution to wetlands conservation while also achieving wise use of wetland resources (Rubec, 1992).

Following suit, the Costa Rican government adopted a Policy for Wetland Protection, the first of the kind in Central America, which promotes the conservation and wise use of the country’s wetland ecosystems (Union Mundial para la Naturaleza [IUCN], 2001).

While some nations are coping with the environmental problems of wide-scale mining and looking ahead to reclamation projects, some countries are just beginning to recuperate. In Tanzania, soils are of low quality and have deteriorated to a level that heavy fertilization is essential if they are to be productive (Mwalyosi, 1988). Large phosphate mines are located at Minjingu Hill in Lake Manyara Catchment basin. The life span of the mine was to be between 10 to 15 years, but the low production may increase the life span to 50 years (Mwalyosi, 1988). A resource conflict has arisen between the mining industry, the indigenous people and the state official because the mine is located near the wildlife conservation areas of Manyara and Tarangire, and

disrupts the major dispersal route of the large herbivores of the Tarangire National Park (Mwalyosi, 1988).

Further complications arise because the mining community is expected to increase exponentially, and residents are occupying the mining area faster than accommodations and social services can be situated. The mine is expected to attract small businesses, shops, and hotels to cater to the growing population. The increased population will require more land for agriculture and livestock, and land-use disagreements are inevitable. Meanwhile, the animals around the mining site are at risk from ecological and environmental changes introduced by the mining industry (Mwalyosi, 1988).

Mwalyosi (1988) urges the development of an environmental monitoring team and recommends that other agencies, such as the Tanzania National Environmental Council and the State Mining Corporations, cooperate to minimize the inevitable impacts. In addition, the author calls for an integrated resource management plan that could evaluate the ecological and socioeconomic factors associated with the rise of the international phosphate industry.

The International Mining Community

Currently, most phosphate rock production worldwide is extracted using opencast dragline or open-pit/shovel excavator mining methods. This method is employed widely in parts of the United States of America, Morocco and Russia (IFA-UNEP, 2001). Underground mining methods are currently used in Tunisia, Morocco, Mexico and India. The area of land surface affected by these operations is generally small and limited to the area immediately adjacent to the access decline or shaft (IFA-UNEP, 2001). The mining of flat-lying thin orebodies, as found in Florida, affects a far wider area of land than the mining of thicker or steeply dipping orebodies as found in Brazil and Idaho.

Some international companies are beginning to adopt life-cycle techniques of mining. The industry has moved from 'end-of-pipe' solutions, towards a pollution prevention strategy. This strategy requires an integrated, holistic view of activities (IFA-UNEP, 2001). Tools have been developed to assist management, including cleaner production, life cycle assessments and industrial ecology. Each aspect of the mine

development looks at the life cycle of the product or service (see Figure 3.1), to identify where the major environmental issues or problems may arise, and where the most cost-effective solutions can be developed. Planning for the life of the mine, including closure and site rehabilitation, allows for a more efficient and environmentally effective outcome. As an example, WMC Australia participates in environmental reporting and records their input and output flow amounts, including emissions, wastes and net land disturbance, to document their environmental performance (IFA-UNEP, 2001).



Figure 3.1: The Mining Life Cycle (IFA-UNEP, 2001).

The concept for the schematic is drawn from the Natural Resources Canada report “Sustainable Development and Minerals and Metals.”

The ISO 14000 Environmental Management System and the European Union Eco-Management and Audit Scheme (EMAS) standards provide frameworks and guidelines that companies can use to develop their own environmental management systems (IFA-UNEP, 2001). Many phosphate mining companies all around the world are looking at the ISO 14000 standards to guide the development of systems that are specific to their needs.

In Jordan, the Jordan Phosphate Mines Company Ltd. attained the ISO 14000 Environmental Management System and ISO 9000 Quality Assurance certification for most of their operations (IFA-UNEP, 2001). Today, the company acknowledges the benefits these systems offered in increasingly more demanding consumer markets. In Brazil, Fosfatados Fertilizantes adopted the “Program 5S” quality management system in 1995. This system, which originated in Japan, is based on the five principles of:

- Seiri – Sense of selection;
- Seiton – Sense of tidiness;
- Seisou – Sense of cleanliness;
- Seiketsu – Sense of health; and
- Shitsuke – Sense of self-discipline.

Based on these principles, Fosfatados Fertilizantes developed the following company objectives for improving their work environment: (1) prevent accidents; (2) provide incentives for creativity; (3) eliminate waste; (4) promote team-work; and (5) improve the quality of service and products (IFA-UNEP, 2001). Success of this system is accomplished by changing behaviors at all levels of the company to improve the working environment, welfare and quality of the products and services. The performance of the system is visible: the company system has won the Brazilian mining safety award for seven years running. Additionally, the economic performance of the company has improved in recent years (IFA-UNEP, 2001).

Along with process improvements, financial compensation is offered to areas ravaged by historical mining activities. The Wellington mining company agreed to pay \$8.5 million U.S. dollars in compensation to the Pacific island nation of Nauru for environmental damage caused by phosphate mining before 1968. It joins Britain and Australia in contributing to a compensation package that totals \$76 million dollars (“Compensation Offered,” 1994).

Currently, countries vary significantly in their approach to wetland monitoring and protection. Some developing countries have yet to establish wetland protection policies. Based on literature searches, it appears that compared to other nations, the United States has developed more advanced inventory and comprehensive study of their

wetlands. Even with an advanced inventory system, however, studies continue to show losses of United States' wetlands to be estimated at over 50 percent of what they were in the early 1600s. Canada and Europe have also experienced losses over 50 percent although neither has monitored wetland status as long or as thoroughly as the United States. Other parts of the world vary as to how many wetlands have been lost or degraded (Marshall, 2000). Most wetland successes originate from Ramsar participants. The Contracting Parties are required to report on their progress of implementing their commitments under the Convention by the submission of triennial National Reports to the Conference of the Contracting Parties. The National Reports then become part of the public record.

Thus, internationally progress is being made. There does seem to be a common philosophy between our national approach and the international approach, but the international standards seem deficient. The international arena does not have an international Clean Water Act for guideline unification, and countries need the equivalent of a global EPA to maintain some level of enforcement power. Treaties that may be binding under the United Nations (U.N.) Charter are not always adequate. Economic sanctions imposed through the U.N. are difficult given the wide range of opinions of the importance of wetland ecosystems.

The fact that international guidelines are distinct from the United States is not unexpected. The hydrology, geology, climate, political structure, social values and the value of wetlands are different depending on the country involved. Globally, however, the overall lack of wetland preservation is astonishing. The commitment to environmental protection and the capacity to set forth the necessary enforcement should be standard.

It is difficult to suggest a cohesive form of environmental conservation. The framework of sound scientific analysis is necessary for appropriate legislation to develop. The myriad of unsuccessful wetland mitigation points to a need for more comprehensive methods of evaluation. One such method has been outlined by Danish scientists. In the 2002 National Environmental Research Institute (NERI) report, "Microorganisms as Indicators of Soil Health," Nielsen and Winding (2002) promote the use of microbial

indicators in terrestrial monitoring programs, and provide recommendations for such indicators to be implemented into a Danish terrestrial monitoring program. In addition, the European Union's COST Action 831 (www.isnp.it/cost/cost.htm), a cooperative project by international scientists, is encouraging the use of microorganisms as indicators of environmental impacts in soil monitoring. A European Monitoring and Assessment Framework on soil has subsequently been proposed to provide policy-makers with relevant information on soil, and to bring together the wealth of soil information derived from current national soil monitoring programs (Nielsen & Winding, 2002).

Microorganisms respond quickly to changes, hence they can rapidly adapt to environmental conditions. The microorganisms that are best adapted, flourish. This adaptation allows microbial analyses to be discriminating in soil health assessment, and changes in microbial populations and activities may therefore function as excellent indicators of changes in soil health (Kennedy & Papendick, 1995; Pankhurst et al., 1995). Microbes also respond quickly to environmental stress as they have intimate relationships with their surroundings, due to their high surface to volume ratio (Nielsen & Winding, 2002). In some instances, changes in microbial populations (or activity) can precede detectable changes in the physical and chemical properties of the soil, thereby providing an early sign of soil improvement or an early warning of soil degradation (Pankhurst et al., 1995). Because microorganisms are involved in many soil processes, they may also give an integrated measure of soil health, an aspect that cannot be obtained with physical/chemical measures alone. In terms of benefits for vegetation, their role in binding sediment particles to improve rooting medium structure is critical (Kadlec & Knight, 1996). Microbial presence in the root zone of plants is also necessary for sustained plant growth.

The activities of microbial communities in sediments are important for sediment quality, and the biological transformations of major nutrients influence plant growth (Kadlec & Knight, 1996). Soil microorganisms also affect the physical properties of the soil, such as the water holding capacity, infiltration rate, erodibility, and susceptibility to compaction. Production of extra-cellular polysaccharides and other cellular debris by microbes maintains soil structure, as these materials function as cementing agents that

stabilize soil aggregates (Nielsen & Winding, 2002). Microbes also contribute to the enhanced biotransformation of toxic chemicals in wetlands (Daane, Harjono, Zylstra and Haggblom, 2001).

Given the multiple functions of soil microbes (e.g., soil formation, toxin removal, contributions to the elemental cycles), it is difficult to identify any one property as a general indicator of soil health. Instead, Nielsen and Winding (2002) suggest using “end points” to determine the soil ecosystem functions of interest (Table 3.3).

Table 3.3: End Points of Terrestrial Monitoring and Corresponding Soil Ecosystem Parameters (Nielsen & Winding, 2002).

End Point	Soil Ecosystem Parameter
Atmospheric balance	C-cycling
Soil ecosystem health	Biodiversity
	C-cycling
	N-cycling
	Microbial biomass
	Microbial activity
Soil microbial community health	Key species
	Biodiversity
	C- cycling
	Microbial biomass
	Microbial activity
Leaching to groundwater or surface run-off	Bioavailability
	N-cycling
Plant health	Bioavailability
	N-cycling
Animal health	Key species
	Microbial biomass
Human health	Bioavailability
	Key species

Microbial indicators of soil health cover a diverse set of microbial measurements, reflecting the multi-functional properties of microbial communities in the soil ecosystem. In the NERI report, microbial indicators included bacteria, fungi and/or protozoa. The authors grouped the indicators according to different soil ecosystem parameters (identified in Table 3.3), and this assemblage is presented in Table 3.4 (Nielsen and

Winding, 2002). These tables can be used to specify the indicator and parameter best suited to a particular mitigation site. Generally, indicators should be selected on the basis of ease of measurements, reproducibility, and their sensitivity towards the variables identified as pertinent to soil health for the selected area. The selection of indicators can be broad to give managers an overview of the wetland system, or detailed to better explain underlying trends. For example, if biodiversity was chosen as the parameter for measuring microbial community health, four techniques (Table 3.4) can be used. This list is not complete, but offers a large number of techniques that are presently used.

Table 3.4: List of Microbial Indicators for Soil Health Monitoring (Nielsen and Winding, 2002).

Soil Ecosystem Parameter	Microbial Indicator	Methods
Biodiversity	Genetic diversity	PCR-DGGE
	Functional diversity	BIOLOG
	Marker lipids	PLFA
	Soil respiration	CO ₂ ⁻ production or O ₂ ⁻ consumption
C-cycling	Soil enzyme activity	Enzyme assays
	Methane oxidation	Methane measurements
	Methanotrophs	MPN PLFA
	Heterotrophic activity	CO ₂ evolution
N-cycling	N-mineralization	NH ₄ ⁺ accumulation
	Nitrification	NH ₄ ⁺ oxidation assay
	Denitrification	Acetylene inhibition assay
	N-fixation: Rhizobium	Pot test
	N-fixation: Cyanobacteria	MPN Nitrogenase activity
Microbial biomass	Microbial biomass: Direct methods	Microscopy PLFA
	Microbial biomass: Indirect methods	CFI, CFE SIR
	Microbial quotient	C _{micro} / C _{org}
	Fungi	PLFA or Ergosterol
	Fungal-bacterial ratio	PLFA
	Protozoa	MPN
	Bacterial DNA synthesis	Thymidine incorporation
	Bacterial protein synthesis	Leucine incorporation
	Community growth physiology	CO ₂ ⁻ production or O ₂ ⁻ consumption
	Bacteriophages	Microscopy for biomass
	Mycorrhiza	Microscopy Pot test
Key species	Human pathogens	Selective plating and/or PCR
	Suppressive soil	Pot test
	Biosensor bacteria	Remedios Microtox
Bioavailability	Plasmid-containing bacteria	Gel electrophoresis
	Antibiotic-resistant bacteria	Selective growth
	Incidence and expression of catabolic	Selective growth
	Genes	

Wetland soil analysis is also being achieved in other countries. Soil microbial parameters are being developed for the Domingos mine, an abandoned cupric pyrite mine in Southeast Portugal. A study designed by Pereira, Sousa, Ribeiro and Goncalves (2006) was initiated to evaluate the long-term effects of heavy-metal contamination on microbial community activity, and subsequently on important soil functions (e.g., nutrients cycling, decomposition of organic matter). Total metal contents, as well as physical and chemical parameters (organic matter moisture, pH and conductivity), were measured. The most sensitive microbial parameters were dehydrogenase activity (DHA) and potential nitrification (POTNI), which were highly depressed in the milling area. Results suggested that the high metal contamination levels and low pH adversely affected the biomass and the activity of soil microorganisms in the mining area, and thus affected the development of the wetland ecosystem (Pereira et al., 2006).

In their study of bacterial community structure in Scotland, Ellis, Morgan, Weightman and Fry (2003) investigated soil samples taken from sites with patchy metal contamination, and assessed diversity with a variety of approaches. The authors contend that the most appropriate methods for determining the effect of the contamination on soil health are assessing microbial activity, rather than the presence or absence of different cell types. Although culture-dependent approaches to microbial community analysis are often criticized for their selectivity, it is perhaps this discrimination that makes them useful for determining the impact of anthropogenic activity (Ellis et al., 2003).

The field of soil microbiology is relatively new, but the advances made and the plethora of laboratory techniques available is impressive. The next chapter will address the potential of microbial analysis for mitigated wetlands in the State of Florida. .

Chapter 4: Soil Microorganisms as Indicators of Functional Wetlands

Introduction

Hydric soils promote a very complex assemblage of microorganisms, which break down and transform a wide variety of substances. Wetland vegetation provides attachment sites for the microorganisms that make up the microbial communities, and when these plants die and accumulate as litter, they create additional material and exchange sites, as well as sources of carbon, nitrogen and phosphorous to fuel microbial processes (Paul & Clark, 1989).

Considerable interest has been focused on the functional assessment of wetlands, especially those created as a result of state mitigation policy. Several studies exist on the hydrology (Bedford, 1996; Niswander & Mitsch, 1995), seed banks (McKnight, 1992), vegetation ecology (Reinartz & Warne, 1993), microorganisms and mine waters (Ledin & Pederson, 1996), and wildlife (Leschisin, Williams & Weller, 1992) of mitigated wetlands, but comparable data is lacking for wetland soil microbiology's participation in mitigation success. Our understanding of the complex processes involving plants, microorganisms, soil matrix, and how these factors of the ecosystem interact is incomplete.

The literature suggests several reasons for our inability to successfully mitigate wetlands. The "successful" ones are those that remain part of the landscape for long periods of time, but these wetlands may not be healthy or properly developed. Recent hydrologic data shows that created wetlands are not achieving structural replacement, a minimum requirement for functional success (Cole & Brooks, 2000). Galatowitsch and van der Valk (1994) outlined provisions for the term "functional success." Thus, although created and mitigated wetlands may meet jurisdictional requirements, measures of their ecological "health" do not suggest a thriving system.

My intent is to supplement this review by suggesting the investigation of the microbial communities of mitigated wetlands. I will review literature describing the laboratory and field procedures necessary to evaluate the proper development of mitigated wetlands. This chapter will provide managers with a source of references that can serve as a framework for future wetland mitigation projects, and will offer effective tools for mitigation monitoring (i.e. comparing the microbial activity of their sites to natural wetlands).

Chapter 4 will discuss wetland soil characteristics (pre- and post-mining) and evaluate the role of microorganisms as bioindicators in the mitigation process. In addition, research projects designed to study the microbial aspects of wetlands will be summarized. Hopefully, in the near future, a microbial standard can be developed and applied to various national sites to support the equivalence of mitigated wetlands and natural wetland systems. This review can also help policy-makers establish an inference of wetland success.

Wetland Soil Characteristics

Wetland soil development is critical for the re-establishment of native vegetation after phosphate mining. Some of the substrates used in constructing a wetland in phosphate-mined areas include overburden, sand tailings, and clay (Nair et al., 2001). Reclamation of phosphate-mined lands usually begins on the clay settling areas. There are an estimated 160,000 acres of clay settling areas in Central Florida (Stricker, 2000). Clay soils are problematic for wetland development, however, because they have:

- Highly compacted soils due to the bulk density of clays;
- A lower amount of free oxygen, which is necessary to support the microorganism community;
- Elevated pH levels (very high levels of calcium);
- Very little soil organics (~0.5percent of soil organic carbon);
- Severe deficiencies in nitrogen available to plants;
- Soil structure instability;
- Extreme seasonal variations in top-zone moisture;

- Relative sterility because of the lack of native flora seed banks; and
- Poor drainage (Stricker, 2000).

On a positive note, clay settling area soils are very high in potassium (K) and phosphorus (P) - two key components of productive soils (Stricker, 2000). The following table describes the differences in mineral content and pH of sandy soils and clay settling area soils in Florida.

Table 4.1: Comparison of Florida Sandy Soil and Phosphatic Clay (Stricker, 2000).

Soil Component	Sandy Soil (ppm)	Phosphatic Clay (ppm)
Calcium	63	5,923
Magnesium	10	2,569
Potassium	67	332
Phosphorus	166	586
Zinc	1	5.6
Manganese	0.3	6.3

In a soil profile (Figure 4.1), the capital letters O, A, B, C and E identify the master horizons, and the lowercase letters are used for further distinctions of these horizons (United States Department of Agriculture National Resources Conservation Service [NRCS], 2006). Most soils have three major horizons -- the surface horizon (A), the subsoil (B), and the substratum (C). Some soils have an organic horizon (O) on the surface, but this horizon can also be buried. The master horizon, (E), is used for subsurface horizons that have a significant loss of minerals (eluviation). Hard bedrock, which is not soil, uses the letter R (NRCS, 2006).

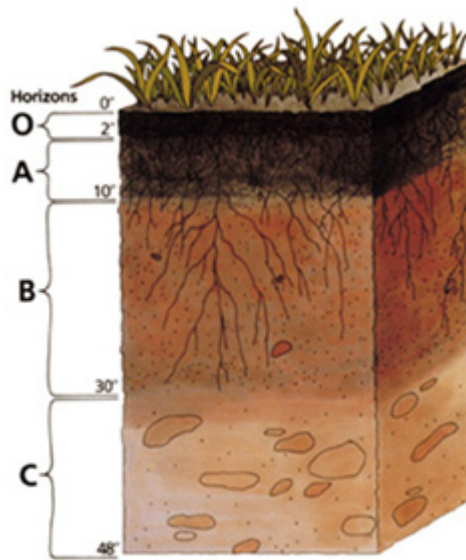


Figure 4.1: A Typical Soil Profile (United States Department of Agriculture National Resources Conservation Service [NRCS], 2006).

In the field, soil color is evaluated by the hue and chroma content. The color is derived from the iron and manganese present, and depends on the soil's current reduced (redox) state. The term *redox* comes from the two concepts of reduction and oxidation. *Oxidation* describes the loss of an electron by a molecule, atom or ion, whereas *reduction* describes the gain of an electron by a molecule, atom or ion (Maier et al., 2000). The U.S. Soil Conservation Service offers the following physical soil traits (Environmental Sciences and Resources [ESR], 2006):

- Mineral Matter (50 percent of the soil composition)
 - Derived from parent rocks;
 - Solid framework of the soil;
 - Contains inorganic material;
 - Can appear mottled or gleyed:
 - Mottled soils have a matrix of one color with blotches and flecks of other colors, and occur where the water table fluctuates and soils may only be saturated for part of the year. In mottled hydric soils, the matrix color is usually grayish

(chroma of 2 or less), and the mottles are usually red, brown or yellowish;

- Gleyed soils are usually light grey with a bluish or greenish tint (chroma of 1 or less). They occur under conditions of long term saturation where essentially all the iron and manganese are reduced.
- Organic Matter (5 percent of the soil composition)
 - Contain carbonaceous substances;
 - Made of organic compounds produced by current/past metabolism in soil;
 - Include remains of living organisms;
 - Types of organic soils:
 - Sapristis (muck) - comprised of 2/3 decomposed material and 1/3 plant fibers;
 - Fibrists (peat) - comprised of 1/3 decomposed material and 2/3 plant fibers
 - Hemists (muck/peat) conditions between sapristis and fibrists
- Air (20 percent of the soil composition); and
- Water (25 percent of the soil composition).

Doran and Safely (1997) consider soil health essential to sustaining biological productivity, promoting the quality of air and water environments, and maintaining plant, animal and human health. A healthy soil functions as an environmental filter for removing undesirable solid and gaseous constituents from air and water. The extent to which a soil immobilizes or chemically alters substances that are toxic, thus effectively detoxifying them, reflects the degree of soil health (Doran & Safely, 1997).

Soil Microorganisms as Bioindicators

Whether discussing evaluation techniques or timeframes, land managers, legislators and ecologists disagree on the most effective way to evaluate the functional equivalency of mitigated wetlands. One of the reasons for unsuccessful mitigation is inadequate monitoring. Singular evaluations of plant species, bioaccumulations of contaminants, sedimentation rates, population demographics, habitat structure, acreage or hydrology do not address the complexity of the whole wetland ecosystem. Rather these factors segregate the wetland and address mitigation in a piecemeal fashion.

Traditionally, wetland communities were analyzed by indices or other analytical metrics. Examples of such techniques include: (1) Similarity (Comparative) Indices; (2) Cluster Analysis and Ordination; (3) Food Web Analysis; (4) Tolerance Indices; (5) Guild Analysis; and (6) Indices of Biotic Integrity (Adamus & Brandt, 2004).

Similarity Indices are metrics that reflect the number of species of functional groups in common between multiple wetlands or time periods. The metrics may be weighted by relative abundance, biomass, taxonomic dissimilarity, or caloric content of the component species. These techniques include Jaccard coefficient, Bray-Curtis coefficient, rank coefficients, overlap indices, and the community degradation index (Ramm, 1988; Adamus & Brandt, 2004). Cluster Analysis and Ordination are procedures that detect statistical patterns and associations in community data, and can be used to hypothesize relationships to a stressor. Such techniques include principal components analysis, reciprocal averaging, de-trended correspondence analysis, TWINSpan, and canonical correlation (Pielou, 1984).

Food Web Analysis measures the length of food chains, number of trophic levels, the ratio of the number of trophic species to trophic links, and similar measures (see Turner, 1988). As yet, this type of analysis has not been tested on stressed wetland systems. Tolerance indices are metrics that reflect the proportionate comparisons of tolerant versus intolerant taxa. "Tolerance," in this case, means tolerance to organic pollution. Examples of tolerance indices include saprobic indices, macroinvertebrate

EPT (Ephemeroptera, Plecoptera and Trichoptera) indices, and the Hilsenhoff index (Micacchion, 2003; Adamus & Brandt, 2004).

The procedures for Guild Analysis involve assigning individual species functional groups (species assemblages) based on the similar facets of their:

- Life history;
- Habitat preference;
- Trophic level, assumed niche breadth;
- Size, biomass, caloric content;
- Toxicological sensitivity;
- Behavioral characteristics;
- Phenological characteristics;
- Sensitivity to human presence;
- Status as an exotic or indigenous species;
- Resident vs. migrant status; and
- Harvested vs. protected status (Adamus & Brandt, 2004).

Finally, indices of biotic integrity are composites of weighted metrics, which describe richness, pollution tolerance, trophic levels, abundance, hybridization, and deformities, and are often used in stream fish studies (Karr, 1981).

Decisions concerning selection of which resources, uses, or functions to protect (or enhance) are inevitably complex, since the criteria for protecting one resource may be contradictory to protecting another (Adamus & Brandt, 2004). Section 404 of the Clean Water Act has outlined a generalized list of wetland functions or uses that might be the focus of protection or restoration (33 CFR 320 (b) (2)), such as:

- Food chain production;
- General habitat;
- Research, education, and refuges;
- Hydrologic modification;
- Sediment modification;
- Wave buffering and erosion control;

- Food storage;
- Ground water recharge or discharge;
- Water purification; and
- Uniqueness/scarcity.

If these parameters have been identified, why look elsewhere for guidance? The answer lies in the continued unsuccessful mitigation projects. Although it may appear from the quantity of literature citations in this review that inland wetlands in some regions have been extensively studied, in reality, relatively little is known about wetland biological response to anthropogenic stressors. Anthropogenic and/or natural stressors are often cumulative and interactive, thus complicating the development of individual indicators. Compared to the monitoring of streams and lakes, sampling wetlands on a recurrent or comparative regional basis has been limited, partly due to lack of government sponsorship of wetland biomonitoring programs (Adamus and Brandt, 2004). Furthermore, the response of a wetland community to anthropogenic stress depends not only on the taxa present and the severity of the stressor, but also on the geomorphic, physical, and chemical environment of the wetlands (Adamus, Clairain, Smith & Young, 1987; Adamus et al., 1991).

In August 1988, EPA's Wetlands Research Program sponsored a workshop in Easton, Maryland, to identify organisms and metrics that might be useful in indicating wetland ecological health. Findings were summarized in the EPA report, "Wetlands and Water Quality: EPA's Research and Monitoring Implementation Plan for the Years 1989 - 1994" (Adamus, 1989). Although the report acknowledged that there are numerous situations where traditional indicators of ecosystem or wetland function (e.g., plant species, bioaccumulation of contaminants, sedimentation rates, population demographics, and habitat structure) are cost-effective and can successfully reflect the ecological condition and sustainability of the wetland, the EPA report requested the synthesis of existing regional literature in ways that would allow candidate bioindicators to be identified and available data to be numerically compiled (Adamus, 1989). One such bioindicator can be the microbial content of soil.

In a recent survey of 156 created wetlands on phosphate-mined lands, 83 were created on graded overburden, while 38 sites were created on a mixture of overburden and sand tailings. Wetlands created on sand tailings alone constituted 8 sites, while 4 of the created sites used sand/clay as the substrate (Ervin, Doherty, Brown & Best, 1997). Construction information on the remaining wetlands in the Ervin et al. (1997) survey was not available. Management of these mitigated sites included the monitoring of vegetation (53 sites), water quality (20 sites), macroinvertebrates (15 sites), wildlife (12 sites), and hydrology (10 sites). Of these 156 sites, only four sites had some soils monitoring and the monitoring protocols were not discussed in depth (Ervin et al., 1997). None of these sites had microbial community sampling or analysis. More recently, Kelly and Tate (1998) studied the size, activity, and structure of microbial communities from remediated and natural soils in the vicinity of a zinc smelter. Results of their work suggested that the microbial community may be a useful indicator of changes in soil quality.

Based on a synopsis of the bioindicator literature (funded by the EPA), it was determined that indicators of ecosystem health should be:

- Socially relevant and easily understood as an indicator of ecological integrity and/or health;
- Capable of evaluating the effectiveness of regulations, control, or management strategies;
- Able to give a maximum amount of information for a minimum cost, and thus be cost-effective and fiscally attractive;
- Capable of being generated from accessible data sources; and
- Capable of providing a warning in time to avoid widespread or irreversible damage of the resource (Adamus & Brandt, 1990).

In order to incorporate these criteria and offer the greatest inference ability, a truly innovative approach is needed to evaluate the success of the management of the wetland systems, and assess the overall mitigation program. Techniques are needed that can both quantitatively address the permit requirements, and provide evidence of how, and to what degree, the resource is changing. Thus, I offer the following case studies to substantiate

the use of soil health and soil microbial communities as indicators of thriving wetlands mitigation projects.

Soil Microorganisms Types and Microbial Community Analysis Techniques

Types of Soil Microorganisms

Microorganisms (Tables 4.2 and 4.3) contribute to proper soil health as they are involved in many important functions such as soil formation, toxin removal, and the elemental cycles of carbon, nitrogen, phosphorous, and sulfur (Borneman et al., 1996). *Autotrophs* (e.g., plants, algae, photosynthetic bacteria, lithotrophs, and methanogens) use carbon dioxide (CO₂) as a sole source of carbon for growth, and reduce the molecule to organic cell material (CH₂O). *Heterotrophs* require organic carbon for growth, and ultimately convert CH₂O back to CO₂ (Maier et al., 2000). Additional essential components of the soil microbial community are symbiotic arbuscular mycorrhiza fungi, which colonize the roots of the majority of terrestrial plants, and rhizospheric bacteria promoting plant growth, which inhabit the plant rhizosphere, the root surface and intercellular root spaces (Tate & Klein, 1985).

Table 4.2: Examples of Important Autotrophic Soil Bacteria (Maier et al., 2000).

Organism	Characteristics	Function
<i>Nitrosomonas</i>	Gram negative, aerobic	Converts ammonia (NH_4^+) \rightarrow nitrite (NO_2^-); 1 st step of nitrification
<i>Nitrobacter</i>	Gram negative, aerobic	Converts $\text{NO}_2^- \rightarrow$ nitrate (NO_3^-); 2 nd step of nitrification
<i>Thiobacillus</i>	Gram negative, aerobic	Oxidizes sulfur (S) \rightarrow sulfate (SO_4^{2-}); sulfur oxidation
<i>Thiobacillus denitrificans</i>	Gram negative, facultative anaerobe	Oxidizes S \rightarrow SO_4^{2-} (functions as a denitrifier)
<i>Thiobacillus ferrooxidans</i>	Gram negative, aerobic	Oxidizes $\text{Fe}^{2+} \rightarrow \text{Fe}^{3+}$

Table 4.3: Examples of Important Heterotrophic Soil Bacteria (Maier et al. 2000).

Organism	Characteristics	Function
Actinomycetes (e.g., <i>Streptomyces</i>)	Gram positive, aerobic, filamentous	Produce geosmins “earthly odor,” and antibiotics
<i>Bacillus</i>	Gram positive, aerobic, spore former	Carbon cycling, production of insecticide and antibiotics
<i>Clostridium</i>	Gram positive, anaerobic, spore former	Carbon cycling (fermentation), toxin production
Methanotrophs (e.g., <i>Methylobacter</i>)	Aerobic	Methane oxidizers that can co-metabolize trichloroethene (TCE) using methane monooxygenase
<i>Alcaligenes eutrophus</i>	Gram negative, aerobic	2,4-D degradation via plasmid pJP4
<i>Rhizobium</i>	Gram negative, aerobic	Fixes nitrogen symbiotically with legumes
<i>Frankia</i>	Gram negative, aerobic	Fixes nitrogen symbiotically with non-legumes
<i>Agrobacterium</i>	Gram negative, aerobic	Important plant pathogen, causes crown gall disease

Microorganisms are essential to ecosystem development and maintenance. Microbes affect the physical properties of the soil (the water holding capacity, infiltration rate, erodibility, and susceptibility to compaction); the soil structure (the production of extra-cellular polysaccharides and other cellular debris help maintain soil structure, as these material function as cementing agents that stabilize soil aggregates); and initial soil development (Nielsen & Winding, 2002).

After a site has been mined, the imported substrate (necessary for initial reclamation) may at first be devoid of organic carbon and nitrogen. The development of biota and soils on mitigation sites require:

1. Carbon-fixation from carbon dioxide (CO_2) by photosynthesis;
2. Nitrogen-fixation by symbiotic or non-symbiotic micro-organisms, or slow accumulation by the retention of small amounts of fixed N deposited from the atmosphere;
3. Phosphorus, which generally is derived from the weathering of the substrate, and therefore is generally present in newly exposed material; and
4. Sulfur weathered from the substrate or accumulated from the atmosphere (Bolin et al., 1981).

Microorganisms participate in the development and maintenance of the biogeochemical cycles of carbon (C), nitrogen (N), phosphorous (P) and sulfur (S). In soil habitats, microorganisms “fix” carbon dioxide (CO_2) from the atmosphere in the synthesis of carbohydrates, which constitute the carbon and energy used by the majority of organisms that do not photosynthesize. Biodegradation is the reverse process - the decomposition of organic material (CH_2O) back to CO_2 , water (H_2O) and hydrogen (H_2). Fungi and prokaryotes (e.g. actinomycetes, clostridia, bacilli, arthrobacters and pseudomonads) play a significant role in biodegradation. Decomposition involves the initial degradation of biopolymers (cellulose, lignin, proteins, polysaccharides) by extracellular enzymes, followed by oxidation (fermentation or respiration) of the monomeric subunits (Maier et al., 2000). The end products include carbon dioxide (CO_2), water (H_2O), hydrogen (H_2), ammonia (NH_3) and sulfide (H_2S). These products

are then used by lithotrophs and autotrophs, and all of these processes and products contribute to the carbon cycle (Figure 4.2).

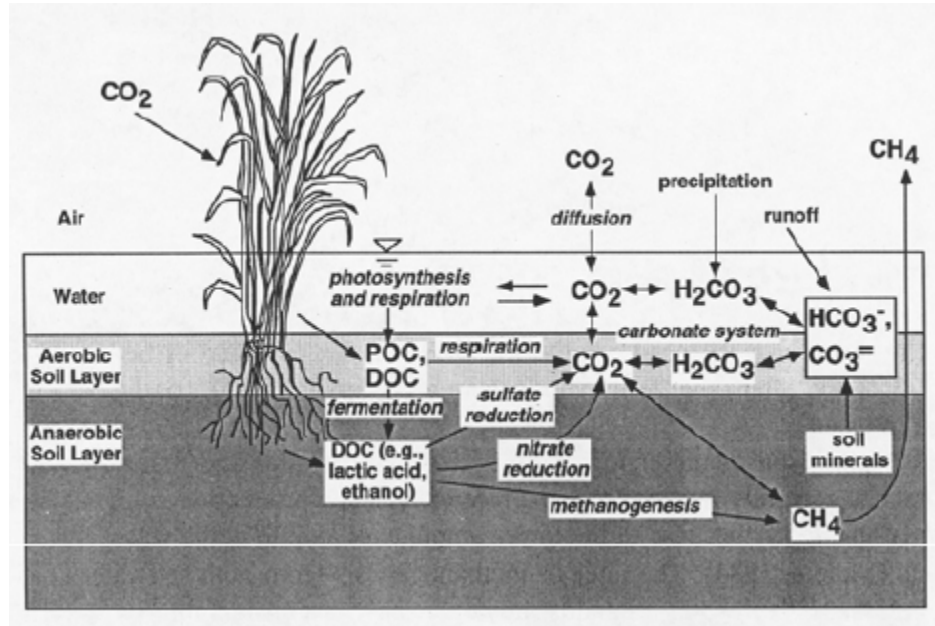


Figure 4.2: The Carbon Cycle (Maser, 2006)

In the nitrogen cycle (Figure 4.3), ammonia (NH_4^+) is oxidized to nitrites (NO_2^-) by nitrosifying bacteria (*Nitrosomonas* and *Nitrosococcus*). Nitrites are then converted to nitrates (NO_3^-) by nitrifying bacteria (*Nitrobacter*) in a process known as nitrification. In denitrification, microorganisms reduce nitrates and nitrites to nitrogen-containing gases (Maier et al., 2000). Nitrogen fixation is the process by which nitrogen (N_2) in the atmosphere is converted into nitrogen compounds useful for other chemical processes (such as, notably, ammonia, nitrate and nitrogen dioxide). Nitrogen fixation is performed naturally by a number of different prokaryotes, including bacteria, actinobacteria, and certain types of anaerobic bacteria. Microorganisms that fix nitrogen are called diazotrophs (Maier et al., 2000).

Nitrogen-fixers add organic matter with a relatively low carbon to nitrogen ratios to the site; the organic matter accumulates and increases the water- and cation-holding capacity of the soil, and the level of available nitrogen in the site rises. The rates of

nitrogen fixation are eventually reduced as the availability of nitrogen increases in the soil, or when phosphate or sulfur become limiting to the nitrogen-fixers. Once the availability of nitrogen in the soil is high, other plants without nitrogen fixing capabilities can grow (Bolin et al., 1981).

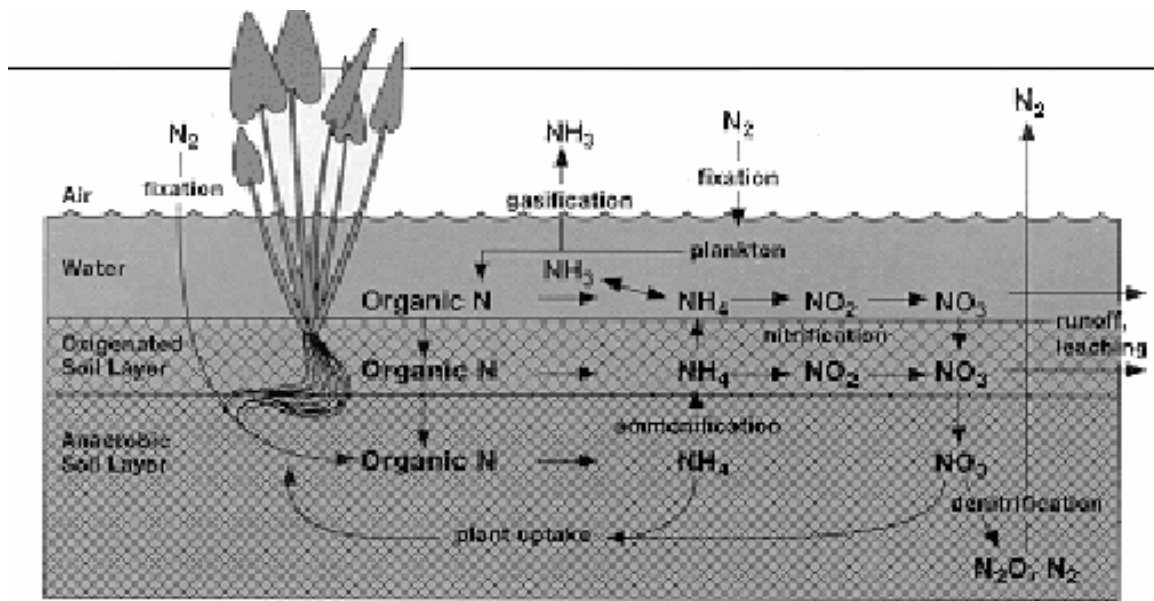


Figure 4.3: The Nitrogen Cycle (Maser, 2006)

Like nitrogen and carbon, microbes can transform sulfur from its most oxidized form (sulfate or SO_4) to its most reduced state (sulfide or H_2S). In the sulfur cycle (Figure 4.4), anoxygenic photosynthetic purple and green sulfur bacteria oxidize H_2S as a source of electrons for cyclic photophosphorylation. Since SO_4 and sulfur may be used as electron acceptors for respiration, sulfate reducing bacteria produce H_2S during a process of anaerobic respiration analogous to denitrification. The use of SO_4 as an electron acceptor is an obligatory process that takes place only in anaerobic environments. The process results in the distinctive odor of H_2S in anaerobic bogs, soils and sediments where it occurs. Sulfur is assimilated by bacteria and plants as SO_4 for use and reduction to sulfide. Animals and bacteria can remove the sulfide group from proteins as a source of S during decomposition. These processes complete the sulfur cycle (Maier et al., 2000).

Plants, algae and photosynthetic bacteria can absorb the phosphate (PO_4) dissolved in water, and incorporate the mineral into various organic forms, including such molecules as nucleic acids and the phospholipids of cell membranes. When the plants are consumed by animals, the organic phosphate in the plant becomes the organic phosphate in the animal and in the bacteria that live with the animal. Animal waste returns inorganic PO_4 to the environment and also organic phosphate in the form of microbial cells (Figure 4.4). Dead plants and animals, as well as animal waste, are decomposed by microbes in the soil. The phosphate eventually is mineralized to the soluble PO_4 form in water and soil, to be taken up again by photosynthetic organisms (Maier et al., 2000).

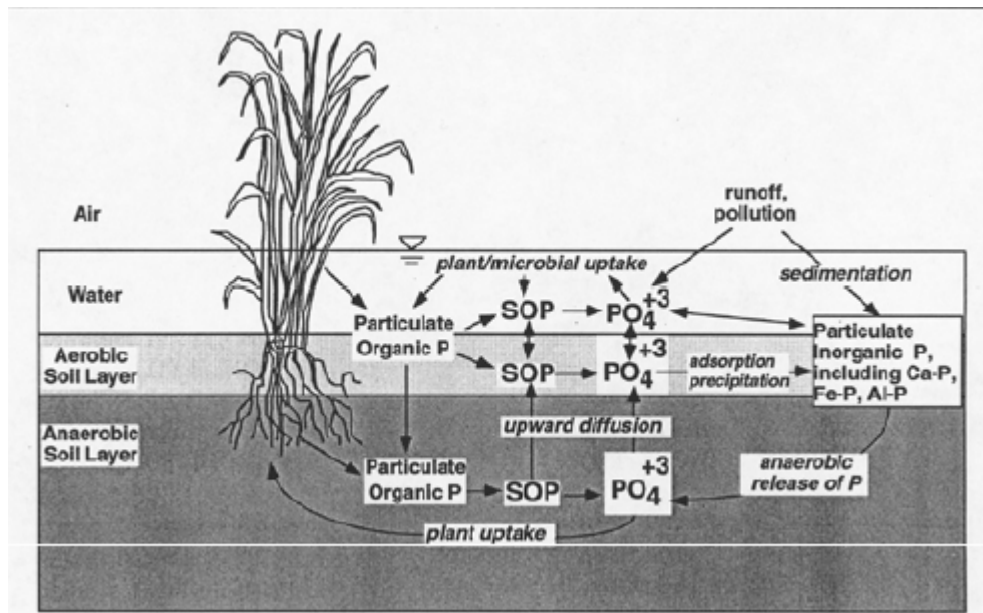


Figure 4.4: Sulfur and Phosphorus Cycling (Maser, 2006)

The soil-plant-microorganism cycle is strengthened by rapidly growing pioneer plants that invade disturbed sites and take advantage of the available light, water, nitrogen, and phosphate to grow. The site then returns towards its predisturbance state, drawing carbon and some nitrogen and sulfur from the atmosphere, and phosphate, nitrogen and sulfur from the soil (Bolin et al., 1981).

Autotrophic and heterotrophic microorganisms dominate global biogeochemistry, accounting for roughly half of global photosynthesis and most of the organic matter

decomposition, nitrification, denitrification, and methane production (Maier et al., 2000). Several aspects of the biogeochemical cycles can be monitored for soil and microbial development.

Though microorganisms are abundant in soils, identifying microorganisms from soil samples is difficult because most bacteria in natural environments cannot be cultured with current techniques (for reviews, see Staley & Konopka, 1985; Roszak & Colwell, 1987a, 1987b; Amann, Ludwig, & Schleifer, 1995). Estimates reveal that only approximately 1 – 10 percent of the microorganisms in soil have been identified (Amann et al., 1995). Table 4.4 lists some of the identified cultural bacteria and their functions in soil.

Table 4.4: Dominant Cultural Soil Bacteria (Maier et al., 2000).

Organism	Characteristics	Function
<i>Arthrobacter</i>	Heterotrophic, aerobic, gram variable. Up to 40% of culturable soil bacteria.	Nutrient cycling and biodegradation.
<i>Streptomyces</i>	Gram positive, heterotrophic, aerobic actinomycete. 5-20% of culturable soil bacteria.	Nutrient cycling and biodegradation. Antibiotic production, e.g., <i>Streptomyces scabies</i> .
<i>Pseudomonas</i>	Gram negative heterotroph. Aerobic or facultatively anaerobic. Posses wide array of enzyme systems. 10-20% of culturable soil bacteria.	Nutrient cycling and biodegradation, including recalcitrant organics. Biocontrol agent.
<i>Bacillus</i>	Gram positive aerobic heterotroph. Produce endospores. 2-10% of culturable soil bacteria.	Nutrient cycling and biodegradation. Biocontrol agent, e.g., <i>Bacillus thuringiensis</i>

The biodiversity of soil microbial communities can be measured with mathematical indices that emphasize richness (the number of species), equitability (the evenness of allocation of individuals among the various species), or combinations of these two. Microbial biodiversity includes the number and distribution of species, as well as the functional diversity and redundancy. Functional diversity is the number of distinct processes or functions that are carried out by a community, whereas functional

redundancy is a measure of the number of different species within the various functional groups or guilds (Gaston, 1996; Bei et al., 2000). Bacteria (and fungi) can be evaluated by viable count methods involving plating on nutrient-rich agar. Direct counting involves counting individual organisms with the naked eye or with a microscope. Plate counting calculates the number of bacterial or fungal colonies that grow from a soil sample (Bei et al., 2000). Phenotypic methods of microbial identification include microbial morphology, Gram staining, enzyme activities, and the utilization of several substrates as sole carbon and energy sources.

Diversity can also be determined by measurements of activity levels, such as the amount of by-products (e.g., CO₂) generated in the soil, or the disappearance of substances, such as plant residue or methane used by portions of the community or by specific groups of organisms. These measurements reflect the total “work” the community can do. Total biological activity is the sum of activities of all organisms, though only a portion is active at a particular time (Maier et al., 2000). Additional activity level measurements include:

- Determining the respiration rates by measuring CO₂ production. This method is limited, as it does not distinguish which organisms (plants, pathogens, or other soil organisms) are generating the CO₂.
- Determining the nitrification rates by measuring the activity of those species involved in the conversion of ammonium to nitrate.
- Determining the decomposition rates by measuring the speed of disappearance of organic residue or standardized cotton strips.

In addition, evaluating the cellular constituents of the microorganisms can also suggest diversity. The total biomass of soil organisms or specific characteristics of the community can be inferred by measuring:

- The amount of nutrients (carbon, nitrogen, or phosphorus) in the cells, which can then be used to estimate the total biomass of organisms. Chloroform fumigation is a common method used to estimate the amount of carbon or nitrogen in all soil organisms;

- The enzymes in the cells or those attached to the soil. Assays can be used to estimate potential activity or to characterize the biological community;
- The “fingerprint” of the community, using phospholipids and other lipids to quantify the biomass of groups such as fungi or actinomycetes;
- The various attributes of the individual nucleic acids (DNA and RNA) (Stephen et al., 1999a, 1999b; Maier et al., 2000, Wright & Reddy, 2001).

All of these techniques (mathematical indices of richness and equitability, non-molecular and molecular measurements of activity levels, and cellular constituents) can be used to as monitoring criteria for mitigation projects, and investigation into these procedures will be examined.

Techniques for Measuring Biodiversity and Community Structure

Indices of Richness and Non-molecular Analytical Techniques

The diversity of a community is expressed as the species richness and the relative contribution each species makes to the total number of organisms present. The diversity of a microbial community is often described by the Shannon-Weaver index (H') (Shannon & Waever, 1949). The number of species has traditionally been determined by taxonomic classification studies, but as these are sub-optimal for microorganisms, molecular and biochemical techniques of estimating abundance and number of each species must be applied. The benefit of a high genetic diversity is currently under debate because it is not always correlated to functional diversity (Nielsen & Winding, 2002).

Ibekwe, Lyon, Leddy and Jacobson-Meyers (2006) studied microbial community composition and water quality changes within free water, surface constructed wetland cells that contained various plant densities and composition. The authors used the Shannon-Weaver index (H') to determine microbial diversity and results indicated that diversity was higher in the wetland cells with 50 percent plant density, rather than those with 100 percent plant diversity. This study provided evidence that wetlands with 50 percent plant cover can promote the growth of diverse microbial communities that facilitate decomposition of chemical pollutants in surface water, and improve water

quality. Thus showing that land managers should focus more on soil development, rather than planting techniques.

Possible horizontal variations of communities in a natural oligotrophic fen, were analyzed by Galand, Fritze, and Yrjälä (2003) using methanogens from two well-defined microsites: *Eriophorum* lawn and Hummock. The community structures were studied at two different layers of the fen, showing, respectively, high and low methane production. Phylogenetic analyses revealed six different clusters of sequences grouping with only two known orders of methanogens. Upper layers of Hummock were dominated by sequences clustered with members of *Methanomicrobiales*, and the sequences that dominated the upper part of the *Eriophorum* lawn were members of the order *Methanosarcinales*. Novel methanogenic sequences were found at both sites at both depths. The authors concluded that the vegetation characterizing the microsites influenced the microbial communities in the layers of the fen where methane was produced.

An additional non-molecular technique for measuring microbial diversity is a multi-stage dispersion and differential centrifugation technique suggested by Hopkins MacNaughton and O'Donnell (1991), used to sample non-filamentous microorganisms from soil. The soil aggregates were dispersed and the microorganisms were dissociated from the soil particles. The released microorganisms were then separated by low-speed centrifugation. The biomass of the microorganisms were determined using traditional methods such as direct microscopic cell counts, adenosine triphosphate (ATP), phospholipid, lipopolysaccharide, ergosterol contents and viable counts. When compared to traditional methods (phenotypic methods of microbial identification such as morphology, Gram staining, and enzyme activities), the centrifugation method yielded extracts that were more enriched with microorganisms (Hopkins et al., 1991).

One approach for indexing the functional diversity of microbial communities has been the use of substrate or metabolic fingerprinting (Bei et al., 2000). Microbial communities are typically screened for their ability to utilize selected carbon substrates by using MicroPlates. Bei et al. (2000) developed a culture-independent strategy to examine bacterial functional redundancy and tested its use on soils collected along a vegetation gradient on reclaimed mine spoils in Rondonia, Brazil. The data suggested that

bacterial functional redundancy increased in relation to the re-growth of plant communities, and therefore represented an important aspect of the restoration of soil biological functionality to reclaimed mine spoils. The authors reiterated the importance of bacterial functional redundancy as an indicator of soil quality and ecosystems functioning.

Molecular Techniques

Molecular microbial identification can be conducted in the laboratory (*in vitro*) or in the field (*in situ*). *In vitro* measurements may involve incubation of a soil sample in the laboratory under standardized conditions. Interpretation of *in vitro* measurements in relation to soil health can be difficult, however, because the results depend on the incubation conditions which may not be comparable to field conditions. Examples of *in vitro* measurements are soil respiration, nitrogen-mineralization, nitrification, denitrification, most probable number dilutions, and other growth-based methods (Brock, Smith & Madigan, 1984).

In situ measurements are based either on direct measurements in the field or fixed samples analyzed in the laboratory. These types of measurements are sensitive to spatial and temporal variation, however, and may underestimate the variability in soil health status (Brock et al., 1984). Examples of *in situ* measurements are phospholipid fatty acid (PLFA), organic matter decomposition, thymidine and leucine incorporation, short-term enzyme assays and most molecular methods (Brock et al., 1984).

Phospholipid fatty acid (PLFA) analysis has been used by several authors for microbial analysis. PLFA provides information about soil microorganism biomass, fungal-bacterial ratios, biodiversity and the occurrence of key species (Stephen et al., 1999a, 1999b). Fatty acids are extracted from microorganisms, characterized, and the ratios generated are compared to known ratios of fatty acids present in cell membranes and other microbial structures (Stephen et al, 1999a, 1999b). The introduction of polymerase chain reaction amplification-denaturing gradient gel electrophoresis (PCR-DGGE) to microbial ecology has provided a second valuable molecular fingerprinting technique for studying microbial community structure (Henckel, Friedrich & Conrad,

1999; Heuer, 1997; Muyzer 1998). Denaturing gradient gel electrophoresis separates and distinguishes deoxyribonucleic acid (DNA) fragments by establishing a denaturing gradient that separates DNA into discrete bands that can then be excised and sequenced for identification. DGGE allows large numbers of samples to be analyzed simultaneously, thus this technique is ideally suited for monitoring the dynamics of microbial communities influenced by environmental changes (Kelley & Hentzen, 2003; Henckel et al., 1999).

A study conducted at a wetland site in Southern Illinois currently undergoing restoration prior to leveeing and agricultural use used PLFA and PCR-DGGE to identify the dominant bacterial populations of the Spunky Bottoms aquatic wetland system (Kelley & Hentzen, 2003). Water samples were collected and analyzed by gas chromatography/mass spectrometry to determine PLFA profiles for each water sample. The PCR-DGGE DNA fragments were excised, sequenced, and predominant microbial species were characterized based upon sequence homology to previously identified sequences contained in the National Center for Biotechnology and Ribosomal Data Project databases (Kelley & Hentzen, 2003). Results identified diverse microbial communities, including microorganisms that may substantially contribute to biogeochemical cycling of elements, including nitrogen and phosphorus. The sequence analysis of DNA fragments showed similarity indices of 0.8 to 1.0 to *α*-Proteobacteria, Flavobacterium, Flexibacter-Cytophaga-Bacteroides groups, Gram-positive and Gram-negative bacteria, Cyanobacteria, and Chloroplasts (Kelley & Hentzen, 2003). The authors emphasized that microbial communities (and populations) could be used to provide a basis of comparison for future research.

Bossio, Fleck, Scow and Fujii (2006) used PLFA analysis to assess the microbial community changes as a site previously under agricultural management transforms to a permanently flooded wetland. The PLFA analysis indicated that the active microbial community of the wetland was different from that of the agricultural field.

Databases of PLFA profiles have been generated for a variety of microorganisms to facilitate their identification. Phospholipid fatty acids analysis has been used to examine microbial communities and their changes for a variety of marine, aquatic, and

terrestrial systems (Stephen et al., 1999a, 1999b). Additionally, PCR-DGGE analysis has been used to isolate microbial DNA and thereby identify predominant microbial populations in a variety of systems (Kelley & Hentzen, 2003).

Synthesis of DNA is a prerequisite for bacterial cell division and, as such, an indicator of bacterial growth. The incorporation of H₃- or C₁₄-thymidine into bacterial DNA can be used to determine DNA synthesis, as thymidine is a nucleoside that is used in DNA synthesis. This method has several requirements: (1) DNA synthesis has to be linearly correlated with the cell growth (balanced growth); (2) all bacteria must take up thymidine through the cell membrane; (3) thymidine should not be catabolized; and (4) the radioactive label (H₃) should not exchange with other molecules (e.g. proteins). It has been shown that only 5 – 20 percent of the H₃-thymidine incorporated into total macromolecules is incorporated into DNA (Baath, 1998, Michel & Bloem, 1993). The procedure involves incubating the soil samples with radio-labeled thymidine for a short time followed by filtration to measure the amount of radiolabel in the cells.

Additional approaches to describe the diversity of natural soil or sediment communities have included DNA re-association experiments (Torsvik, Goksoyr & Daae, 1990), 16S ribosomal DNA (rDNA) retrieval and analysis (Borneman et al., 1996), fractionation of total bacterial DNA by G1C content (Holben & Harris, 1995), and cross-hybridization of bacterial DNA from two communities (Ritz & Griffiths, 1994). All of these techniques have pointed to highly complex assemblages of bacterial populations. For example, Torsvik et al. (1990) estimated that 10³ to 10⁴ different genomic equivalents were present in 1 gram of soil, while Borneman et al. (1996) recovered 124 previously undescribed 16S rRNA gene sequences from an agricultural soil. These studies of genetic diversity have provided valuable and qualitative insights into microbial community composition.

Recombinant DNA and molecular phylogenetic methods have recently provided means for identifying the types of organisms that occur in microbial communities without the need for cultivation (see Hugenholtz & Pace 1996 for review). Hugenholtz and Pace compiled a table that summarizes the environmental distribution of sequences by habitat type, derived from most of the available 16S rRNA-based clonal analyses (86 studies

contributing nearly 3, 000 sequences). An expanded version of this table, which details division-level representation in the individual studies, is available at <http://crab2.berkeley.edu/pacelab/176.htm>.

The amount of ribonucleic acid (RNA) in individual cells can represent protein synthesis and, thus, microbial activity. The number of active cells can be detected by fluorescent *in situ* hybridization (FISH) (Amann et al., 1995). By this method, individual cells carrying high concentrations of rRNA are quantified by fluorescence microscopy and/or reverse transcriptase polymerase chain reaction (RT-PCR), where rRNA extracted from soil is detected by creating a DNA copy and separating by gel electrophoresis (Duineveld et al., 2001).

The polymerase chain reaction (PCR) technique geometrically increases concentrations of specific DNA fragments to provide enough genetic material for identification by techniques such as gel electrophoresis. PCR allows a small amount of the DNA molecule to be amplified many times, in an exponential manner, and is used to amplify a short, well-defined parts of the DNA strand. PCR requires several basic components:

- The DNA template;
- Two primers, which determine the beginning and end of the region to be amplified;
- Taq polymerase, which copies the region to be amplified;
- Deoxynucleotides-triphosphate from which the DNA-polymerase builds the new DNA; and
- Buffers, which provide a suitable chemical environment for the DNA-polymerase

The PCR technique was used by Henckel et al. (1999) to investigate the structure of the methanotrophic community in rice field soil. The soil microbial community was monitored by performing DGGE during the oxidation process with different PCR primer sets based on the 16S rRNA gene and on functional genes. A universal small-subunit (SSU) rDNA primer set and 16S rDNA primer sets specifically targeting type I methylotrophs (members of the γ subdivision of the class *Proteobacteria* [γ -

Proteobacteria]) and type II methylobionts (members of the α -*Proteobacteria*) were used. Functional PCR primers targeted the genes for particulate methane monooxygenase (*pmoA*) and methanol dehydrogenase (*mxoF*), which code for key enzymes in the catabolism of all methanotrophs. The yield of PCR products amplified from DNA in soil that oxidized CH₄ was the same as the yield of PCR products amplified from control soil when the universal SSU rDNA primer set was used, but was significantly greater when primer sets specific for methanotrophs were used.

The DGGE patterns and the sequences of major DGGE bands obtained with the universal SSU rDNA primer set showed that the community structure was dominated by non-methanotrophic populations related to the genera *Flavobacterium* and *Bacillus* and was not influenced by CH₄. The structure of the methylobiont community, as determined with the specific primer sets, was less complex; this community consisted of both types I and II methanotrophs related to the genera *Methylobacter*, *Methylococcus*, and *Methylocystis*. DGGE profiles of PCR products amplified with functional gene primer sets that targeted the *mxoF* and *pmoA* genes revealed that there were pronounced community shifts when CH₄ oxidation began. High CH₄ concentrations stimulated both types I and II methanotrophs in rice field soil with nonsaturated water content, as determined with both ribosomal and functional gene markers (Henckel et al., 1999).

Zhou et al. (2002) identified possible determinants for microbial community structure in soil by characterizing microbial communities from 29 soil samples using a SSU rRNA-based cloning approach. The authors concluded that while microbial communities in low- carbon, saturated, subsurface soils showed dominance, microbial communities in low-carbon surface soils showed remarkably uniform distribution. Analysis using the SSU technique has been used for agricultural sites (McGarvey, Miller, Sanchez & Stanker, 2004), rice field soils (Fey & Conrad, 2000), and nitrifying bacteria diversity in wastewater (Princic et al., 1998). Overall, the use of nucleic acid-based methods for soil microbial community identification has revealed high prokaryote diversity (Bintrim et al., 1997; Borneman & Triplett, 1997; Felske, Wolterink, Lis & Akkermans, 1998; Dunbar et al., 1999). In Europe, a database (<http://rrna.uia.ac.be/ssu>) has compiled all complete or nearly complete SSU RNA sequences. Additional

information (i.e., literature reference, taxonomy, secondary structure models and nucleotide variability maps) is also available at this website.

Microbial diversity can also be measured using a process known as BIOLOG. This technique characterizes and identifies microbial isolates by examining their carbon source utilization profiles (Nielsen & Winding, 2002). BIOLOG tests the ability or inability of microorganisms to metabolize (i.e. degrade via oxidation) a large and diverse set of chemicals substrates. Identification of the microorganism is generated by a computerized matching of the metabolic properties of the isolate to a large database of patterns (Nielsen & Winding, 2002).

Although molecular techniques are powerful, they show some limitations in the case of diversity studies. One shortcoming of PCR is that it involves sampling a heterogeneous matrix of soils, a point which frequently receives little consideration in study design (Amann et al., 1995). Grundman and Gourbiere (1999) designed a soil sampling procedure that considered bacterial spatial distribution within the soil matrix. The results of their study suggested that the micro-fragmentation of soil (fractions of soil, sampling of specific habitats, dissection of minute pieces of soil) prior to culturing can help minimize cell interactions during sample processing for isolation. Tilman (1994) supported this finding and added that soil fragmentation decreases interspecies competition, allowing growth of minor populations in enrichment cultures.

Molecular tests do not have to be time consuming or costly. When samples are collected for multiple tests, the soil samples can be divided into three parts - one portion of the sample is used for determining moisture content, one for organic content, one for microbial community analysis based on extractable lipid profiles or microbial DNA (Ogram, 1987).

The moisture content of soil can be determined by weighing out a wet soil sample then drying it overnight in an oven at 105°C. Lipids can be extracted from each sample using a Bligh-Dyer extraction buffer (Ogram, 1987). In addition to using microbial lipids as a biochemical marker for community structure, microbial genomic DNA can be extracted from soil samples then subjected to PCR. By designing different PCR primers, the whole community (or a subset of the community) can be analyzed.

Measurements of Microbial Activity – Analysis of the Biogeochemical Cycles

Changes (natural or anthropogenic) in soil chemistry and structure can disrupt the requisite biogeochemical cycles and prolong wetland development. For example, the impairment of soil carbon cycling can trap the element in wetland soils. Because nitrifying bacteria (e.g., *Nitrosomonas* and *Nitrobacter*) are sensitive to acidity and require aerobic conditions, waterlogged soils can become anoxic and may not support nitrification. Nodulation of legumes can be impeded by some kinds of pollutant molecules and thus the symbiotic nitrogen fixation phenomenon can be disrupted. Phosphorus cycling could conceivably be impeded by anything that interferes with mycorrhizal fungi (Sims, 1990).

Instead of testing for the pollutant, the microbial facets of the soil can be evaluated. Nitrogen-fixing and ammonia-oxidizing bacteria were studied by Yeager et al. (2005) to examine the effects of forest fire on these important groups of nitrogen-cycling bacteria. Analysis of *nifH* and *amoA* PCR amplicons was performed on DNA samples from unburned, moderately burned, and severely burned soils of a mixed conifer forest following the Cerro Grande Fire (an intensive crown fire) near Los Alamos, New Mexico. The results indicated that a decreased microbial biomass and shift in nitrogen-fixing and ammonia-oxidizing communities was still evident in fire-impacted soils collected 14 months after the fire. The site was not showing signs of deficiency, but these authors showed that the bacterial components of the system had been affected and remained compromised months later. The authors also suggested long-term monitoring for successful rehabilitation.

Key biogeochemical processes such as organic matter decomposition, pollutant degradation, and non-symbiotic nitrogen fixation occur at accelerated rates in the rhizosphere and greatly influence ecosystem functions (Anderson, Guthrie & Walton, 1993; Daane et al., 2001). Recent work has indicated that the stimulation of microbial activity in the rhizosphere of plants can also stimulate biodegradation of various toxic organic compounds (Anderson et al., 1993). The rhizosphere soil is the zone of soil under the direct influence of plant roots, which usually extends a few millimeters from the root surface and is a dynamic environment for microorganisms (Daane et al., 2001).

The rhizosphere microbial community is comprised of microorganisms with different types of metabolic and adaptive responses for various environmental conditions. The production of mucilaginous material and the exudation of a variety of soluble organic compounds by the plant root play an important part in root colonization and maintenance of microbial growth in the rhizosphere (Daane et al., 2001).

Bacteria of the genera *Rhizobium* are abundant in soil and form symbiotic associations with legume roots. The bacteria reside in nodules where they fix atmospheric nitrogen and provide the plant with elemental nitrogen for growth. In return, the plant provides the bacteria with organic substrates for growth (Hungria, Chueire, Coca & Megias, 2001). Numbers of *Rhizobium* has previously been proposed as an indicator of soil health based on the organisms' sensitivity to pesticides and heavy metals (Hungria et al., 2001). The frequency and diversity of *Rhizobium* in soil can be determined by a simple pot test, where a diverse set of legume seeds are sowed in the test soil and the number of nodules formed are determined after a specific growth period (Nielsen & Winding, 2002). Alternatively, the bacteria may be quantified by direct isolation from soil using selective growth media together with morphological and physiological characterizations (Hungria et al., 2001).

A study was conducted to isolate the polycyclic aromatic hydrocarbon (PAH)-degrading bacteria from the rhizosphere of the salt marsh grasses *S. alterniflora*, *J. gerardi*, *D. spicata* and *S. airoides* by enrichment using naphthalene, phenanthrene, or biphenyl as the sole source of carbon and energy (Daane et al., 2001). The authors found both gram positive (predominantly nocardioform) and gram-negative (predominantly pseudomonad) bacteria, and a pasteurization technique prior to enrichment, resulted in the isolation of spore-forming bacteria (exclusively *Paenibacillus* sp.). These results demonstrated the wide variation between the PAH-degrading isolates, indicating that the rhizosphere of *S. alterniflora* contained a diverse population of PAH-degrading bacteria (Daane et al., 2001).

Polymerase chain reaction (PCR) primers that amplify parts of the *narG* and *nirK* genes can be used to create a community profile that focuses on the denitrification process (Maier et al., 2000). When cloned and sequenced, these PCR products can be

used to identify the bacteria involved. Afterwards, the bacterial growth rate (number of cells formed per unit time) can be calculated.

Soil respiration can be determined by either carbon dioxide (CO₂) production or oxygen (O₂) consumption. Measurements of CO₂ concentration are more sensitive, because the atmospheric concentration of CO₂ is only 0.033 percent versus the 20 percent of O₂. Determination of CO₂ production from soil samples can be made in the laboratory by simple and inexpensive techniques based on alkaline CO₂ traps followed by chemical titration, or by more sophisticated automated instruments based on electrical conductivity, gas chromatography or infrared spectroscopy (Alef, 1995). Combined with automated sampling from test soil samples, automated instruments make it possible to determine CO₂ production as a function of time for several days. (Zibilske, 1994). Respiration is highly influenced by temperature, soil moisture, nutrient availability, and soil structure. Pre-conditioning and standardization of the soil before measuring respiration is necessary to minimize the effect of these variables. Soil respiration measurements have been used as an indicator of pesticide and heavy metal toxicity (Brookes, 1995).

Wright and Reddy (2001) studied the influence of phosphorous loading on aerobic and anaerobic heterotrophic microbial activities. The authors measured CO₂ and methane (CH₄) production in detritus and soil collected from a Water Conservation Area in the Everglades. This field study involved the measurement of CO₂ production under drained and flooded soil conditions. Schott media bottles with screw-type lids containing 10 grams of drained, moist soil and a sodium hydroxide (NaOH) trap were sealed under an atmosphere of 21 percent O₂ to facilitate aerobic conditions. A flooded (anaerobic) treatment containing 10 grams of wet samples of flooded soil with an atmospheric nitrogen headspace was also included. For substrate induced respiration (SIR) measurements, both drained and flooded treatments were supplemented with glucose at an excess concentration of 25 mg C g soil⁻¹, based on results of previous experiments (Wright & Reddy, 2001). Results revealed that both CO₂ and CH₄ production significantly correlated with soil P parameters and microbial biomass.

In a study by Rooney-Varga, Richard, Robert & Hines (1997), phylogenetic diversity and community composition of sulfate-reducing bacteria in salt marsh sediment and in the rhizosphere of *Spartina alterniflora* were investigated. The authors chose a community of sulfate-reducing bacteria (SRB) inhabiting a salt marsh sediment. Sulfate-reducing activity correlates with plant growth stages, suggesting that plant-SRB interactions in the *S. alterniflora* rhizosphere play an important role in salt marsh biogeochemical cycles (Hines, 1991; Hines, Knollmeyer & Tugel, 1989). While molecular studies of soil-sediment microbial communities have suggested extremely high complexity, with up to 104 species present in a gram of soil (Torsvik et al., 1990), the community structure or quantitative distribution of individual phylotypes remains poorly understood. In this study, the results suggested that while the overall sediment community may be highly diverse, there were a small number of well-adapted species in the sediment habitat that play a significant role in microbial community dynamics (Rooney-Varga et al., 1997)

Wetlands are considered important sites for methane oxidation, as these areas receive a high input of organic material. Methane is produced by methanogenic *Archaea* and consumed by aerobic methane oxidizing bacteria, the methanotrophs (Maier et al., 2000). Methane oxidation is measured by spiking a soil sample with methane and incubating the sample in a closed jar in the laboratory. Loss of methane is subsequently determined by gas chromatography. Methanotrophs can be quantified directly in soil by fluorescent *in situ* hybridization (FISH) (Amann et al. 1995) or standard growth-dependent most-probable-number (MPN) counts. Analyses of methanotrophic communities can also be done with PCR-DGGE using methanotrophs-specific 16S rDNA primers (see Ritchie, Edwards, McDonald & Murrell, 1997).

Methanogens possess several potential biomarkers, including coenzyme F420, isoprenoids, and lipid ethers in the cell membranes, as well as coenzyme M (CoM) (Elias, Krumholz, Tanner & Suflita, 1999). The previously developed techniques for assaying CoM include a bioassay and high-performance liquid chromatography (HPLC)-based method which measures fluorescent isoindole derivatives of thiols (Elias et al., 1999). The latter method was not standardized for biomass determination. Both of these

methods are cumbersome and time-consuming, require strictly anaerobic conditions and were not designed for use with sediments. Elias et al. (1999) modified the HPLC-based procedure to quantify CoM within hours of sample collection without a requirement for anoxic conditions. The authors also standardized the technique with pure cultures so that it could be used to quantify methanogen biomass in a variety of environmental matrices. The MPN assays used three-tube assays to a 10^{-8} dilution. The method resulted in an efficient and relatively simple method to estimate methanogenic biomass.

In addition to analyzing aspects of the biogeochemical cycles, Tate and Klein (1985) suggested investigating soil microbial biomass. The authors stress that microbial biomass is fundamental to the transformation and flow of carbon, nitrogen, sulfur and phosphorus. The microbial biomass is the fraction of the soil responsible for the energy and nutrient cycling and the regulation of organic matter transformation (Nielsen & Winding, 2002). Microbes transform organic nitrogenous and sulfuric compounds to forms that can be utilized more readily to plants. Wetland floras are dependant on the microbial oxidation of these compounds by microorganisms for survival. In addition, soil microorganisms also immobilize nutrients that would be otherwise lost to leaching (Tate & Klein, 1985).

Determination of soil microbial biomass by direct methods (microscopy or PLFA analysis) provides results that very closely represent the *in situ* soil conditions. Although the methods are time-consuming, they are currently used for soil monitoring purposes (see Bloem, Bolhuis, Veninga & Wieringa, 1995). Indirect methods are generally cheaper, faster and easier to use than the direct methods. Results obtained by the indirect methods have been documented to be very close to the direct measurements (Carter et al., 1999), thus providing confidence in the utility of indirect methods.

Analytical techniques for microbial biomass determination include the chloroform fumigation incubation method (CFI) and the chloroform fumigation extraction method (CFE) (Carter et al., 1999). In both cases, the chloroform vapor kills the microorganisms in the soil, and subsequently the size of the killed biomass is estimated either by quantification of respired CO_2 over a specified period of incubation (CFI) or by a direct extraction of the soil immediately after the fumigation followed by a quantification of

extractable carbon (CFE). The release of CO² after fumigation is the result of germinating microbial spores utilizing the carbon substrate provided by the killed microbial cells (Carter et al., 1999).

Another common indirect method is substrate induced respiration (SIR). This method measures only the metabolically active portion of the microbial biomass (Carter et al., 1999). Substrate induced respiration measures the initial change in the soil respiration rate as a result of adding an easily decomposable substrate (e.g. glucose). Soil microbial biomass is subsequently calculated using a conversion factor (Carter et al., 1999).

Measurements of Microbial Activity – Soil Organic Matter

Many soil properties impact soil quality, but organic matter deserves special attention. It affects several critical soil functions, can be manipulated by land management practices, and is important in most agricultural settings across the country (McCauley, Jones & Jaconsen, 2003). Because organic matter enhances water and nutrient holding capacity and improves soil structure, managing for soil carbon can enhance productivity and environmental quality, and can reduce the severity and costs of natural phenomena, such as drought, flood, and disease. In addition, increasing soil organic matter levels can reduce atmospheric CO₂ levels that contribute to climate change (McCauley et al., 2003).

Because soil microorganisms require external sources of organic carbon, the activity and sustainability of the microbial community depends on whether this organic matter is replaced after mining. The rate at which stable organic matter accumulates on disturbed systems will be largely determined by the biological characteristics of the mine site (McCauley, Jones & Jaconsen, 2003). The soil organic matter (SOM) content is equal to the net difference between the amount of SOM accumulated and the amount decomposed. Soil organic matter cycling consists of four main processes carried out by soil microorganisms (Figure 4.5):

1. Decomposition of organic residues;
2. Nutrient mineralization;

3. Transfer of organic carbon and nutrients from one SOM pool to another; and
4. Continual release of CO₂ through microbial respiration and chemical oxidation (McCauley, Jones & Jaconsen, 2003).

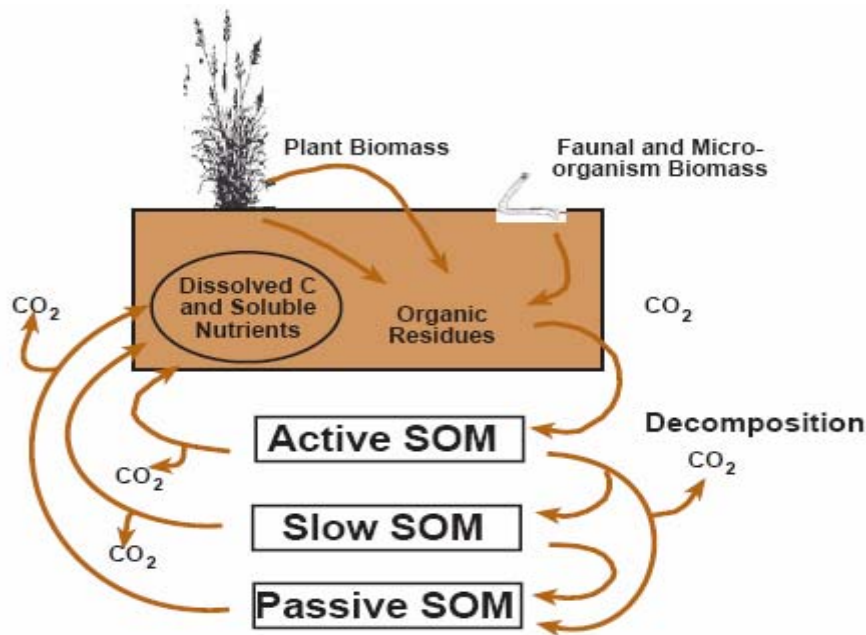


Figure 4.5: Organic Matter Cycle (McCauley et al., 2003).

This figure is modified from Brandy and Weil, 1999. The three main pools of SOM, determined by their time for complete decomposition, are active (1-2 years), slow (15-100 years), and passive (500-5000 years) (Brandy and Weil, 1999).

Soils high in clay and silt (as found at mining sites) are generally higher in SOM content than sandy soils. This is attributed to aeration in finer-textures soils reducing the rate of organic matter oxidation, and the binding of humus (a dark brown, porous, spongy material that provides a carbon and energy source for soil microbes and plants) to clay particles. This process further protects the soil from decomposition (McCauley et al., 2003).

The SOM content is a key indicator of soil quality and is correlated to a number of important soil processes that occur in wetlands such as respiration, denitrification, and phosphorus sorption. To better understand the differences in the SOM content of created, restored, and natural wetlands, 11 created/restored and natural wetland pairs were

sampled in North Carolina (Bruland & Richardson, 2006). Bruland and Richardson hypothesized that the SOM content of paired created/restored and natural wetland would be similar. Results indicated that all of the individual sites had significantly different SOM contents, with created/restored wetlands having significantly lower mean SOM than their paired natural wetland on average. The authors thus concluded that if there is a choice in mitigation options (restoration or creation), wetlands should be restored rather than created (Bruland & Richardson, 2006).

Noyd, Pflieger, Norland and Sadowsky (1995) studied the effect of mitigation treatments on actinomycetes, fungi, free-living nitrogen-fixing bacteria, and aerobic heterotrophic bacteria when compared in field plots in coarse taconite tailing. The authors concluded that incorporation of a moderate rate of organic matter can ameliorate the stressful conditions of coarse taconite tiling, and can enhance the initiation of a functional soil ecosystem able to support the establishment of seeded native prairie grasses. Noyd et al. also suggested that such amendments could provide a long-term solution to the reclamation of taconite tailing.

When soils are disturbed, organic matter previously protected from microbial action is decomposed rapidly because of changes in water, air, and temperature conditions, and the breakdown of soil aggregates accelerates erosion; a soil with high organic matter is more productive than soil where much of the organic matter has been reduced through tillage, excavation and poor management practices and transported by surface runoff and erosion (McCauley et al., 2003). Nair et al. (2001) consider organic matter accumulation to be one of the most important indicators of functional wetlands.

In a study conducted in 2001, Nair and colleagues compared the soil characteristics of created wetlands and native wetlands in phosphate-mined areas in central and north Florida. The authors took 151 samples (with soil cores 20-30 centimeters deep) along the hydrologic gradients of the created site, adjacent created and natural wetlands of the same wetland type, wetlands with different plant growth, and wetlands of different construction ages. The method of sampling and sampling strategy may differ per mitigation site, but the authors emphasize that it is important to get a representative sample of the microbial community in the mitigation area. Results of the

Nair et al. (2001) study showed that organic matter accumulation increased across transect going from uplands toward the center of the wetland, and thus showed promise of the constructed wetlands returning to “typical” wetlands.

Nair et al. also called attention to the use of reference wetlands. Wetland soils reflect the surficial geology and characteristics of parent material. This fact is critical for soil maintenance and important when comparing mitigated wetlands to natural, reference wetlands. Several studies have used eco-regions (areas differentiated by regional soil differences) as the framework for evaluating restoration and creation projects (Abbruzzese, Allen, Henderson & Kentula, 1988; Brooks & Hughes, 1988). Overall, information on soils of wetland ecosystems is essential in order to re-establish native vegetation after phosphate mining. The advantages of a reference wetland approach include:

- Allowing for explicit goals for compensatory mitigation through the identification of reference standards derived from data that typify sustainable conditions in a region;
- Providing templates from which restored and created wetlands can be designed; and
- Establishing a framework whereby a decline in functions (resulting from adverse impacts) or recovery of functions (following restoration) can be estimated both for a single project over a larger area accumulated over time (Brinson & Rheinhardt, 1996).

To establish reference standards, conditions inherent to highly functioning sites must be identified for classes of wetlands that share similar geomorphic settings, are of about equal age and have similar sedimentary regimes and vegetation densities (Brinson & Rheinhardt, 1996). This is because microbial communities respond strongly to changes in sediment organic matter, which usually accumulates with wetland age. Pratt and Cairns (1985) found that recently disturbed ponds had fewer microbe species than did natural and older reclaimed ponds on a surface-mined site. However, the microbial communities in ponds more than two years old were indistinguishable from those in older reclaimed, unreclaimed, and natural ponds despite differences in water quality. Other factors that could be important to standardize among collections of microbial

communities include: light penetration (water depth, turbidity, shade), temperature, sediment oxygen, baseline chemistry of waters (particularly pH and conductivity), detention time, current velocity, vegetation density, dominant vegetation species, and moisture (Adamus & Brandt, 2004).

Conclusions

The incorporation of soil microbiology can ensure a functional reclaimed wetland. Legislation thus far has identified three characteristics that comprise a wetland – its hydrology, vegetation and soil. The soil component is being thoroughly investigated and is proving relevant and pertinent. Microorganisms in wetland ecosystems play a much more vital role than the legislation acknowledges. Microbial indicators should not serve as a “quick fix” for reclamation problems, but instead should be considered a major part of the planning and implementation of reclamation projects. The science may seem complicated, but it is essential to include all of the components of natural wetlands and mitigate wetlands that are functionally, biologically and physically equivalent to natural systems.

Chapter 5: Ecological Research and Environmental Policy

Introduction

The abundance of wetland literature should reflect the diversity and totality of the mitigation policies, but it does not. In terms of the interface between science and policy, the discrepancies continue to increase and the guidelines remain outdated and selective. One explanation for this “gap” is the value of certain ecological systems is difficult to determine, and thus remains subjective. Murtaugh (1996) suggests the use of ecological indicators to determine environmental quality, and the establishment of statistical models to evaluate the usefulness of the indicator. Some authors suggest that ecologists should think more like policy makers, and some suggest the opposite (Pouyat, 1999). A point of contention lies at the interface between scientists and nonscientists, and what ensues when they attempt to communicate (Rykiel, 2001). Weber and Word (2001) emphasizes that non-scientists assumes that scientists are advocating a position, while scientists believe that they are only providing objective information. These authors recommend that the education of biologists and ecologists should include a communications component aimed at understanding the multifaceted interactions between what scientists say and what nonscientists hear (Weber & Word, 2001).

Although critiques of the wetland mitigation process are available to scientists and policy-makers (i.e. Race, 1985), the guidelines remain inadequate. Government agencies have the dual responsibility of safeguarding the nation’s wetland and providing avenues for development. Debate continues as federal policy mandates permits that are to be given only if impacts to wetlands are “unavoidable,” but in reality mitigation is used to compensate for most wetland habitat destruction. On a landscape level, the fragmentation of these wetlands also translates into fragmented habitats and the disruption of natural systems (Bedford, 1996). According to National Research Council’s reports, mitigation efforts cannot yet claim to have duplicated lost wetland functional values (NRC, 1992).

Based on preliminary literature reviews, it seems that most ecologists are calling for a reordering of priorities. They envision policy-makers realizing the need for improved compliance, generating thriving wetlands and maintaining a true baseline. My final objective is to reinforce the wealth of mitigation literature by documenting the relevant literature and highlighting the outstanding arguments. In doing so, I can provide a more distinct scientific and policy review for researchers, land managers and legislators. Chapter 5 begins with an outline the current literature, continues by synthesizing the varying point of view, and concludes with some possible ways to bridge the science-policy gap.

Ecological Research and Policy

Congressional policymaking has had a significant impact on the management of wetlands in the United States. Tzoumis (1998) examined 240 congressional hearings and 1,569 witnesses who testified from 1789 to 1995 on wetlands. His findings showed three distinct eras exist in wetland policymaking:

- Era I (1789-1945), which was characterized by a dominant monopoly of agricultural and development issues in congressional policymaking
- Era II (1946-1965), which was the beginning of public attention to environmental issue
- Era III (1966-1995), which marked a period of wetland policymaking that was characterized as “contentious,” and no one issue controlled legislation.

Unlike Era I, today’s congressional policymaking reflects many different issues. The growth of wetland science and continued public interest presents a challenge to the dominant agriculture and development interests that once controlled congressional policymaking. Wetland policy debates now include issues such as leisure, environmental protection, private property, and economics.

A series of Local, State, Federal and private programs are available for wetland protection, but implementation of these programs has been unsuccessful, therefore a continued loss of natural wetlands remains (Whigham, 1999). Even though Section 404 of the Clean Water Act establishes the major Federal regulatory programs protecting

wetlands, no explicit goals for their management have been set (Dennison & Berry, 1993). In terms of biodiversity, current wetland protection policies fail because restored and created wetlands are often distinct from natural wetlands. Wetland policies often do not address the preservation or restoration of critical ecological processes, such as nutrient cycling, because they mostly focus on individual wetland and ignore the fact that wetlands are integral parts of landscapes (Whigham, 1999).

Wetland mitigation projects often result in the exchange of one type of wetland for another and thus result in a loss of wetland functions at the landscape level. The most striking weakness in the current national wetlands policy is the lack of protection of dry-end wetlands. From an ecological perspective, dry-end wetlands (such as isolated seasonal wetlands and riparian wetland associated with first order streams) may be the most important landscape elements. They often support a high biodiversity and are impacted by human activities more than other types of wetlands (Mitsch & Gosslink, 1993). The failings of current wetland protection and mitigation policies also reflect the lack of ecologically sound wetland assessment methods for guiding decision making processes.

Examples of current wetland functional methodologies used for assessment are the Hydrogeomorphic Classification Model (HGM) (Brinson, 1993; Smith, 1995), and the Wetland Evaluation Technique (WET) (Adamus, 1991). The HGM is based on three fundamental factors that influence how wetlands function: (1) the position in the landscape (geomorphic setting), (2) hydrology, and (3) the flow and fluctuation of the water once in the wetland or hydrodynamics (Brinson, 1993). This assessment technique is described further in Chapter 1, Wetlands Classifications and Settings. For the WET approach, eleven functions and values are addressed:

- Ground water recharge;
- Ground water discharge;
- Flood-flow alteration;
- Sediment stabilization;
- Sediment/toxicant retention;
- Nutrient removal/transformation;

- Production export;
- Wildlife diversity/abundance;
- Aquatic diversity/abundance;
- Recreation; and
- Uniqueness/heritage (Adamus, 1991).

The WET approach also provides a procedure to evaluate habitat suitability for 14 waterfowl species groups, 4 freshwater fish species groups, 120 species of wetland-dependent birds, and 133 species of saltwater fish and invertebrates (Adamus, 1991). The method evaluates the probability that a function will occur as high, moderate, or low.

In addition, Fonseca et al. (2000) developed the Habitat Equivalency Analysis Technique which computes the amount of habitat to be restored to compensate for on-site lost ecological services. The Wetlands Reserve Program (Wetlands Reserve Program [WRP], 1993) provides a bibliography of other wetland mitigation evaluation techniques (Appendix C).

Political jurisdictions do not necessarily correspond to ecologically significant boundaries. Attempts at engaging more than one agency or organization to jointly set goals for a particular location have remained mixed. The breadth of wetland issues, the various agencies and private landowners, and the saturation of disjointed wetland research make it difficult to generalize about any one case or answer any specific questions. Thus the planning process is hindered and questions about the planning process remain unanswered (Dennison & Berry, 1993). One solution is adaptive management.

Adaptive management involves continually improving management policies and practices by learning from the outcomes of operational programs. (Holling, 1978; Walters, 1986; Walters & Holing, 1990). This form of management requires communication and cooperation, and that all groups understand the overall goal (Brown, 2005). In all, the stakeholders, including scientists, miners, regulators and local citizens, are involved in a program of co-evolution of science, management and policy that somewhat parallels the practice of modern resource management known as adaptive management (Holling, 1978; Walter & Holling, 1990; Gunderson & Holling, 2002).

The fate of wetland preservation depends on scientists, the phosphate industry and governmental regulators all sharing common goals. Ongoing research is needed to provide information and valuable insight into the necessary corrections that can be made and new approaches to defining success, but the information becomes scattered and disjointed. Because it is difficult for governmental officials to keep up with the latest development and technical details of environmental research, they rely on review articles, government summary reporters and information from professional scientists with whom they interact (Race, 1985). In addition, many published reports are misleading and an inaccurate picture of the status of restoration attempts is projected (Race, 1985). The time has come to start synthesizing all of the material and use it to generate answers instead of sparking more discussions.

Regulators and managers routinely use simple indicators of success. Normally 3-5 years of once- or twice-per-year monitoring is required, with easily measured parameters such as plant lists, animal identification, and percentage of vegetation cover as the overall indicators. Assessing success is then based on comparing these parameters with a simple set of criteria that were stipulated in the original permit for the project, but these criteria may or may not accurately reflect wetland function (Mitsch & Wilson, 1996). The problem is that the natural factors are not used cohesively in mitigation projects.

Managers are also unable to critically analyze the success of restoration projects because of the great variability in nature and scope, and the inconsistencies in definitions and classification schemes in the literature. Not only are there several definitions for wetlands, but there are various definitions for how to monitor a reclaimed wetland. Of the various monitoring techniques, the two types that are particularly relevant to restoration are implementation and effectiveness monitoring. Implementation monitoring is used to assess if a directed management action has been fulfilled. Implementation monitoring quantifies changes immediately after treatments, and evaluates whether treatments were done as prescribed (Noss & Cooperrider, 1994). Effectiveness monitoring is used to determine whether the action achieved the ultimate objective. This type of monitoring requires response variables to be clearly articulated so that they can be measured accurately and precisely. Typical response variables for wildlife are related to species'

habitats or populations (Noss & Cooperrider, 1994). If the goal of restoration is not clearly stated and monitoring procedures do not consider the ecosystem functions necessary for wetland restoration, monitoring becomes nothing more than literally “watching the grass grow.” In addition, consultants and landscape architects only work for 1 year, so effective and consistent monitoring becomes a problem.

In addition, the land structural components of wetlands, rather than the dynamic processes necessary for wetland development (hydrology, sedimentology, etc.), are usually of primary concern in mitigation projects. Short-term projects cannot predict complex, long-term ecological processes. Some aspects of a project (such as the taxa or richness of epibenthic organisms, fish and birds) may be indicative of system maturity, but most (like sediment organic content) need longer time (7 or more years) for comparable results (Zedler, 1988). The effectiveness of the short-term indicators depends on the predictors used; the predictors have to be able to describe patterns, trends and variability in natural wetlands as the system matures and responds to disturbance and natural variability.

In a study to test the predictability of the long-term processes and the short-term expediency of managers, Simenstad and Thom (1996) analyzed 16 ecosystem functional attributes of the Gog-Le-Hi-Te Wetlands in the Puyallup River Estuary in Puget Sound, Washington which were historically modified by industrial development. The remaining wetlands were small, fragmented and contained industrial wastewater. The authors monitored topography, sediments, vegetation, water chemistry, emergent plant growth and survival, benthic and invertebrate composition, and fish and bird species density. Over the 7-year study, results indicted that only a few factors showed functional trajectories toward equivalency with natural wetlands (epibenthic organisms and fish in high density), and some factors (e.g., sediment levels) indicated an immature system (Simenstad & Thom, 1996).

The restoration process is influenced by the type of wetland being restored, the region of the country, the ecological functions of interest, the type and degree of degradation, the surrounding land uses, and the ability to establish and maintain the appropriate hydrology. Attributes of a similar, nearby wetland can be used for

comparison, but success depends on the choice of the reference site. Managers have to look at the wetland type, size, ecological setting, land use, and position in the watershed. Thus by comparing populations of wetlands, an assessment of the cumulative effectiveness of restoration can be made. In addition, managers should be flexible in their mitigation parameters. For example, if the wetland system is in poor condition, but represent a rare wetland type, a manager should choose to improve rather than replace.

The EPA inventoried the CWA Section 404 permit records from the 1970s and 1980s from 8 states (Oregon, Washington, California, Texas, Arkansas, Alabama, Mississippi, and Louisiana), and documented 724 permits in the cumulative record, with 898 wetlands impacted and 745 compensatory wetlands required (Zedler, 1996). Not only has the distribution of wetlands within these states been altered, but the ecosystem dynamics have become diminished because land managers tend to replace the same types of wetlands. Overall, the EPA concluded that mitigation goals depend on the parameters defined for assessment and the local and regional landscape structure, and in these 8 states, the mitigation projects were largely unsuccessful in meeting their stated goals.

Several authors have criticized that the United States has based legislation on the premature adoption of ideas. For example, Race (1985) contends that the Army Corps of Engineers' experiments to establish vegetation on dredged soil was extrapolated by some to mean that entire habitats could be created. Then, on the presumption that experimental projects would succeed, policy makers adopted the technology as a mitigation measure in the permitting process (Race, 1985). The nation is thus basing our policies on the success of a few projects. On the academic side, published reports do not provide the variability of the site in comparison to other sites; consequently it is difficult to distinguish between failures, limited success and ongoing projects (Race, 1985).

We are at the point that a framework needs to be established for successful and effective mitigation, and we should work together to use upcoming data to modify, refine, and improve policy. The number of permits granted is small, but the repercussions of these permits are astounding. The goal of wetland restoration should be to get wetland area back, reverse loss of area, reestablish natural ecological processes (hydrology, geochemical) and reestablish wetland function. The importance of the resource is not in

question, but despite extensive efforts, it remains difficult to quantify the range of benefits provided by wetlands, mainly due to the dynamic nature of the resource. Additionally, a discrepancy exists between the short life span of monitoring programs (3 to 5 years) and reclamation success. All of these factors contribute to an overall feeling of unrest and dissatisfaction with current wetland policy.

Proposed Solutions to the Science-Policy Gap

My first recommendation is the incorporation of microbial analysis in wetland mitigation. Microorganisms have proven to be a steadfast indicator of ecosystem health, and the procedures involving laboratory and field tests have proved (and continue to prove) very valuable (see Chapter 4).

A second recommendation is the generation of annual reviews of the progress of mitigation projects. The subsequent reviews would then refer those previously completed for baselines, but would report new findings. In any given year, a tremendous amount of literature is produced, so annual summaries could prove valuable. Academic summaries can exist, as well as those that summarize projects from local, national government or private organizations. Multiple reviews would allow for different points of view or emphasis. These reviews could be submitted to legislative bodies for inclusion into new policy. International summaries would also need to be completed; we need to stay abreast of international problems and solutions because there is a need to start addressing ecological problems with more of a global approach.

A third recommendation is the inclusion of academic researchers (and/or scientists) and wetland organizations into the process of mitigation banking. Land owners or managers that need to compensate for authorized impacts to wetlands associated with development activities may choose to purchase credits from an approved mitigation bank, rather than reclaiming wetlands on or near the development site. The value of a mitigation bank is determined by quantifying the wetland values restored or created in terms of “credits” (Brown & Lant, 1999). Various techniques are used to establish credits for mitigation banks.

Banking can provide more cost effective mitigation and reduce uncertainty and delays for qualified projects, especially when the project is associated with a comprehensive planning effort. This technique eliminates the temporal losses of wetland values that typically occur when mitigation is initiated during or after the development impacts occur. Consolidation of numerous small, isolated or fragmented mitigation projects into a single large parcel of may have greater ecological benefit. In addition, the money generated from mitigation banks can then be used for acquisition of land suitable for future restoration made in anticipation of technology and knowledge to come. A mitigation bank can bring scientific and planning expertise and financial resources together, thereby increasing the likelihood of success in a way not practical for individual mitigation efforts. It is my opinion that to compel managers to develop ecosystems that they do not understand is futile, and only contributes to poor reclamation results and wasted mitigation dollars.

Disadvantages of mitigation banking include the on-site versus off-site debate surrounding wetlands. Mitigated banks are often placed in areas outside the scope of the wetland damaged and thus no longer benefit the local ecosystem. This transplanted wetland may also not fulfill functions present in the degraded wetland. For example, migratory birds can be displaced and vulnerable other wetland species dislocated. Mitigation banking is not a perfect solution and in some cases may not be the best solution, and we have to look for answers within the framework and resources we have available. Federal support exists for mitigation banking and interagency guidance for the establishment and use of mitigation banks is currently being developed. As of 2002, there were 219 active mitigation banks in the United States, encompassing approximately 120,000 acres in 29 states (Spieles, 2005). Table 5.1 lists some of the mitigation banks in the United States and Figure 5.1 shows their distribution.

The Environmental Reorganization Act of 1993 directed the FDEP and state water management districts to adopt rules governing mitigation banking. The FDEP admits that although mitigation banking encourages restoration and promotes interconnected tracts of wetlands, it allows the destruction of smaller wetlands that provide important habitats for certain species such as the wood stork (FDEP 2006b). For

more information on mitigation banking in Florida, please visit the DEP mitigation banking website (<http://www.dep.state.fl.us/water/>).

Spieles (2005) conducted a study to determine if mitigation banks successfully support native wetland vegetation and if success differs by mitigation method (created, restored, or enhanced), geomorphic class, age, or area. Three measures of ecological status were evaluated: (1) the prevalence of wetland vegetation; (2) the pervasiveness of non-native species; and (3) plant species richness. Sites ranged from less than 2 acres to over 1,300 acres, and included 17 created wetlands, 19 restored wetlands, and 9 enhanced wetlands. Spieles (2005) concluded that the plant community homogeneity increased with age, which indicated a period of self-organization and a potential trend toward vegetative equivalence with natural wetlands.

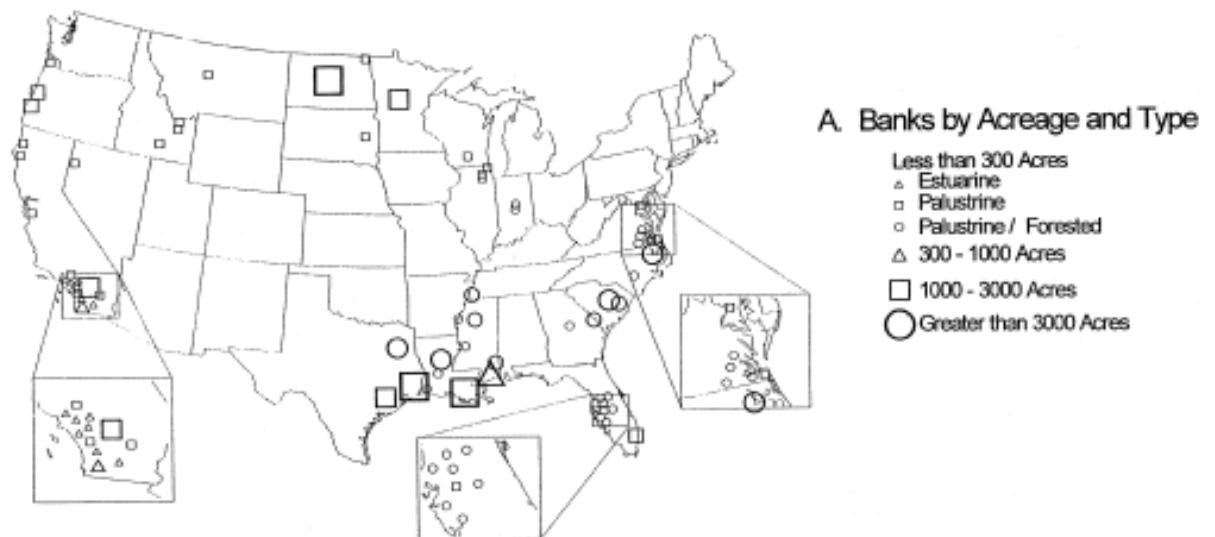
Another study examined whether 68 wetland mitigation banks in existence in the United States in 1996 had achieved “no net loss” of wetland acreage nationally and regionally (Brown & Lant, 1999). Although 74 percent of the individual banks achieve “no net loss” by acreage, overall, wetland mitigation banks are projected to result in a net loss of 21,328 acres of wetlands. The authors stated that although most wetland mitigation banks are using appropriate compensation methods and ratios, several of the largest banks use preservation or enhancement, instead of restoration or creation. Ten of these banks, concentrated in the Western Gulf Coast Region, accounted for over 99 percent of the wetland lost. Even with these results, Brown and Lant advocate mitigation banking as “a sound environmental policy and planning tool” but only if applied according to recently issued guidelines that ensure “no net loss” of wetland functions and values (Brown & Lant, 1999, p. 333).

Table 5.1: Wetland Mitigation Banks of the United States (Brown & Lant, 1999).

Year est.	Bank name	Location	Acreage	Service area	Comp. method	Comp. ratio	Project acreage gain (loss)
Lacustrine							
1984	Port of Los Angeles	Orange Co., CA	18	Cst. pl.	creation	1:1	0
Estuarine							
1984	Goose Creek Tidal	Isle of Wight Co., VA	11	Cst. pl.	creation	2:1	6
1984	Cabrillo Marina	Orange Co., CA	18	Cst. pl.	rest.	1:1	0
1984	Port of Long Beach, Pier A	Orange Co., CA	29	Cst. pl.	rest.	1.5:1	10
1985	Pac Tex, Batiquitos Lagoon	San Diego Co., CA	383	Cst. pl.	rest.	1.5:1	128
1986	Sea World Eelgrass	San Diego Co., CA	1	W'shed	rest.	1.2:1	0.2
1986	Anaheim Bay Project	Orange Co., CA	120	Cst. pl.	rest.	1.32:1	29
1986	Naval Amphibious Base	San Diego Co., CA	10	Cst. pl.	rest.	1:1	0
1986	SMAP-Port of Pascagoula	Jackson Co. MS	4675	W'shed	preserv.	1:1	(4675)
1988	Huntingdon Restoration	Orange Co., CA	25	W'shed	rest.	1:1	0
Palustrine emergent							
1975	Rogers & Drayton Interch.	Pembina Co., ND	175	W'shed	rest.	1:1	0
1980	Bracut Marsh	Humboldt Co., CA	6	County	rest.	3:1	4
1984	Henderson Marsh	Coos Co. OR	420	W'shed	creation	1:1	0
1984	Fina LaTerre	Terrebonne Parish, LA	7014	W'shed	enhance	2:1	(3507)
1987	Astoria Airport	Clatsop Co., OR	33	County	rest.	1:1	0
1987	Minn. Wetland Habitat	Minnesota (40 sites)	1750	DOT	creation	1:1	0
1987	North Dakota State Hwy.	ND (numerous sites)	5000	State	rest.	1:1	0
1987	West Eugene	Lane Co. OR	980	County	enhance	1:1	(980)
1988	Wetlands Acct. Systems	Kingsbury Co. SD	25	W'shed	rest.	1:1	0
1988	San Joaquin Marsh	Orange Co., CA	18	County	enhance	1:1	(18)
1988	Mid City Ranch	Humboldt Co., CA	8	County	rest.	1:1	0
1988	Washoe Lake	Washoe Co., NV	89	W'shed	enhance	0.3:1	(208)
1988	Patrick Lake	Dane Co., WI	165	COE	rest.	1:1	0
1988	Mud Lake	Jefferson Co. ID	125	W'shed	enhance	1:1	(125)
1989	Old Beaver	Clark Co., ID	16	W'shed	rest.	1:1	0
1989	Montana DOT	Montana (3 sites)	289	W'shed	rest.	1:1	0
1991	Acequia	Minidoka Co., ID	28	W'shed	creation	1:1	0
1991	Southeast Mitigation	Hillsborough Co., FL	31	W'shed	enhance	1:1	(31)
1993	Maryland State Highway	Prince George Co., MD	170	W'shed	rest.	1:1	0
1993	Mission-Viejo-ACWHEP	Orange Co., CA	35	W'shed	enhance	3:1	(12)
1994	Land & Water Resources	Kane Co., IL	48	W'shed	rest.	1:1	0
1994	Blue Elbow Swamp	Orange Co. TX	3343	W'shed	preserv.	1:1	(3343)
1994	City of Virginia Beach	Vir. Beach Co., VA	80	W'shed	rest.	1:1	0
1995	San Marcos Creek	San Marcos Co., CA	57	W'shed	creation	1:1	0
1995	Santa Ana River	Riverside Co., CA	1500	W'shed	creation	1:1	0
1995	Mystice Lake	Riverside Co., CA	135	County	creation	1:1	0
1995	Florida Wetlandsbank	Broward Co., FL	358	W'shed	rest.	1.2:1	60
1995	Ferson Creek	Kane County, IL	82	W'shed	rest.	1:1	0
1995	Wetland Research, Inc.	Lake Co., IL	118	W'shed	rest.	1.5:1	39
1995	Greens Bayou	Harris Co., TX	1450	County	preserv.	1:1	(1450)
Palustrine forested							
1982	Louisiana Dept. of Trans.	Grant, LaSalle P., LA	2944	State	preserv.	1:1	(2944)
1985	Company Swamp	Bertie Co., NC	1031	State	preserv.	1:1	(1031)
1987	Pridgen Flats	Sampson Co. NC	127	DOT	rest.	1:1	0
1987	Georgia Dept. of Trans.	Georgia	250	W'shed	rest.	2:1	125
1988	Dahomey NWR	Bolivar Co. MS	162	COE	enhance	1:1	(162)
1988	Malmanson Wildlife Area	Grenada Co., MS	318	COE	preserv.	1:1	(318)
1988	Pitcher Plant Bog	Greene Co., MS	360	COE	rest.	1:1	0
1989	Wetsenfeld-Meadow	Orange Co., FL	235	W'shed	preserv.	6:1	(39)
1990	Millhaven, Inc.	Burke, Screven Co. GA	350	County	rest.	1:1	0
1990	Otterdam Swamp	Greenville Co., VA	14	COE	creation	2:1	7
1990	Polk Parkway	Polk County, FL	3	County	creation	2.5:1	2
1990	Hillsborough County	Hillsborough Co., FL	13	County	creation	1:1	0
1990	Northlakes Park	Hillsborough Co., FL	11	County	rest.	2.5:1	7

1991	Cheval Players Club	Hillsborough Co., FL	27	County	creation	1:1	0
1991	Turner Citrus, Inc.	Desota Co., FL	47	W'shed	rest.	2:1	24
1992	Pine Flatwood	Cameron Co. LA	150	State	enhance	1:1	(150)
1992	Geist Reservoir	Marion Co., IN	25	County	rest.	1:1	0
1992	Morse Reservoir	Hamilton Co. IN	15	County	rest.	1:1	0
1993	Delta Land Trust	Warren Co., MS	100	W'shed	rest.	1:1	0
1993	Cabin Creek	Prince George Co, VA	9	DOT	creation	2:1	5
1993	Fort Lee	Prince George Co, VA	34	DOT	creation	2:1	17
1993	Faulkbern Tract	Sumter Co. SC	1709	W'shed	enhance	3:1	(570)
1993	State Highway Bank	Williamsbury Co., SC	1000	State	rest.	1:1	0
1994	Anderson Tract	Smith Co. TX	2473	W'shed	preserv.	1:1	(2473)
1995	T.L. James & Co.	Louisiana	200	W'shed	rest.	1:1	0
1995	West Tennessee	Shelby Co., TN	389	W'shed	creation	2:1	195
1995	White Cedar LLC	Suffolk Co. VA	100	County	creation	2:1	50

Figure 5.1: The Distribution of Mitigation Banks in the United States (Brown & Lant, 1999).



Allowing wetland ecologists to panel the executive board of the mitigation banks may offer more effective results. In addition, organizations, such as FIPR, should be involved. FIPR can fund projects to continuously develop more efficient methods of monitoring and their link to the banking process will allow a grass-roots approach to the future policy making.

A fourth suggestion to enhancing the science and policy gap is the adoption of newer technology for wetland analysis. (Mathiyalagan, Grunwald, Reddy & Bloom, 2005) have developed a centralized repository and mechanism to share geospatial data, information and maps of Florida's wetlands and adjacent agricultural ecosystems. They

have also developed an interactive WebGIS and geodatabase for Florida's wetlands, which provides map and data services. Managers can use ArcIMS (commercially available software) which can be extended using the MSAccess database, Java, Visual Basic and Active Server Pages. The Interactive website of Florida's wetlands is accessible at <http://www.giswetlands.ifas.ufl.edu>.

Cohen, Prenger and Debusk (2005) used visible-near infrared reflectance spectroscopy (VNIRS) to detect changes in soil quality. Soils are scanned under artificial illumination using a laboratory spectrometer. A multivariate data technique was then used to relate the post-processed reflectance spectra to laboratory observations, such as pH, organic content, total nitrogen, carbon and phosphorus, and extracellular enzyme activity. The VNIRS technology can provide a low-cost, nondestructive, rapid analysis method of land use assessment that retains high analytical accuracy for numerous soil performance measures (Cohen et al., 2005). Research has primarily targeted agricultural applications, but implications for assessing ecological systems are significant.

In June 1988, FIPR founded the Central Florida Regional Planning Council to develop a geographical database for the southern mining district. The database was designed to depict present land use and projected land use to 2010; recognizing that by 2010, most of the minable phosphate ore will have been extracted from the northern portions of the district and the industry's emphasis will shift almost exclusively to restoration (Brown, 2005).

Currently, in Europe, an effort to monitor and assess the environmental impact of mining has produced a project known as MINEO (Chevrel, Reinhard & Klemens, 2002). The objective of the project was to develop hyperspectral remote sensing methods that can be used to measure and monitor mining and pollution, and to provide baseline standards across the European Union (EU). Hyperspectral imaging sensors produce data that can characterize the chemical and/or mineralogical composition of the imaged ground surface. The advantages of this imaging technique are the reduction in hazardous, time-consuming and expensive field sampling methods, and the capability to gather repeat data and monitor mining pollution (Chevrel et al., 2002). This technique was tested at the Steirischer Erzberg iron ore mine located near the city of Graz in the

province of Styria, Austria. Several intricate maps were generated that targeted that landscape degradation that had occurred due to mining activities, the changes in vegetation, and the levels off contamination from carbonatic iron ore. This technology could greatly benefit Florida mining sites and should be investigated for future use.

A few publications (Costanza, 1993; Mitsch, Straskraba & Jorgensen, 1988) advocate use of simulation modeling in wetland ecology but this tool has some difficulty predicting what would be considered “mitigation success.” A final component would be to develop connections between structural measurements (e.g., vegetation density, diversity, productivity) and functional components (e.g., organic sediment acceleration, nutrient retention). Simply having a list of plant species is inadequate for lawmakers and managers to effectively estimate the success of mitigated wetlands.

All of these technologies and recommendations can be applied to the phosphate-mined reclaimed wetland mitigation, and I hope that in the future they will.

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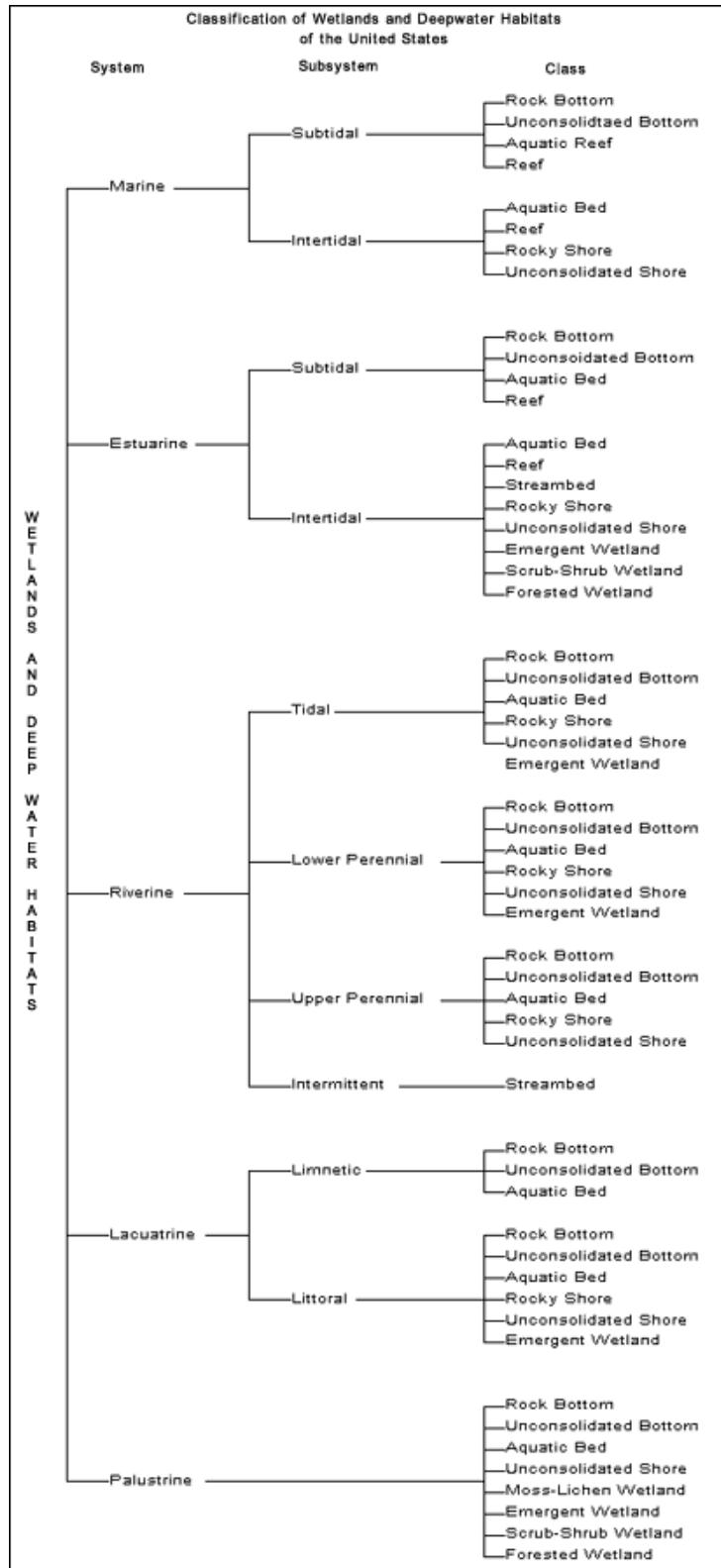
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Appendices

Appendix A: Wetland Classification

Wetlands are classified by the United States, Department of the Interior, Fish and Wildlife Service in a comprehensive hierarchical method that includes five systems and many subsystems and classes. The method is explained in the Classification of Wetlands and Deepwater Habitats of the United States (Cowardin et al., 1979). This classification method includes the marine system and the estuarine system which are ocean based systems and beyond the scope of this document. The other systems are the riverine, Lacustrine and Palustrine systems. The riverine system includes freshwater wetlands associated with stream channels, while the Lacustrine system includes wetlands associated with lakes larger than 20 acres. The Palustrine system includes freshwater wetlands not associated with stream channels, wetlands associated with lakes of less than 20 acres and other wetlands bounded by uplands. Most forested wetlands are in the palustrine system.



**Appendix B: Rules Pertaining To Landscape Restoration as Set Forth by the
Bureau of Mine Reclamation in Chapter 62c-16, Florida Administrative Code.**

B.1. 62C-16.0051 Reclamation and Restoration Standards

This section sets forth the minimum criteria and standards which must be addressed in an application for a program to be approved.

(1) *Backfilling and contouring*: The proposed land use after reclamation and the types of landforms shall be those best suited to enhance the recovery of the land into mature sites with high potential for the use desired.

(a) Slopes of any reclaimed land area shall be no steeper than 4 ft horizontal to one foot vertical to enhance slope stabilization and provide for the safety of the general public.

(2) *Soil zone*

(a) The use of good quality topsoils is encouraged, especially in areas of reclamation by natural succession.

(b) Where topsoil is not used, the operator shall use a suitable growing medium for the type vegetative communities planned.

(3) Wetlands which are within the conceptual plan area which are disturbed by mining operations shall be restored at least acre-for-acre and type-for-type.

(4) *Wetlands and water bodies*: The design of artificially created wetlands and water bodies shall be consistent with health and safety practices, maximize beneficial contributions within local drainage patterns, provide aquatic and wetlands wildlife habitat values, and maintain downstream water quality by preventing erosion and providing nutrient uptake. Water bodies should incorporate a variety of emergent habitats, a balance of deep and shallow water, fluctuating water levels, high ratios of shoreline length to surface area and a variety of shoreline slopes.

(a) At least 25% of the highwater surface area of each water body shall consist of an annual zone of water fluctuation to encourage emergent and transition zone vegetation. This area will also qualify as wetlands under the requirements of subsection (4) above, if requirements in paragraph 62C-16.0051(9)(d) are met. In the event that sufficient shoreline configurations, slopes or water level fluctuations cannot be designed to accommodate this requirement, this deficiency shall be met by constructing additional wetlands adjacent to and hydrologically connected to the water body.

(b) At least 20% of the low water surface shall consist of a zone between the annual low water line and 6 ft below the annual low water line to provide fish bedding areas and submerged vegetation zones.

(c) The operator shall provide either of the following water body perimeter treatments of the high water line:

1. A perimeter greenbelt of vegetation consisting of tree and shrub species indigenous to the area in addition to ground cover. The greenbelt shall be at least 120 ft wide and shall have a slope no steeper than 30 ft horizontal to one foot vertical.

2. A berm of earth around each water body which is of sufficient size to retain at least the first one inch of runoff. The berm shall be set back from the edge of the water body so that it does not interfere with the other requirements of subsection (5).

(5) Water quality

(a) All waters of the state on or leaving the property under control of the taxpayer shall meet applicable water quality standards of the Florida Department of Environmental Protection.

(b) Water within all the wetlands and water bodies shall be of sufficient quality to allow recreation or support fish and other wildlife.

(6) Flooding and drainage

(a) The operator shall take all reasonable steps necessary to eliminate the risk that there will be flooding on lands not controlled by the operator caused by silting or damming of stream channels, channelization, slumping or debris slides, uncontrolled erosion or intentional spoiling or diking or other similar actions within the control of the operator.

(b) The operator shall restore the original drainage pattern of the area to the greatest extent possible. Watershed boundaries shall not be crossed in restoring drainage patterns; watersheds shall be restored within their original boundaries. Temporary roads shall be returned at least to grade where their existence interferes with drainage patterns.

(7) *Revegetation*: The operator shall develop a revegetation plan to achieve permanent revegetation which will minimize soil erosion, conceal the effects of surface mining, and recognize the requirements for appropriate habitat for fish and wildlife.

(a) The operator shall develop a plan for the proposed revegetation, including the species of grasses, shrubs, trees, aquatic and wetlands vegetation to be planted, the spacing of vegetation and, where necessary, the program for treating the soils to prepare them for revegetation.

(b) All upland areas must have established ground cover for 1 year after planting over 80% of the reclaimed upland area, excluding roads, groves or row crops. Bare areas shall not exceed one-quarter (1/4) acre.

(c) Upland forested areas shall be established to resemble pre-mining conditions where practical and where consistent with proposed land uses. At a minimum, 10% of the upland area will be revegetated as upland forested areas with a variety of indigenous hardwoods and conifers. Upland forested areas shall be protected from grazing, mowing or other adverse land uses to allow establishment. An area will be considered to be reforested if a stand density of 200 trees/acre is achieved at the end of 1 year after planting.

(d) All wetland areas shall be restored and revegetated in accordance with the best available technology.

1. Herbaceous wetlands shall achieve a ground cover of at least 50% at the end of 1 year after planting and shall be protected from grazing, mowing or other adverse land uses for 3 years after planting to allow establishment.

2. Wooded wetlands shall achieve a stand density of 200 trees/acre at the end of 1 year after planting and shall be protected from grazing, mowing or other adverse land uses for 5 years or until such time as the trees are 10 ft tall.

(e) All species used in revegetation shall be indigenous species except for agricultural crops, grasses and temporary ground cover vegetation

Appendix C: Wetland Mitigation Bibliography



WRP Technical Note WG-EV-6.1

January 1993

Wetlands Mitigation Evaluation: A Bibliography

PURPOSE: This technical note provides a bibliography of wetland mitigation evaluation techniques literature.

BACKGROUND: In 1990, the U.S. Army Corps of Engineers (USACE) and the U.S. Environmental Protection Agency (EPA) signed a Memorandum of Agreement (MOA) that provided guidance on wetland mitigation activities for Section 404 of the Clean Water Act of 1977 (formerly Federal Water Pollution Control Act), 33 USC 1344, as amended. The MOA references many complex scientific and technical provisions that involve wetland mitigation sequencing (avoidance, minimization, and compensation of adverse project impacts) to obtain full functional replacement for the impacted wetlands. One of the first essential steps in determining the present status of the scientific and technical body of knowledge was to conduct a review of the literature pertinent to wetland mitigation evaluation techniques.

APPROACH: In 1990, an initial literature review was conducted using state-of-the-art computer retrieval systems. Approximately 272 citations were identified. Most of the publications (or the publication abstracts) were obtained and reviewed for scientific or technical content. Policy or philosophical publications were not included. The final list of 24 references on wetland mitigation evaluation techniques was compiled into this bibliography.

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CONCLUSION: This bibliography provides planning, operations, natural resource management, and regulatory personnel with a bibliography of wetland mitigation techniques literature, thereby minimizing intensive, duplicative, and costly literature searches.

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