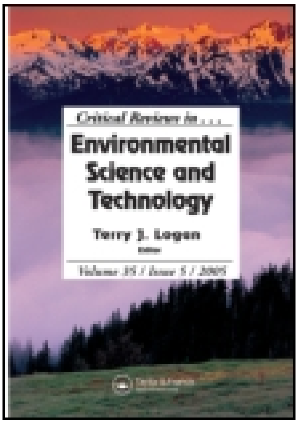

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Landscape Patterns of Significant Soil Nutrients and Contaminants in the Greater Everglades Ecosystem: Past, Present, and Future

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Landscape Patterns of Significant Soil Nutrients and Contaminants in the Greater Everglades Ecosystem: Past, Present, and Future

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The primary goal of this review and synthesis effort is to summarize present landscape patterns of key soil constituents such as carbon (C), phosphorus (P), sulfur (S), and mercury (Hg), all of which are of historical and present interest with respect to Everglades restoration. A secondary goal is to highlight the importance of landscape scale monitoring and assessment of soils in the Everglades Protection Area (EPA) with respect to present and future restoration efforts. Review of present information derived from the two independent landscape scale studies revealed significant patterns of soil thickness, organic matter, and P in the EPA. Two soil constituents of concern, Hg (biological toxicity) and S (linked to increased P

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cycling), also exhibit spatial patterns at the landscape scale, suggesting a need for focused efforts of restoration. Significant patterns of soil enrichment and change suggest a dynamic interaction between environmental stressors and soil biogeochemical properties across the landscape. Trends and patterns at the landscape scale in the EPA suggest that landscape scale monitoring and assessment is necessary and critical to determining the success of restoration efforts. However, several key questions, surrounding appropriate temporal and spatial sampling scales, the standardization of sampling methods, and the significance of short range variability must be addressed to facilitate future landscape scale assessment efforts.

KEYWORDS: Everglades, phosphorus, sulfur, carbon, mercury, soils, landscape

1 INTRODUCTION

The Florida Everglades is one of the most extensive freshwater marshes in the world and due to the uniqueness of the habitats and biota found across the Everglades landscape, it is recognized as a priority ecosystem for conservation (Davis and Ogden, 1994; Ramsar Convention, 2006). Intensive drainage and water control projects, beginning in the 1880s and continuing through much of the 20th century, have left the Everglades ecosystem in a fragile and reduced state. Significant drainage and compartmentalization of large areas of wetlands for agriculture and flood control have initiated a cascade of environmental stressors across the Everglades landscape (Davis and Ogden, 1994; Light and Dineen, 1994; South Florida Water Management District [SFWMD], 1992). These stressors, catalyzed by alterations to historic hydrology, include significant water and soil quality degradation in the form of eutrophication and contamination, extensive soil subsidence, and widespread habitat degradation and loss. Effects of these environmental stressors are observed as significant changes in unique vegetation communities and patterns on the landscape as well as reduction and, in some cases, loss of native fauna.

In recognition of the imperiled status of the Everglades, the Water Resources Development Acts of 1992 and 1996 charged the U.S. Army Corps of Engineers (USACE) with the task of researching the Everglades dilemma and developing a plan to restore and protect the remaining Everglades ecosystem while simultaneously providing for other water resource needs in the area. The Comprehensive Everglades Restoration Plan (CERP) was the result of an effort led by the USACE and the South Florida Water Management District (SFWMD) in collaboration with over 100 scientists from multiple federal and state agencies and the academic community. The CERP was officially authorized by Congress in the Water Resource Development Act of 2000, and provides the framework for Everglades restoration.

The Florida Everglades is presently the focus of the largest ecological restoration effort ever undertaken. As such, the process is expected to take over 30 years and cost in excess of $11 billion. Because of the immense costs and temporal scale of Everglades restoration efforts, CERP requires assessment of key performance measures to track the progress and effectiveness of restoration efforts. These performance measures are strategically defined to be indicators of a healthy, functioning ecosystem. To track performance measures and assess the effectiveness of implemented restoration projects, extensive monitoring of ecosystem attributes associated with performance measures is required. Monitoring efforts have several objectives, the most important being the documentation of baseline condition and associated variability, identification of temporal trends, detection of responses to restoration efforts, and improving scientific understanding of cause and effect relationships associated with specific performance measures (Scheidt and Kalla, 2007). Successful assessment of restoration efforts requires monitoring on several scales, from minute to spatially extensive, such as the scale of microbial processes, vegetation community structure, food web interactions, and finally the landscape-level patterns of the ecosystem.

In effect, the scale of the performance measure and the distribution of its components dictate the scale at which monitoring must be conducted to observe changes. Therefore, monitoring and assessment of restoration becomes an issue of scale. Because multiple restoration targets and performance measures revolve around the retention and restoration of unique landscape patterns and ecological functions of the Everglades ecosystem, a system-wide approach to monitoring and assessment is warranted. This work summarizes the results to date of unprecedented system-wide monitoring of several significant characteristics of soil throughout the EPA at the scale of the Everglades landscape. The soil parameters included are directly related to existing performance measures under CERP and have significant ecological relevance to Everglades restoration efforts (Comprehensive Everglades Restoration Plan [CERP], 2007).

2 RATIONALE FOR USING SOILS TO MONITOR RESTORATION SUCCESS

In wetland systems, particularly peat accreting systems such as the Everglades, soil can serve as a sink or storage pool for ecologically significant nutrients and elements. Likewise, soils can be a source in the biogeochemical cycling of these nutrients and elements, and thus their ecological role as biogeochemical modulators is a significant one (Reddy and DeLaune, 2008).

Nutrient inputs to the oligotrophic Everglades ecosystem are primarily stored in peats, either through deposition of organic material from primary productivity fed from enriched inflows or via sequestration in microbial biomass within the soils, enhanced by increased nutrient availability (Grunwald et al., 2008; Marchant et al., 2009; Newman et al., 1997; Reddy et al., 1993). Soils, therefore, are considered to be integrators of long-term water quality conditions (DeBusk et al. 1994; Newman et al., 1997). Likewise, changes to soil properties occur at a much slower rate in relation to the relatively rapid temporal and spatial changes observed in overlying water quality (Reddy et al., 1999; Stober et al., 2001). When water quality is restored, soils may continue to be a source of nutrients and contaminants of concern for some time. This "memory" or legacy effect is of great interest (Reddy et al., 2005) as it may confound assessment of ongoing restoration activities. Ecological stressors such as water management, soil loss, and water/soil quality, are interrelated and thus efforts to assess and manage these stressors must likewise be integrated. (Scheidt and Kalla, 2007).

The importance of soils in the functioning of the Everglades ecosystem has been recognized by CERP, which includes several conceptual ecological models and performance measures that incorporate soil conditions as indicators (Ogden et al., 2005). Examples of specific performance measures that address or influence soil conditions include water inundation, soil loss, eutrophication, sulfate contamination, and mercury contamination. For instance, a major impact to ecosystem integrity has been the overdrainage of large areas in the northern Everglades, resulting in significant soil subsidence. As the Everglades represents the largest single body of Histosols (organic soils) in the world (Stephens, 1956), abatement of soil loss through subsidence, by restoration of natural soil formation processes and accretion rates, is considered a priority success indicator for CERP (Scheidt et al., 2000). In the Everglades, surface water inundation and vegetative communities are intricately interrelated with soil characteristics. The origin and perpetuation of peat and marl soils are greatly dependent on water depth, the duration of surface water inundation. Diminished surface water inundation has caused soil loss and changes in soil composition, which have in turn resulted in altered vegetative communities. These altered plant communities may cause further changes in soil type and thickness as this different plant community eventually decomposes and forms altered soil (Scheidt and Kalla 2007). Because the aforementioned linkages between soil quality, water quality, and other aspects of ecosystem integrity, such as vegetation, are widely accepted (Daust and Childers, 2004; Davis et al., 2005; Hagerthey et al., 2008; Ogden, 2005), soil represents an ideal ecosystem component for assessing baseline status of the EPA prior to CERP activities (Bruland et al., 2006; CERP, 2007; Reddy et al., 2005; Scheidt and Kalla, 2007; Scheidt et al., 2000). Regular sampling to determine long-term change is, therefore, a critical component in the successful management and restoration of the Everglades (Childers et al., 2003). Because the spatial extent of the EPA encompasses such diverse ecotypes and different levels of stressors and impacts, it is important to monitor the system as a whole versus select areas (Scheidt and Kalla, 2007). Determination of the spatial distribution of soil nutrients and contaminants is, therefore, an effective means to evaluate long-term ecosystem impacts and crucial to assessment of CERP activities (Bruland et al., 2006; Reddy et al. 2005; Scheidt and Kalla, 2007; Scheidt et al., 2000).

3 BRIEF HISTORY OF SOIL RESEARCH IN EVERGLADES

Soils research in the Everglades has a long and varied history. Work most relevant to Everglades restoration and this discussion was initiated by Davis (1946) who in 1943 surveyed depths of the extensive Histosols throughout most of the system while conducting a geologic survey of Florida. This work was followed by Jones (1948) who also reported on the condition of soils and water control in the Everglades. These early studies provided the baseline for comparisons to present extent of soils in the EPA and are the basis for estimating the loss of organic soils (Scheidt and Kalla, 2007; Scheidt et al., 2000). With the advent of drainage within the EAA circa 1910–1920, subsidence of organic soils began. Beginning in 1913, extensive soil subsidence within the EAA was investigated and documented (Gleason and Stone, 1994; Snyder, 2005; Snyder and Davidson, 1994; Stephens, 1956).

The ecological role of soils in the Everglades was not fully recognized until more recently in Everglades history when in the 1970s through the 1990s observations of Everglades decline were linked to phosphorus enrichment. This resulted in much attention and research concerning P loading to the Water Conservation Areas (WCAs)and subsequent ecosystem changes, such as vegetative community shifts and habitat degradation, which are attributed in part to excess soil phosphorus (Davis, 1994; McCormick et al., 2002; Noe et al., 2001; SFWMD, 1992).

Due to overwhelming evidence of P enrichment in the northern Everglades and subsequent ecological impacts, several studies have been conducted to investigate P enrichment in soils of the Everglades (Amador and Jones, 1993, 1995; Chambers and Penderson, 2006; Craft and Richardson, 1993; Daust and Childers, 2004; Koch and Reddy, 1992; Miao and Sklar, 1998; Newman et al., 1996; Newman et al., 1998; Noe et al., 2002; Noe et al., 2003; Qualls and Richardson, 1995) and, in some cases, changes to soil condition over time (Childers et al., 2003). The previously mentioned studies, which often used transects along known phosphorus gradients, were limited in spatial scope or often used soil chemical characteristics as variables while investigating ecosystem responses. This approach was critical to understanding ecosystem function and response and invaluable to developing restoration goals; however, these investigations did not create a systematic evaluation of soil conditions across the Everglades. Documentation of soil conditions at the landscape scale was first approached in WCA-1 in 1991 (Newman et al., 1997), WCA-2A in 1992 (DeBusk et al. 1994), and WCA-3 in 1992 (Reddy et al., 1994). These studies were the first to address soil properties on a spatial scale with sampling densities robust enough to allow spatial modeling.

These first large-scale, spatially intensive studies were conducted concomitantly with the U.S. Environmental Protection Agency's (USEPA) first spatially intensive investigations in the Everglades arena. In 1993, USEPA Region 4 initiated the South Florida Ecosystem Assessment Project, which investigated numerous ecosystem attributes at a large spatial scale across the EPA and Big Cypress National Preserve. This multimedia program later became the Regional Environmental Monitoring and Assessment Program (REMAP), which is presently the largest and oldest ongoing spatially intensive monitoring program for soils (along with other compartments such as surface water, porewater, periphyton, macrophytes, and aquatic fauna) in the Everglades. The REMAP program completed system-wide sampling in three phases from 1993–2005, utilizing a probability based sampling design with a total of 1,145 sites. Each iteration of system-wide sampling included hundreds of sample locations across the EPA, enabling temporal as well as spatial interpretation of patterns and trends in soil attributes. The preeminent strength of the probability-based design used by REMAP is the ability to make quantitative statements across space about the status of indicators of ecological health with known confidence limits across space. (Liu et al., 2008a, 2008b; Scheidt and Kalla, 2007; Scheidt et al., 2000; Stober et al., 2001; Stober et al., 1998; USEPA, 1993). For example, the 2005 sampling indicated that 24.5 ± 6.4% of the EPA had soil total phosphorus concentrations above 500 mg/kg, Florida's definition of impacted for the Everglades, whereas 49.3 ± 7.1% exceeded 400 mk/kg, the CERP restoration goal (Scheidt and Kalla, 2007).

In 2003, Reddy et al. (2005), in conjunction with the SFWMD and the Restoration, Coordination, and Verification (RECOVER) Assessment Team, conducted a system-wide soil-sampling effort, the Everglades Soil Mapping project (ESM), at 1,358 sites in a single phase, utilizing a stratified random sampling design. The areas sampled included the WCAs, the Holey Land and Rotenberger Wildlife Management Areas, Big Cypress National Preserve, Everglades National Park (ENP), and tracks east of the ENP collectively termed the Model Lands. This effort represents the most spatially dense set of observations in a single phase effort to date. The ESM data have been successfully used to investigate soil chemical changes over time by comparison to previous spatial data from the WCAs (Bruland et al., 2007; DeBusk et al., 2001; Grunwald et al., 2008; Grunwald et al., 2004; Marchant et al., 2009).

4 PRESENT STATUS OF SELECT SOIL NUTRIENTS AND CONTAMINANTS

The following discussion of spatial distributions of soil nutrients and contaminants is a compilation of information derived from both the REMAP and ESM projects. Although sampling designs and temporal scales were different among these projects, combined, these two efforts embody the extent our present understanding of landscape-scale patterns in soil nutrients and contaminants (Reddy et al., 2005; Scheidt and Kalla, 2007). We have chosen to discuss only a few of the most ecologically significant and temporally relevant soil parameters (C, P, S, and Hg) that were measured during the completion of these projects. Although all of these constituents are found naturally in the environment and many (C, P, and S) are plant essential nutrients, excessive concentrations of these constituents can have deleterious effects on an ecosystem. In the Everglades, P is a limiting nutrient and in excess is considered a contaminant due to negative effects of P enrichment. Because S has been linked to P cycling and Hg methylation, S can also be considered a contaminant when found in excessive concentrations in the Everglades.

4.1 Organic Matter and Carbon Distributions

One of the most significant attributes of the Everglades ecosystem is the vast area of organic soils it contains (Bruland and Richardson, 2006; Davis, 1946; Davis and Ogden, 1994; Stephens, 1956). The large peat deposits of the northern and central Everglades represent over 5,000 years of soil accretion and, as such, are a repository for nutrients accumulated in this organic matrix over that time period. The extensive Everglades Histosols are also a considerable storage of carbon, which has particular relevance to global climate change. The mean total C content of peat soils in the northern and central Everglades is approximately 47%; however, TC for ENP soils can be much greater due to high inorganic C of marl soils (Reddy et al., 2005). Of more relevance, however, is the organic nature of these soils, which in conjunction with the hydrology of the Everglades provide a unique biogeochemical environment for storage of environmentally significant nutrients and contaminants.

Spatial distribution of soil organic matter indicates that the northern and central Everglades contain a majority of the organic soils in the EPA (Figure 1; Reddy et al., 2005). Soils with the highest organic matter (OM) contents (<90% OM) are found in the A.R.M. Loxahatchee National Wildlife Refuge (WCA-1) and trend down in organic content to the southern portion of WCA-3. The ENP, due to the predominance of more shallow wet prairies and marl prairies, contains lesser peat deposits. Peat soils of the ENP are located in

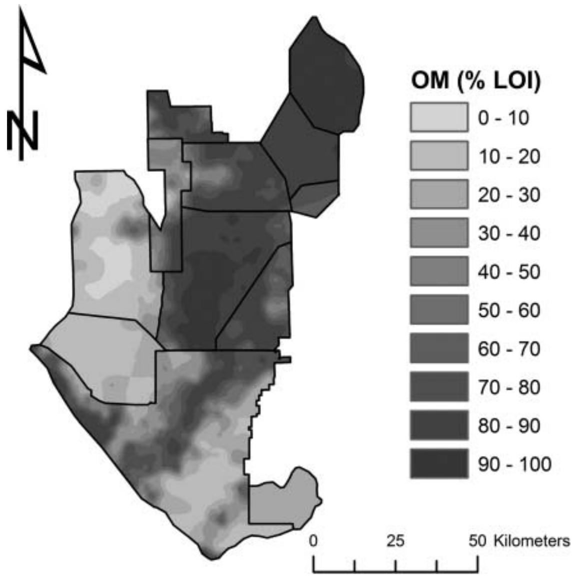

FIGURE 1. Landscape scale patterns of soil organic matter in the Greater Everglades as loss on ignition (LOI). Figure adapted from Reddy et al. (2005). Map by L. R. Ellis.

the Shark River Slough, a predominant drainage feature on the landscape, which is lower in elevation than the surrounding ENP lands and subsequently has a longer hydroperiod. Similar spatial trends of soil organic matter were reported by the REMAP monitoring effort (data not shown; Scheidt and Kalla, 2007; Scheidt et al., 2000). Models of both datasets suggest a trend toward lower organic matter content from the north to the south. The major exception to that trend is an area of interest in the northwestern corner of WCA-3A (Figure 1). This area is also noted to be an area of significant soil subsidence due to chronic over drainage (Scheidt et al., 2000) reported significant soil depth decreases in this area from 1946 to 2005. Comparison of historic soil depth documentation in the Everglades (Davis, 1946) to soil depth measurements conducted under REMAP suggests extensive subsidence in northern WCA-3A (Figure 2). Scheidt et al. (2000) estimated that up to 28% of organic soils have been lost from public lands in the EPA between 1946 and 1996. As of 2005, about 25.1 ± 2.0% of the Everglades had a soil thickness of less than 1.0 foot (Scheidt and Kalla, 2007). The differential between the time involved in accreting organic soils and the relatively short time required to oxidize them is reason for great concern in Everglades restoration. This hysteresis in soil creation and loss has been noted as a driving force in shaping the present-day ecosystem (DeAngelis, 1994) and one that makes it highly unstable (Maltby and Dugan, 1994).

Because Everglades soils contain large storages of nutrients and contaminants, the oxidation of these soils only exacerbates present eutrophication

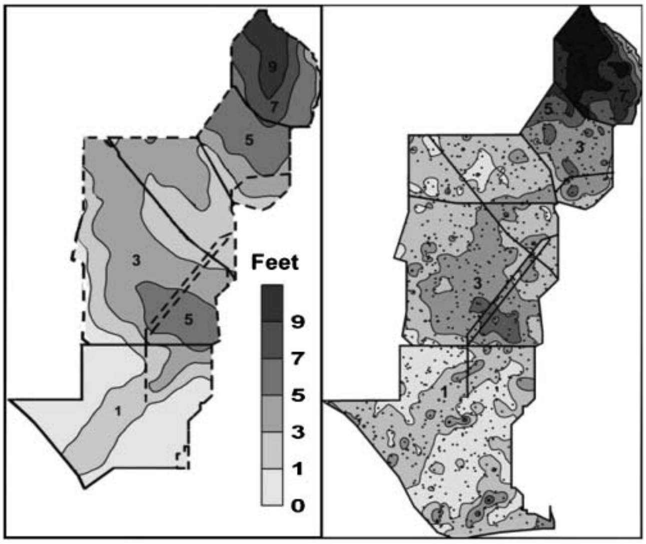

FIGURE 2. Soil depth maps from 1946 (left pane) and from REMAP 1995–2003 (right pane). Comparison of maps indicates significant soil loss in several key areas of the Greater Everglades. Maps created by D. J. Scheidt from Davis (1946) and Scheidt and Kalla (2007).

and contamination problems (Scheidt et al., 2000). Soil subsidence via oxidation releases those nutrients and contaminants that were bound in the organic substrates of soils and protected by the anaerobic conditions prevalent under flooded conditions. This suggests that chronic over drainage due to water diversion or management may be a significant factor responsible for the spread of nutrient and contaminant enrichment in areas less directly affected by agricultural runoff, such as northern WCA-3A.

4.2 Phosphorus

Excessive amounts of the limiting nutrient phosphorus have been the focal point of much research concerning Everglades restoration (Davis and Ogden, 1994). Eutrophication of extensive areas of the northern Everglades via P laden runoff from the EAA was one of the significant catalysts in the movement to restore the Everglades ecosystem and is included in several conceptual ecological models and performance measures used in framing restoration planning (Ogden, 2005; CERP, 2006). The degradation of the northern Everglades marshes due to P enrichment is extremely well documented (LOTAC II, 1990; Childers et al., 2003; DeBusk et al., 2001; Hagerthey et al., 2008; McCormick et al., 2002; Noe et al., 2001; Scheidt et al., 2000) and, as such, significant scientific evidence exists to aid restoration efforts with respect to P.

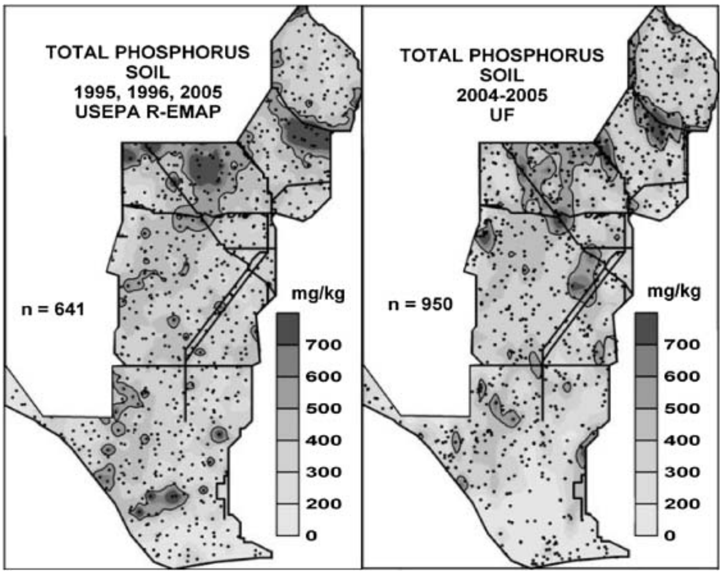

FIGURE 3. Spatial distribution of soil total phosphorus for the 0–10 cm soil profile for both REMAP (Scheidt and Kalla, 2007) and ESM (Reddy et al., 2005) data sets. Maps by D. J. Scheidt.

Soils are known to be long-term integrators of water quality and significant storage pools for P. Florida defines phosphorus impact with areas where soil P exceeds 500 mg kg^{-1}. In addition, CERP instituted a restoration goal of maintaining or reducing longterm average soil TP concentrations at 400 mg kg^{-1}. These goals were based on several studies identifying enriched or impacted soils and correlation with soil TP and resulting expansion of Typha (DeBusk et al., 2001; DeBusk et al. 1994; Doren et al. 1996, Newman et al. 1998; Payne et al., 2003).

Results of the REMAP and ESM sampling efforts in the EPA indicate several areas of concern with respect to soil TP (Figure 3). Although the sampling designs were somewhat different, as discussed previously, similar spatial patterns in TP were found for the 0–10 cm surface soils. Both studies suggest that TP enrichment of soils in WCA-1 is contained in the peripheral areas as P laden agricultural waters often do not penetrate to the interior of the marsh. It is important to note that several more sensitive ecosystem components respond to phosphorus enrichment before increasing TP is manifested in the soils. These initial changes include loss of water column dissolved oxygen and changes to periphyton and macrophyte communities (McCormick et al., 2002). WCA-2A, the site of much historical eutrophication work, maintains a distinct nutrient gradient in the northern portion extending south from the S-10 series outfall structures. However, the ESM data suggest that the area of impact may be smaller than that of the REMAP data. This could be due to variability of sampling locations or nuances of the geostatistical modeling. Interestingly, there appears to be a new area of enrichment, present in both datasets, on the western corner of WCA-2A that was not prevalent in previous sampling efforts in 1990 and 1998 by Reddy et al. (1998) and DeBusk et al. (2001), respectively.

Both R-EMAP and ESM studies indicate significant enrichment in northern WCA-3A, proximate to the Miami Canal outfalls. A smaller area of enrichment is also noted proximate to the L-28 extension canal outfall in west central WCA-3A, which is within the Miccosukee Tribe of Indians Federal Reservation. Spatial models from R-EMAP and ESM indicate that northern WCA-3A has enriched soils; however, no clear gradients exist. This suggests P remobilization from upstream sources is likely the source. Scheidt and Kalla (2007) pointed out that when soil TP is reported across the EPA on a volumetric basis, normalized for bulk density (a highly variable mesure of soil density), many of the enriched areas seen in central WCA-3A are no longer categorized as enriched. Similarly, for marl soils in ENP, which appear to be highly enriched when TP is expressed volumetrically, these soils are not considered enriched. Scheidt et al. (2000) pointed out that >500 mg kg^{-1} is not necessarily indicative of enrichment in mineral and marl soils of ENP. The ENP continues to be the least impacted unit of the EPA with respect to TP.

Because the REMAP program is a multiphase effort, comparisons to previous landscape samplings within that program are possible. Scheidt and Kalla (2007) reported the 1995–1996 sampling revealed $16.3 \pm 4.1\%$ of the soils exceeded the 500 mg kg^{-1} threshold and $33.7 \pm 5.4\%$ exceeded 400 mg kg^{-1}. Subsequent sampling efforts in 2005 indicated $24 \pm 6.4\%$ exceeded 500 mg kg^{-1} and $49.3 \pm 7.1\%$ exceeded 400 mg kg^{-1}. These findings indicate that soil TP has been increasing over the 10-year period, even while P loading has been decreased in the northern Everglades inflows (Scheidt and Kalla, 2007). Supporting the conclusions of Scheidt and Kalla (2007), the ESM sampling effort in 2003 indicated approximately 21% of sites exceed the 500 mg kg^{-1} threshold of enrichment and 42% exceed 400 mg kg^{-1} (Reddy et al., 2005).

Although the ESM program was a single-phase sampling effort, historical datasets have been used to investigate changes to soil TP. Initial spatial mapping of ESM data by hydrologic unit such as WCA-1 (Corstanje et al., 2006), WCA-2 (Rivero et al., 2007), and WCA-3 (Bruland et al., 2006) have been compared to reconstructed spatial datasets to infer changes to soil properties over time. Marchant et al. (2009) compared ESM data from WCA-1 (Corstanje et al., 2006) to the first spatial sampling in 1992 (Newman et al., 1997), revealing significant changes in TP. Likewise, Grunwald et al. (2008) compared 2003 ESM spatial trends (Rivero et al., 2007) with two prior spatial samplings of this unit in 1990 (DeBusk et al., 1994) and 1998 (DeBusk et al., 2001). This analysis revealed extensive areas of both increase and decrease in soil TP suggesting both a reduction in P loading to WCA-2A and evidence for internal cycling of P within the unit. This finding has a significant implication to future restoration efforts because while P loading was decreased, significant enrichment continues to occur due to internal cycling of P.

Bruland et al. (2007) compared 1992 spatial data from WCA-3 (Reddy et al., 1994) to the 2003 ESM spatial data (Bruland et al., 2006) to find significant areas of increase and decrease in TP across the unit.

Bruland et al.'s (2007) WCA-3 study was the first comparison study to come from the ESM program and is illustrative of the value of these landscape-scale investigations to restoration efforts. For example, comparison of the 2003 ESM data (Bruland et al., 2006) to the 1992 work by Reddy et al. (1994) indicated TP increase in 53% of the 0–10 cm soils in WCA-3 (Figure 4). Also, 30% of the surface soils measured in 2003 were considered enriched (>500 mg kg^{-1}) in contrast to 21% of soils measured in 1992. This equates to roughly 1% per year increase in spatial area of soil enrichment. Calculated changes in spatial distributions of soil TP (Figure 4) suggest that significant enrichment occurred in northern WCA-3A, possibly associated with the Miami canal outfalls. It has been suggested that soil oxidation in that area due to subsidence and possibly fire contributed internal loading of P (Bruland et al., 2007; Scheidt and Kalla, 2007; Scheidt et al., 2000). Significant soil subsidence in northern WCA-3A supports this assertion (Scheidt and Kalla, 2007). Of equal interest is the noted decrease in surface soil TP in western WCA-3A at the outfall of the L-28I canal, which in 1992 was a significant area of P enrichment (Figure 4). In 2003, this area indicates significant decrease in surface soil TP, suggesting that changes in water quality and deliver via this canal has significant positive impacts to the surrounding area with respect to P enrichment of soils. Finally, Bruland et al. (2007) echoed the discussion of Scheidt and Kalla (2007) concerning the values of assessing soils with respect to TP on a volumetric basis versus mass basis, as this method can significantly change the outcome of the assessments for a given investigation.

4.3 Sulfur

Although sulfur (S) is a plant essential nutrient, its presence in the Everglades in excess can be a stressor on the system. Sulfur naturally exists in the environment in several forms, for example, as a constituent in organic matter, and is fairly benign. Mineralization of organic S in an aerobic environment results in the environmentally ubiquitous ion sulfate (SO_4^-). Sulfate, an oxidized form of S, can be used as an alternate electron acceptor by sulfate-reducing bacteria (SRB), via a process called sulfate reduction, that results in reduction of sulfate to sulfide (S^{2-}; Reddy and DeLaune, 2008). Sulfate reduction is a dominant biogeochemical pathway in brackish and

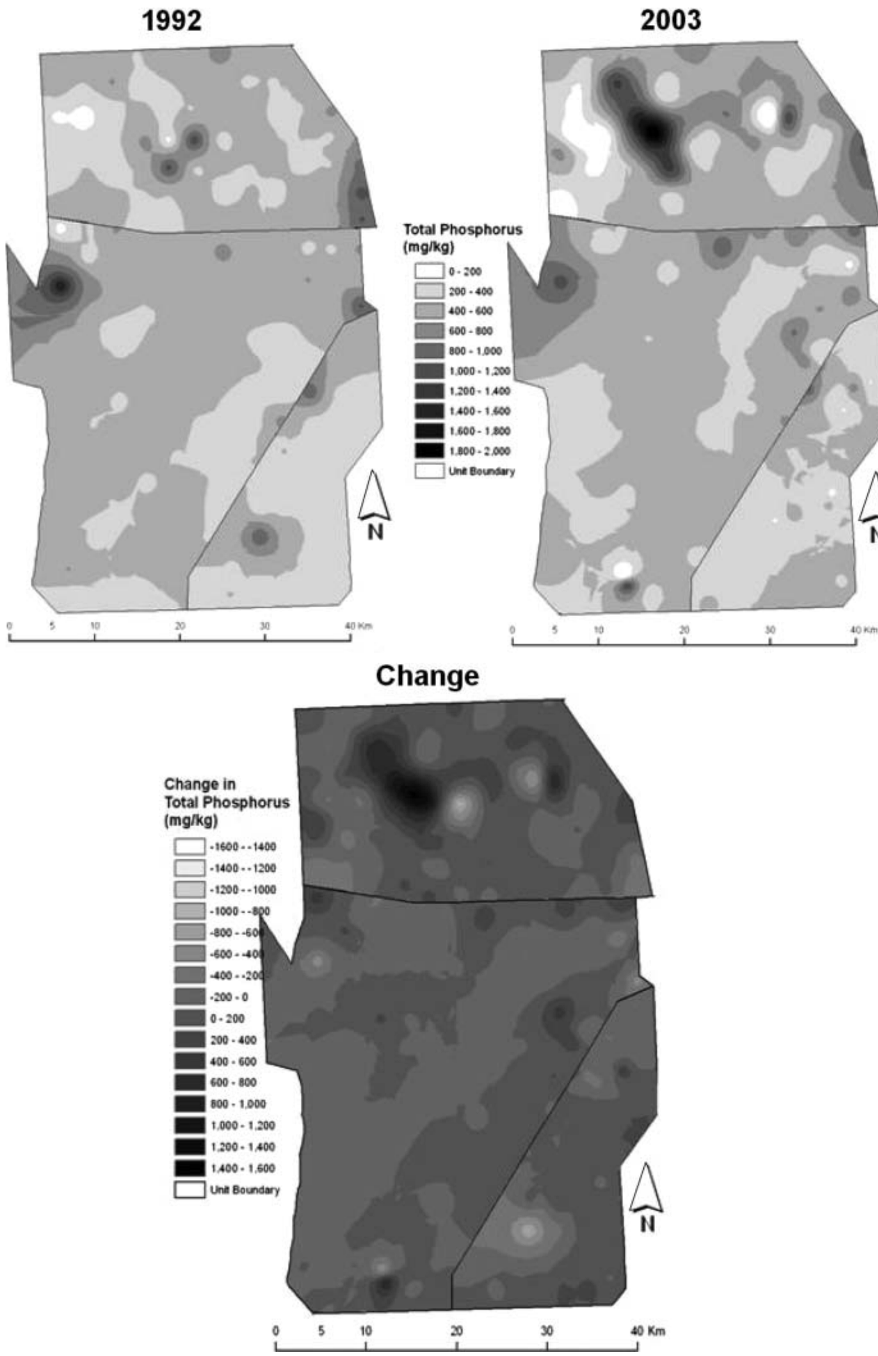

FIGURE 4. Spatial distributions of total phosphorus (TP; mg kg^{-1})in the 0–10 cm soil profile of WCA-3A in 1992 (upper left), 2003 (upper right), and map of change between 1992 and 2003 (lower center). Adapted from Bruland et al. (2007). Maps by G. L. Bruland.

salt marshes worldwide, including the mangrove swamps of the southern Everglades, however, in freshwater, excessive sulfate reduction can have undesired effects.

Sulfur is an element of concern in the Everglades for two main reasons. First, the process of sulfate reduction, mediated by SRB, has been linked to mercury methylation in several studies (Axelrad et al., 2008; Fink and Rawlik, 2000; Gilmour et al., 1992; Jeremiason et al., 2006). Although sulfide can bind Hg into a relatively unreactive form (HgS; Benoit et al., 1999; Gilmour et al., 1998), this reaction is rapidly reversed in the presence of oxygen and thus not a viable mechanism to reduce Hg methylation in the Everglades. Second, the introduction of an alternative electron acceptor (sulfate) en mass to a soil environment characterized by high organic matter and very low oxygen availability can result in accelerated C mineralization and concomitant nutrient and contaminant remobilization. As discussed previously, soil oxidation is greatly reduced when soils are flooded, due to the effective reduction of available oxygen for respiration. In the event that considerable sulfate is available for respiration, nutrient regeneration and soil oxidation can continue undeterred by redox condition.

There is little conjecture that stormwater discharged from the EAA is the main source contributing sulfate to the EPA (Bates et al., 2002; Gabriel et al., 2008; Scheidt et al., 2000; Stober et al., 2001). Bates et al. (2002) identified large soil pools of S in EAA soils, which were linked to elemental sulfur application for fertilizer enhancement. However, Shueneman (2001) reported that S mineralized from soil subsidence, not agricultural use of fertilizers, is the primary source in the EAA. Axelrad et al. (2008) reported that groundwater was not a significant source of sulfate to the EPA, and Gabriel et al. (2008) reported that although atmospheric deposition of sulfate from multiple sources such as marine aerosols does contribute to the sulfate loading in the EPA (0.5–5 mg l^{-1}), it does not constitute a major source in relation to EAA canal sources (5–200+ mg l^{-1}; Gilmour et al., 2007). Conveyance of sulfate-laden agriculture waters into the northern Everglades WCAs is evidenced by surface water sulfate and sulfide distributions reported by Scheidt and Kalla (2007). The highest sulfate and sulfide concentrations occur in WCA-2A, which receives up to 100 mg l^{-1} sulfate, as compared to marsh background of less than 0.2 mg l^{-1}. In 2005 about 57% of the Everglades marsh exceeded the CERP water quality target of 1 mg l^{-1} for sulfate (Scheidt and Kalla, 2007). Spatial patterns of sulfate enrichment in waters of the northern Everglades are indicative of canal water inputs to the WCAs.

As a result of the 2003 soil survey by Reddy et al. (2005), Osborne et al. (2008) reported the spatial distribution of total sulfur (TS) in soils of the EPA and surrounding areas (Figure 5). Spatial patterns of sulfur enrichment in floc are very closely related to agricultural water inputs in the northern Everglades (data not shown). In WCA-1, patterns of soil and floc TS enrichment are very

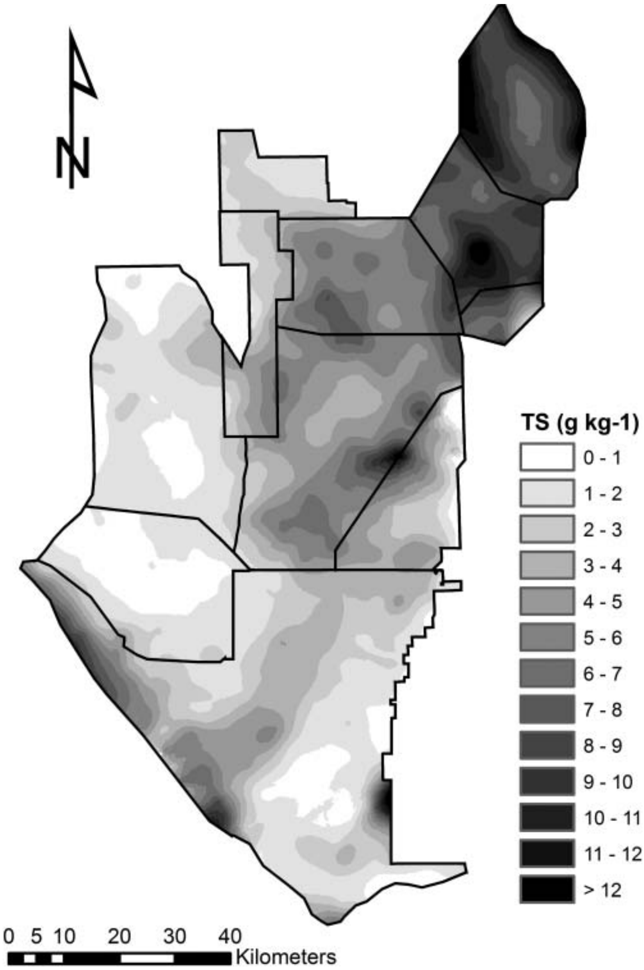

FIGURE 5. Spatial distribution of total sulfur (TS; g kg^{-1}) in the 0–10 cm soil profile of the Greater Everglades. Map adapted from Osborne et al. (2008). Map by L. R. Ellis.

similar and suggest inflow waters, and thus enrichment, is typically limited to the periphery. However, Wang et al. (2009) noted that sulfate from canal discharge is impacting even the interior portions of the Refuge. However, edaphic S enrichment of WCA-2A indicates enrichment zones below the known P enrichment zone in northern portion of the unit. This pattern suggests that much of the S expected in the soils along the eutrophication gradient has been reduced and therefore not observed in the TS analysis. This conclusion is echoed by the patterns in porewater sulfide presented by Scheidt and Kalla (2007). These patterns suggest that a significant portion of S in soils has been reduced in soils impacted by P enrichment. Further, these patterns of high porewater sulfide are also generally associated with higher concentrations of surface water MeHg, suggesting that at least some portion of Hg methylation may be occurring within the soil profile. There appears to be a general spatial association of TS and THg in WCA-1 and WCA-2, however, this association breaks down in WCA-3 and is not present in the ENP. This may be explained in part due to differences in soil organic matter (Figure 1). Of special interest is the marked enrichment of TS in the northern portion of WCA-3 where surface water sulfate and sulfide have been reported to be low (Scheidt and Kalla, 2007). This enrichment front is due north of the largest enrichment area of THg in soils, and with present soil subsidence activity in that area, the potential for continued S remobilization, in concert with loading of agricultural waters, suggests that future migration of S enrichment south may provide for significant increases in Hg methylation in the future.

Finally, TS enrichment of the ENP, the receiving body of waters from the northern Everglades, appears to be relatively low as previously reported by Chambers and Pederson (2006). One area of interest is the headwaters of Taylor Slough, a secondary drainage feature on the landscape and a significant focal point of conservation. The area of enrichment is also spatially similar to that of recent P enrichment and requires further investigation. Other enriched areas in the ENP are associated with zones of marine influence in the mangrove interface, and are not of immediate concern; however, predicted sea level rise may subject those areas to significant sulfate loading and ensuing accelerated OM mineralization in the future.

4.4 Mercury

Beginning in early 1990, mercury contamination has been a concern in the Everglades. Specifically, the methylated or organic form of mercury (MeHg) is a neurotoxic compound that can be bioaccumulated in the tissues of biota and biomagnified as it progresses up the food chain. Exceptionally high levels of MeHg have been observed in top predators (fish, avifauna, panthers) in the Everglades with alarming regularity (Axelrad et al., 2008; Liu et al., 2008b; Roelke et al., 1991; Scheidt and Kalla, 2007; Ware et al., 1990).

Recent studies have identified that the major sources of Hg to the Everglades are both external and internal in nature. Guentzel et al. (2001) reported that roughly 95% of the external loading of Hg to the Everglades is atmospheric in origin. This Hg is deposited via rainfall (wet deposition) or by particulate settling (dry deposition). It has also been reported that local atmospheric sources of Hg, mostly from municipal incineration and industrial emissions, have been significantly reduced in the last two decades (Stober et al., 2001). Other potential sources, such as agriculture drainage water from the EAA, once thought to be a significant input of Hg to the Everglades, have been shown to be very small (~2%; Atkeson et al., 2003; Stober et al., 2001).

Although external sources of Hg to the Everglades system have been reduced, recent investigations into MeHg content of fish reveals slight decreases in tissue MeHg content of fish in the northern Everglades and dramatic increase in tissue content of fishes in the southern Everglades (Axelrad et al., 2008). These findings suggest that internal loading of Hg to the system is likely significant. This shifts our focus to environmental compartments that contain substantial amounts of Hg in the Everglades and elucidating the environmental conditions that are conducive for methylation of mercury within the marsh and subsequent bioaccumulation (Krabbenhoft et al., 2001; Cleckner, L. B. et al., 1999; Vaithiyanathan, P. et al., 1996).

Liu et al. (2008a) reported that analysis of the REMAP data indicated that THg content increased in storage compartments in the Everglades in the order of water, periphyton, floc, and, finally, soil. Although water is often an order of magnitude lower in THg, it often has the highest MeHg:THg ratio. This is likely due to the presence of available C and sulfate for SRB activity.

Several studies (Arfstrom et al., 2000; Delfino et al., 1994; Rood et al., 1995) have reported on Hg distribution in Everglades soils; however, these studies were spatially limited in scope and observational density. Therefore, these studies, although valuable in documenting Hg contamination, did not provide a comprehensive assessment of Hg spatial patterns across the EPA. Stober et al. (2001; 1998) provided the first insight Hg patterns across the EPA when reporting Total Hg (THg) and MeHg concentrations in several media (soil, water, periphyton, and fish) from 1995 and 1999 sampling efforts. These observations were expanded on in 2005 when another phase of REMAP sampling occurred (Scheidt and Kalla, 2007). Cohen et al. (2009) also reported system-wide spatial distributions of soil THg from the ESM system-wide survey of soils in 2003 (Figure 6). This study indicated high levels of enrichment in central WCA-3A as well as southeastern WCA-1 and WCA-2A. Scheidt and Kalla (2007) reported elevated THg in soils in both 1995 and 2005 in geographically similar areas to Cohen et al. (2009). Likewise, THg and MeHg concentrations in surface water, as reported by Scheidt and Kalla (2007), showed highest concentrations in the northern Everglades, which are also areas of THg enrichment in soils as reported by Cohen et al. (2009). However, Cohen et al. (2009) reported the highest values for soil THg (917 μg kg^{-1}) in relation to Stober et al. (2001), who reported values up to 330 μg kg^{-1}, suggesting a possible increase in Hg accumulation in soils. However, median values reported by REMAP have stayed consistent at 130–140 μg kg^{-1} from 1995–1996 to 2005 (Scheidt and Kalla, 2007), and are consistent with the median of 130–150 μg kg^{-1} reported by Cohen et al. (2009).

It is important to note that the presence of Hg is not necessarily a significant risk in that specific conditions must be present to transform Hg into its toxic form MeHg. The Everglades, in comparison to other Hg-contaminated sites, is relatively low in THg in the soil; however, its ratio of MeHg to THg (methyation efficiency) is extremely high. The biogeochemical process that

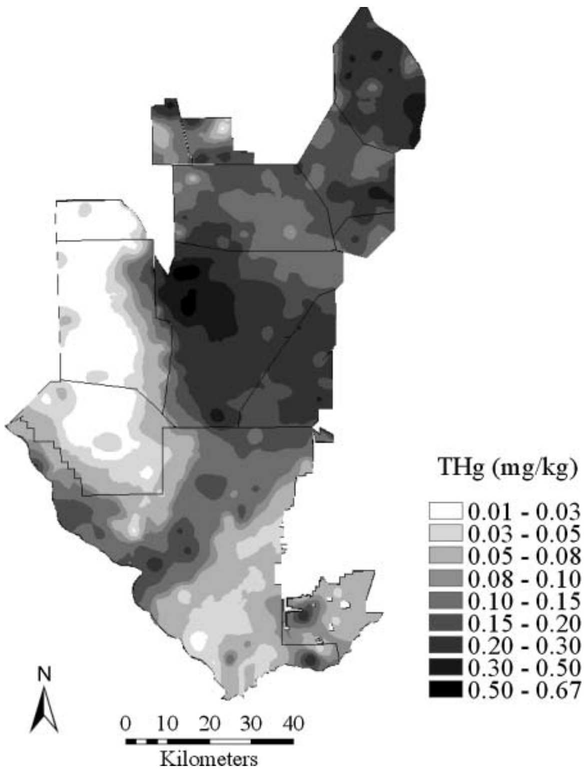

FIGURE 6. Spatial distribution of total mercury (THg; mg kg^{-1}) in the 0–10 cm soil profile. Map adapted from Cohen et al. (2009). Map by M. J. Cohen.

transforms Hg to MeHg, although still poorly understood, has been linked to SRB. It is believed that although using sulfate as an alternate electron acceptor in the absence of oxygen, SRB reduces a sulfate to sulfide (a sometimes toxic compound) and promotes the addition of a methyl group to Hg. Interestingly, the buildup of reduced sulfur compounds such as sulfide can bind Hg, making it unavailable for methylation (Drexler et al., 2002). Similarly, organic matter in soils or dissolved organic matter (DOM) can bind Hg, effectively reducing methylation potential. Drexler et al. (2002) reported that different Everglades peats and DOM had different affinities or binding capacities for Hg, suggesting high variability is possible across the EPA. In fact, the mercury Bioaccumulation Factor (BAF) within the Everglades marsh from the water column to prey fish is very high and varies by a factor of 10 across space. BAF is strongly correlated with dissolved organic carbon, and forms of sulfur including sulfide and sulfate (Scheidt and Kalla, 2007).

The excessive soil subsidence in northern WCA-3A reported by Scheidt and Kalla (2007) is just upstream from the highest observed soil THg observed by Cohen et al. (2009). This pattern suggests that remobilization from soil oxidation processes could be a very significant source for this particular area. Other significant inputs are likely relegated to aerial deposition as canal inputs are known to be low in Hg. Guentzel et al. (1998) reported that dry deposition to foliage is not washed off, but rather is absorbed into plant tissues and accumulated over the life of the plant and that this process has implications for relatively high soil concentrations in some areas. It is unknown at this time if there are depositional patterns that may explain some variability in the Everglades with respect to THg in soils. It has also been reported that freshly deposited Hg is methlyated more readily than older depositional Hg, suggesting that binding to organic substrates is likely an effective means of reducing methylation (Axelrad et al., 2008). Therefore, aerial deposition is not the most probable explanation for the observed hotspot in western WCA-3A.

In WCA-1, the southeastern corner of the unit has the highest concentration of THg in soils. This local enrichment area is not associated with P or S enrichment in WCA-1. The lack of correlation with P enrichment is not surprising (Cohen et al., 2009); however, the lack of sulfur enrichment may indicate that the means for methylation and thus mobilization of Hg from soils is not present. A similar trend is observed in the spatial distribution of THg in WCA-2A. Again, the enrichment is in the southeastern corner, and the highly impacted areas with respect to P are north and west of this zone. Interestingly, THg is highest in an area that is enriched with S, suggesting that S is not directly influencing THg storage. WCA-2A experiences drawdown and some soil oxidation in the northern and central portions during the dry season, as does WCA-1. Liu et al. (2008b) reported increases in Hg methylation in dry season conditions when sulfate is more readily available to SRB. This could potentially explain the concentration effect in deep-water zones of WCA-1 and WCA-2; however, this pattern is not observed in WCA-3A. The ENP does exhibit patterns that could be explained by this process as most of the THg is in soils of Shark River Slough, which has the longest hydroperiod of freshwater ENP lands. This area also contains the highest fish concentrations of MeHg (Scheidt and Kalla, 2007). This finding further supports that assertion as the lower hydroperiod areas surrounding Shark River Slough often dry out during low rainfall, providing an opportunity for methylation and therefore mobilization of Hg from the surrounding landscape.

4.5 Summary of Patterns and Concerns

Several landscape patterns identified in the studies reviewed here give reason for concern. Extensive areas in WCA-1 (Loxahatchee NWR) continue to be impacted by P and S enrichment even in light of recent improvements to water quality from the EAA. Similarly, WCA-2A appears to have some stabilization in soil P enrichment along the eutrophic gradient in the northern portion of the unit, however, recent comparisons indicate translocation of

TP and possible internal cycling. The significant effect of internal cycling of P is that of increased timeline for restoration and continued Typha expansion on the landscape. If internal loading and recycling of P continues, restoration success may be undetectable for some time, effectively masking positive actions to abate P enrichment. Similarly, as P is mineralized and translocated downstream, soil enrichment continues to provide avenues of Typha expansion. It is unknown how long it may take to move P through the EPA and return soil pools to preenrichment conditions. With similar results, sulfur enrichment in WCA-2A, combined with present sulfide distribution in surface soils suggests that extensive sulfate reduction is occurring in the eutrophic areas, further complicating P and Hg problems. Treating S similar to P with respect to control may be a necessary protocol to protect EPA from continued P mobilization and Hg methylation.

The extensive soil subsidence in WCA-3A, as evidenced by Scheidt and Kalla (2007), may be one of the most significant landscape-scale issues with respect to soil nutrients and contaminants. The process of soil oxidation is much more rapid than soil accretion; thus, soil oxidation has great impact on the surrounding landscape as nutrients and contaminants stored in these organic soils is mobilized, causing further exacerbation of eutrophication and contamination downstream. As P enrichment is significant in this area, subsidence here has great potential to increase this trend, along with the resultant expansion of Typha. Spatial patterns of S and Hg suggest extensive accumulation of both of these moieties south of the most severe soil subsidence. This could be an effect of previous subsidence and mobilization; however, the most relevant threat is remobilization via soil oxidation in these areas of concentration. Reversing organic soil oxidation in this area via hydrologic restoration must be a priority to maintain ecosystem health and integrity.

Finally, positive findings of minimal P and S enrichment in southern WCA-3 and ENP give credence to the prioritizing of thee areas to protect from future encroachment by excessive nutrients and contaminants. However, although soil concentrations of P, S, and especially Hg, are low in the ENP, recent evidence suggests accelerated methylmercury bioaccumulation in the ENP waters, a phenomenon unexplained by any soil landscape pattern observed by these studies. Again, hydrologic restoration may be key in preventing further subsidence related liberation of stored nutrients and contaminants upstream from ENP and southern WCA-3.

5 RELEVANCE TO RESTORATION AND FUTURE DIRECTIONS

The spatial studies discussed here have critical importance to the future of restoration efforts by providing a whole-system perspective, enabling assessment of condition and lending insight into interrelated factors at the landscape scale. These works not only provide the system-wide baseline necessary for future comparison and evaluation of restoration efforts, they enable identification of areas of concern, or hotspots, which require prioritization in the restoration effort. These hotspots are also areas in which positive change is most likely to be sought and observed. Concomitant with the identification of hotspots, system-wide monitoring allows for identification of areas of relatively low or no impact. These areas are likewise crucial in future restoration planning and assessment as they represent the frame of reference that should be used as restoration targets. Further, these areas should be prioritized to minimize potential impacts and monitored regularly to ensure that status.

As Everglades restoration moves forward, system-wide sampling efforts need to continue at some predetermined time interval to enable temporal assessment of change and identification of system-wide trends. Because monitoring at the ecosystem scale in the Everglades is a time- and cost-intensive endeavor, it is important to decide early which schedule is appropriate and plan accordingly. The cost of landscape sampling within the framework of restoration funding is somewhat small; however, with respect to research funding available for restoration, it is very significant indeed. Many questions concerning the ecology of the Everglades remain unanswered and therefore research continues to be an integral part of restoration efforts. System-wide monitoring may provide insight into relationships not conceived or suggested by localized studies. Stober et al. (2001) asserted that there is a significant need for a source of consistent scientifically credible information to assist decision making for Everglades restoration. Further, consistent long-term monitoring is the only way of evaluating the success of restoration efforts (Reddy et al., 2005; Scheidt and Kalla, 2007; Scheidt et al., 2000). Even though the expense of such large-scale sampling programs may be great, the cost of failing to move forward through adaptive management and assessment of restoration success could be even more costly in the future.

Toward successful strategies of attaining landscape-scale information, several key questions must be addressed. First, how often should this sampling occur? The REMAP and ESM programs use different sampling designs and time scales; however, both programs have been highly successful in identification of system-wide trends, local areas of concern, and comparative trajectory analyses that provide key information for restoration planning and evaluation. It is likely that both programs should proceed on their given schedules (REMAP in 5-year intervals, ESM in 10-year intervals); however, cross-program planning may be inherently beneficial to the value of information derived from these efforts.

Decisions concerning sampling strategies such as revisiting former sites (sentinel approach) or continually adopting new random sites have yet to be determined. Each utilizes a scientifically defendable approach; however, there is some doubt as to whether the sentinel approach is feasible considering high short-range variability and available GPS technology. Likewise, reasonable doubt exists concerning the level at which spatial variability can confound interpretations or detection of change if the sampling scale is coarse.

Assuring that field sampling methods across programs are consistent so that data are comparable is an issue that can be addressed by coordination across programs. This includes standardizing a method for the determination and collection of floc, the layer of unconsolidated detrital material above the soil surface. This floc layer is the most biologically active of the detrital pools and is of great significance when assessing nutrient and contaminant impacts.

Other issues surrounding future landscape-scale assessments include potential standardization of modeling strategies, the appropriate consideration of short range variability in evaluation of spatial models, and a uniform method of data analyses (mass vs. volumetric basis) of edaphic properties such that interpretation is more uniform (Bruland et al., 2007; Reddy et al., 2005; Scheidt and Kalla, 2007). Regardless of the decisions made concerning the aforementioned questions, it is certain that landscape-scale monitoring is crucial to assessment of ecosystem condition. Repeated landscape scale monitoring efforts undoubtedly are required to effectively determine change and thus restoration success in the Everglades.

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