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Incorporating science-based approaches into the rapid assessment of wetlands and streams : validation, restoration trajectory, and method development

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INCORPORATING SCIENCE-BASED APPROACHES INTO THE RAPID ASSESSMENT OF WETLANDS AND STREAMS: VALIDATION, RESTORATION TRAJECTORY, AND METHOD DEVELOPMENT

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This work is dedicated to my parents Nancy, Jerry, Allison, and Mike, as well as my wife Tanya whose patience and support made this dissertation possible.

Geaux Tigers.

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ABSTRACT

Human alterations within wetlands and streams have resulted in a decrease in ecological functions and associated benefits to society. The scientific literature highlights the functional benefits provided by ecosystems including flood protection, nutrient cycling, and habitat maintenance. Additionally, legislation and regulatory policy require mitigation and restoration as compensation for declines in ecological functions. As a result, the need for practical, repeatable, and technically sound ecosystem assessment methods remains essential to natural resource management. However, few studies determine the validity of rapid assessment approaches by applying quantitative parameters, especially with respect to biogeochemical functions. We assessed biogeochemical functions applied to restored wetlands in the Mississippi River Valley, USA. Significantly higher rapid assessment outcomes were associated with increased ecosystem functionality (r=0.64-0.86). Findings suggest that rapid assessment tools serve as reliable proxies for measurements of nutrient and biogeochemical cycling. Further, a framework for identifying restoration trajectory metrics was established, with four rapid assessment variables yielded positive restoration trajectories within <20 years (r = 0.59-0.89). Rapid assessment components were classified as rapid response, response, and stable variables categories and restoration milestones should focus on rapid response variables. In order to evaluate rapid ecological assessment in different environments, we examined proxy measures of biogeochemical function in headwater stream systems. Biogeochemical cycling proxies of C and N input and processing significantly, positively correlated with the results of a rapid assessment approach (r = 0.64-0.81). Also, stream loading equations demonstrate that N and P transport, sediment, conductivity, and temperature significantly, negatively correlated with rapid assessment scores (r = -0.56-0.81). Significant differences in nutrient processing, stream loading, water quality, and rapid

assessment results were also observed between headwater streams located in recently altered (e.g., mined) and older second growth forested catchments (U = 0.01-0.24). Findings indicate that rapid assessment scores respond to a combination of alteration type and recovery time. An analysis examining the time and economic requirements of biogeochemical proxy measurements highlights the benefits of rapid assessment methods in evaluating biogeochemical functions. Based on these findings, a technical standard for rapid ecological assessment was developed. The technical standard establishes nine testable components that promote validity and defensibility in the development and application of rapid ecological assessment approaches.

CHAPTER 1: INTRODUCTION AND REVIEW OF WETLAND AND STREAM RAPID ECOLOGICAL ASSESSMENT APPROACHES: METHOD DEVELOPMENT, APPLICATION, AND REGIONAL COVERAGE

1.1 Abstract

Ecosystems provide a number of functions, processes, goods, and values beneficial to society, and the need for reliable assessment of ecological function forms the cornerstone of effective natural resource management. Historical alterations within wetland and stream habitats have resulted in the degradation of aquatic ecosystems and a decrease in ecological function. Further, monitoring, regulation, and restoration of wetlands and streams requires ecological assessment, with rapid assessment approaches evolving of the past three decades. The components of rapid ecological assessment approaches are introduced, including the development of a conceptual diagram linking rapid assessment measurements with functions and ecological integrity. We examine 62 wetland and stream rapid ecological assessments currently applied in the United States and investigate key elements of 1) method development, 2) application, 3) regional coverage, and 4) method modification. Method development elements include the makeup of development groups, whether the method utilizes a classification scheme, whether the method assesses ecological function or condition, the basis for development, and whether the method underwent independent peer review. Method application examines the type of data collected (e.g., quantitative or qualitative), whether an onsite visit is required, the objectivity of assessment protocols, and the time required to complete the assessment. The regions covered by assessment methods and challenges to method modification are also addressed. Results demonstrate the need for development of a technical standard promoting science-based, defensible rapid ecological

assessments, expanded geographical coverage, and further training of natural resource professionals.

1.2. Introduction

1.2.1. Context and key findings

Natural ecosystems exhibit a variety of characteristic processes related to ecological service, goods, and values beneficial to society (UK National Ecological Assessment (2011; Figure 1.1). The need for reliable assessment tools capable of characterizing ecosystem functions and processes remains an essential component of resource management, forming the basis of ecosystem management within the larger context highlighted below (Figure 1.1).

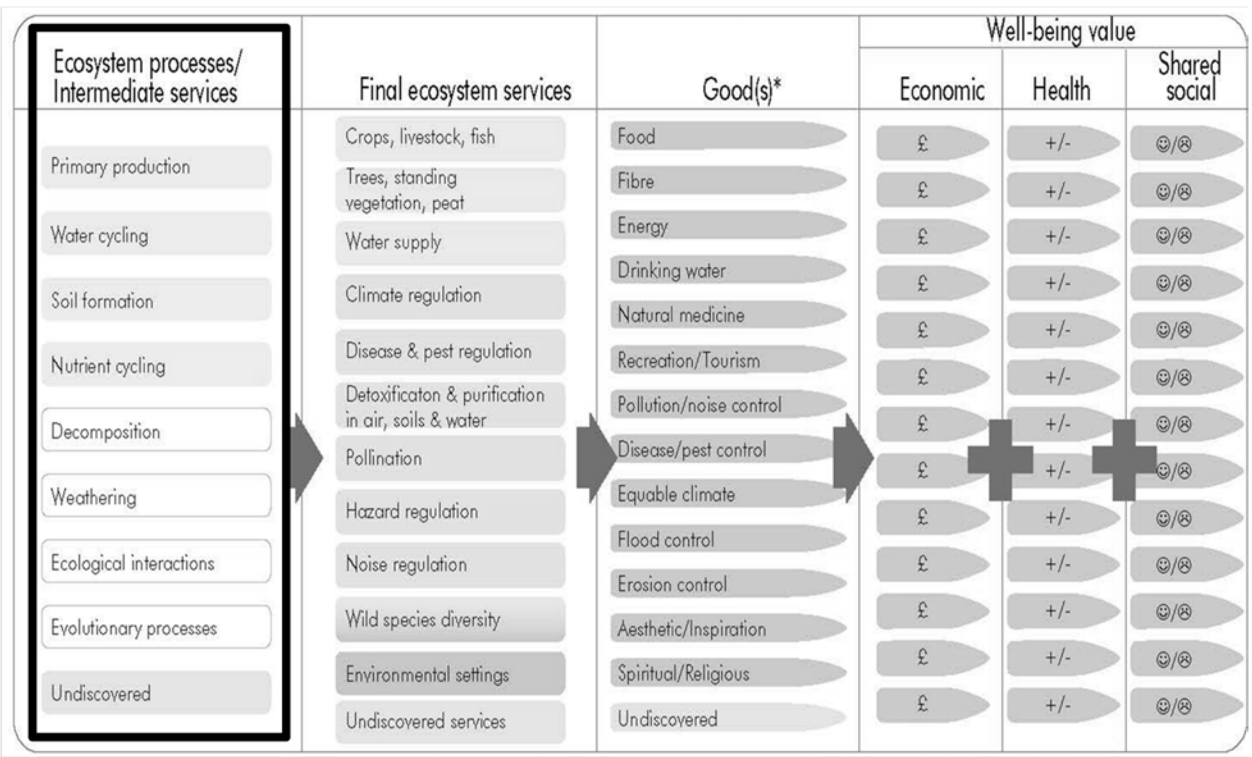

Figure 1.1. A schematic of the United Kingdom National Ecological Assessment outlining the relationships between ecological processes, services, goods, and values. The black box highlights the area of research examined. Modified from UK National Ecosystem Assessment (2011).

As a result, the dissertation research examines the validity of several ecological assessment approaches, develops a framework for identifying useful ecosystem evaluation tools, and proposes a technical standard for improving approaches for assessing ecological systems.

Key findings include:

- Nutrient cycling rapid assessment outcomes significantly correlated with measurements of total carbon, nitrogen, and microbial biomass carbon (r = 0.64-0.86; p = <0.001-0.17).
- Export of organic carbon rapid assessment outcomes significantly correlated with measurements of total organic carbon, loss on ignition, and onsite hydrology (r = 0.79-0.80; p = <0.001).
- Improve water quality rapid assessment outcomes significantly correlated with measurements of microbial biomass carbon, potentially mineraizable nitrogen, and onsite hydrology (r = 0.72-0.84; p = 0.005-0.29).
- Biogeochemical rapid assessment outcomes significantly correlated with measurements of leaf litter carbon and nitrogen inputs (r = 0.78-0.81; p = 0.005-0.008).
- Biogeochemical rapid assessment outcomes significantly correlated with measurements of leaf litter carbon and nitrogen processing (r = 0.67-0.81; p = 0.002-0.016).
- Biogeochemical rapid assessment outcomes significantly, negatively correlated with measurements of stream nitrogen and phosphorous loading as well as sediment, conductivity, and temperature water quality parameters (r = -0.56-0.81; p = 0.002-0.026).
- Rapid ecological assessments were capable of differentiating between intact forested study areas and recently disturbed areas based on measurements of carbon and nitrogen processing, phosphorous loading, and sediment, conductivity, and temperature water quality parameters (U = 0.01-0.24).

- Rapid assessment components can be categorized into rapid response, response, and stable variables categories and restoration milestones should focus on rapid response variables. Rapid response variables can be used to develop restoration trajectories.
- An analysis examining the time and economic requirements of biogeochemical proxy measurements highlights the benefits of rapid assessment methods in evaluating biogeochemical functions.
- The findings outlined above were utilized in the development of a technical standard for rapid ecological assessment was developed. The technical standard establishes nine testable components that ensure validity and defensibility in the development and application of rapid ecological assessment approaches.

1.2.2. Wetland and stream ecological functions

Historical human alterations have resulted in negative impacts to wetlands, with a 53% decrease in the original wetland area within the lower 48 states (Dahl, 1990; Dahl and Johnson, 1991). Additionally, the US Environmental Protection Agency (USEPA, 2013) reports that 55% of streams and waterways exhibit degraded or poor condition following decades of anthropogenic disturbance. The environmental movements of the 1960s and 1970s initiated a growing awareness of the benefits wetlands and streams provide to society (The Conservation Foundation, 1988). These include ecological, social, and economic benefits (Clairain, 2002). Traditionally considered important habitat for waterfowl species (Low, 1941; Munro, 1949; Courcelles and Bedard, 1978), aquatic ecosystems began to receive recognition as important for other fish and wildlife species (Ohmart and Anderson, 1978; Hendrix and Loftus, 2000). In addition to habitat, investigators began documenting that intact wetlands and streams reduce flooding by retention of floodwaters (Carter et al., 1978; Verry and Boelter, 1978), improve

water quality (Kibby, 1978; Blahnik and Day, 2000) and retain large amounts of sediments (Boto and Patrick, 1979). Further, Lee et al. (1978) described the retention of heavy metals by wetlands, while Nixon and Lee (1986) and Hiley (1995) examined nutrient cycling benefits in wetlands and streams.

The benefits provided by wetlands and streams have been identified as functions, broadly defined as the processes and manifestations of processes that occur in aquatic ecosystems (National Research Council, 1995). More specifically, ecological functions comprise processes and outcomes maintaining environmental quality (Montgomery et al., 2001). Examples of functions include: floodwater retention, nutrient cycling, maintenance of plant and animal communities, and removal of elements and compounds (Brinson et al., 1998; Hollands and Magee, 1985). The National Research Council (NRC; 1995) provides a general overview of ecological functions, common examples of functions separating into hydrologic, biogeochemical, and habitat categories (Table 1.1).

Table 1.1. Common wetland and stream functions (adapted from NRC 1995).

Function	**Effect**
Hydrologic functions	
Surface water storage	Reduced downstream flooding
Maintenance of high water tables	Maintains hydrophytic plant communities
Biogeochemical functions	
Elemental cycling	Maintains nutrient stocks
Retention of dissolved particles	Reduced transport of nutrients downstream
Improve water quality	Immobilization of toxic compounds
Habitat support functions	
Maintenance of plant communities	Food, nesting, and cover for animals
Maintenance of energy flow	Supports vertebrate population

For example, biogeochemical cycling represents a suite of functions performed by all aquatic ecosystems (Mitsch and Gosselink, 2007). Examining biogeochemical cycling functions demonstrates that soil organic carbon stocks provide for maintenance of plant communities

(Bormann and Likens, 1970; Whittaker, 1975; Perry, 1994), allowing plant communities (producers) to supply the food and habitat structure needed to maintain animal communities (consumers) (Fredrickson, 1978). As time progresses, plant and animal matter convert to detritus; supporting decomposers who break down organic materials into simpler elements and compounds that reenter the cycle (Reiners, 1972; Dickinson and Pugh, 1974; Schlesinger, 1977; Singh and Gupta, 1977; Hayes, 1979; Harmon et al., 1986; Vogt et al,. 1986). Thus, biogeochemical cycling functions represent beneficial processes occurring in aquatic ecosystems, involving producers, consumers, and decomposers (Nobel et al., 2010). Disturbance of wetland and stream ecosystems causes degradation or complete loss of these functions (Johnston, 1994) and continued human development pressure has resulted in regulation and permitting procedures for activities impacting ecosystem functions (Cole, 2006; Ainslie, 1994).

During the 1970s and 1980s the United States Fish and Wildlife Service (USFWS) along with other federal, state, local and private organizations realized the negative impacts of wetland functional degradation and began promoting aquatic ecosystem conservation and restoration efforts (US Congress, 1985; Haynes et al., 1995). Additionally, the Clean Water Act (Environmental Laboratory, 1987) requires mitigation for permitted impacts to wetlands and other waters of the United States. For example, section 33 U.S.C. §1251 requires that ecosystem restoration efforts result in a "…balanced integrated, adaptive community of organisms having a species composition, diversity, and *functional* organization comparable to the natural habitat of the region" (Karr and Dudley, 1981; USEPA, 2002). Further, Ebersole et al. (1997) and Moerke and Lamberti (2004) state that following decades of human alteration restoration project goals should focus on returning ecosystems to "a condition that promotes re-expression of natural ecosystem structure and function." As a result, both federal law and the available scientific

literature emphasize the need to restore functionality within impacted areas, which requires that resource managers assess (i.e., evaluate the nature) of ecological function within aquatic environments.

The need to assess and quantify ecological function remains rooted in section 404 of the Federal Water Pollution Control Act of 1972 and the ‘no net loss’ policies implemented during the 1980s (USC, 2011; Carletti et al., 2004). The 2008 Mitigation Rule further outlines the requirement that aquatic resource functional losses be offset by mitigation (USACE and USEPA, 2008). Requirements of the Clean Water Act and the Mitigation Rule include characterization of baseline information, consistent mitigation credit determination methodology, ecological performance standards, and monitoring; all of which necessitate ecosystem assessment (USEPA, 2006).

1.2.3. Ecological assessment techniques

Kolkwitz and Masson (1908; 1909) developed the saprobic system, the first biotic index used for ecological assessment (Sharma and Sharma, 2010). The saprobic system inferred stream water quality and the degree of organic waste pollution within aquatic systems by examining indicator plant species. Since that time a number of ecological assessment methodologies have developed to evaluate the capacity of a habitat to perform ecological functions and measure or infer ecosystem performance (Collins et al., 2008; Zedler, 1996). Ecological assessments determine functional gains or losses of potential impacts to an aquatic ecosystem and monitor wetland or stream condition over time (Montgomery et al., 2001). The ideal approach for assessing ecological functions involves direct measurements of wetland and stream functionality and functional rates over a period of several years (Brinson et al., 1984; Harmon et al., 1986). However time, technical expertise, and funding limitations restrict resource managers’ ability to

conduct assessments involving direct functional measurements (Smith et al., 1995). Thus, assessment techniques must provide adequate sensitivity capable of detecting changes in wetland function, while adhering to budget, time, geographical, and other constraints (Stein and Ambrose, 1998). Sutula et al. (2006) outlines the relationship between assessment scale and intensity for various ecological assessment approaches (Figure 1.2). It should be noted that the degree of intensity associated with rapid ecological assessments ranges from qualitative to quantitative, and Sutula et al. (2006) provides a discussion of the role of qualitative and quantitative measures within ecological assessment.

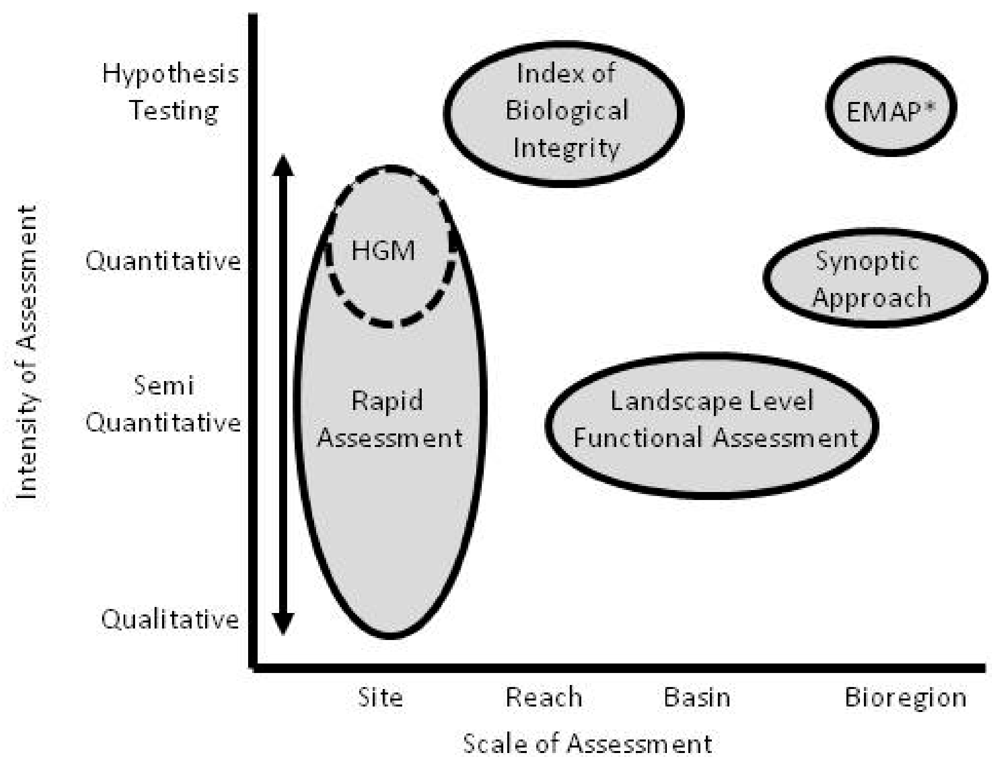

Figure 1.2. Example of ecological assessments methods with respect to the geographical scale and sampling intensity required. Modified from Sutula et al. (2006). *Environmental Monitoring and Assessment Protocol (EMAP).

Smith et al. (1995) and Fennessy et al. (2007) developed conceptual frameworks relating individual structural components and processes with various levels of ecological function [Figure 1.3(A)]. We further the discussion by outlining a five level hierarchical model of ecosystem function by linking 1) structural components with 2) processes, 3) functions, 4) integrated suites of functions (hydrology, habitat, and biogeochemical cycling), and 5) ecological integrity [Figure 1.3(B)]. Ecological integrity is defined as "the function that encompasses all ecosystem structures and processes in an ecosystem" (Fennessy et al., 2004).

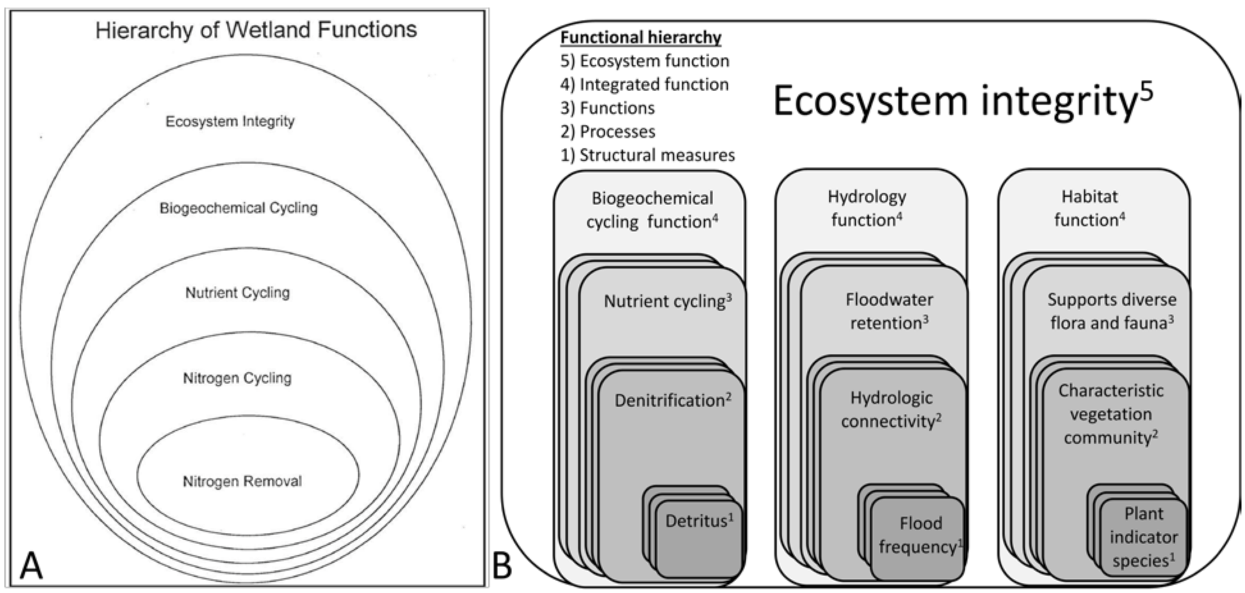

Figure 1.3. A) Conceptual diagram demonstrating the linkages between processes and various levels of ecological function (Smith et al., 1995), B) Updated conceptual model demonstrating the relationship between structural measures, processes, functions, integrated function, and ecological integrity.

1.2.4. Rapid ecological assessment

As a result of the budgetary and time constraints outlined above, rapid assessment techniques evolved to infer wetland and stream functions based on structural indicators corresponding to wetland condition and function (Van Dam et al., 1998; Mack, 2001). The development of rapid ecological assessments represents a similar approach to the original work

outlined by Kolkwitz and Marsson (1908; 1909). Rapid assessments exhibit a wide range of intensity (Figure 1.2) between methodologies based on quantitative and qualitative data. In ecological assessment approaches utilizing quantitative data collections, the measurements of structural ecosystem components related to ecological function have proven efficient and repeatable (Berkowitz et al. 2011). Further, several studies successfully link structural measures with wetland function as observed in conceptual diagrams (Bohonak and Bauder, 2011; Hill et al., 2006; Stein et al., 2009; Figure 1.3).

The USEPA (2004) defines rapid as "as taking no more than two people a half day in the field and requiring no more than a half day of office preparation and data analysis to come to an answer." In response, a large number of the rapid assessment methods currently in use employ environmental indices (Smith et al., 2006; Potter et al., 2006; Gregorich et al., 2005). Rapid assessment methods utilizing functional indices provide a compromise between ease of use and comprehensiveness by reducing information requirements while retaining the most essential information (Ott, 1978).

While rapid assessment approaches display a wide variety of characteristics, each assessment method can be described and categorized based on a few key components (Carletti et al., 2004). Notably, elements of 1) assessment approach development, 2) assessment application (i.e., protocol), 3) geographical applicability, and 4) the degree of method modification are useful in comparing across methods and defining recommendations for improving rapid ecosystem assessment strategies.

1.2.4.1. Assessment approach development. Assessment development includes the individuals involved in drafting the assessment approach, determining the goals of the approach, how ecosystem types are addressed within the approach (i.e., classification), the basis for

assessment scaling, and what level of technical peer review was required prior to implementation. Assessment method developers range from individual resource managers to teams of ecological experts working together. Clairain (2002) recommends the formation of an interagency, interdisciplinary team of ecological experts in the development of assessment approaches. The size and make-up of the development team impacts assessment method outcomes, with more experienced and interdisciplinary teams capable of producing methods applicable to larger regional scales across multiple ecosystem types (Smith et al., 2013).

Assessment developers establish the goals of the assessment method including whether the document is intended to evaluate ecosystem condition or function. Ecosystem condition is defined in the Mitigation Rule as the relative ability of an aquatic resource to support and maintain a community of organisms having a species composition, diversity, and functional organization comparable to reference aquatic resources in the region (Federal Register, 2008). As described previously, ecosystem functions include the physical, chemical, and biological processes that occur in ecosystems (Smith et al., 1995). Assessment approaches examining condition include the California Rapid Assessment Method (CRAM) which evaluates aquatic resource characteristics (e.g., plant or animal communities), while the Hydrogeomorphic (HGM) approach focuses on determining individual ecosystem functions (e.g., biogeochemical cycling) (CWMW, 2012; Noble et al., 2010). Many assessment approaches provide a mixture of conditional and functional measures.

The classification of ecosystems has also been identified as critical in any assessment approach (Karr and Chu, 2006; 1997; Stoddard et al., 2006). Ecosystems display variability within the same natural region (e.g. a forested floodplain vs. a tidal marsh) and classification reduces the effect of variability on an assessment method's output; increasing the ability to

discern differences among individual sites (Stein et al., 2009; 2009b). Ecosystem classification also increases the accuracy and repeatability of assessment methods by defining the target ecosystem (Wharton, 1978; Smith et al., 1995), which enables in-kind comparisons required by the Mitigation Rule and other policies or legislation. In streams, common classification schemes include stream order, flow regime (e.g., perennial), and the approaches outlined by Dave Rosgen (Wildland Hydrology, Inc.) (Noble et al., 2010; Rosgen, 1996). Wetland classification systems include vegetation, hydrologic, and geomorphic characteristics (Cowardin et al., 1979; Brinson, 1993).

The calibration (i.e., standard of comparison) developed for an assessment methods also impacts assessment outcomes. Common calibration approaches include the comparison of assessment method scores with scores generated in reference areas representing undisturbed, natural conditions (Hughes et al., 1986; Brinson, 1993; Smith et al., 1995). Wigham (1999) supports the use of reference based assessment approaches for promoting efficiency, consistency, and establishing a framework for comparison between impacted, unimpacted, and potential mitigation areas. Other approaches employ calibrations derived from available literature or from the opinions of resource professionals (i.e., best professional judgment [BPJ]) (USACE, 1999). Assessment developers often employ a combination of BPJ, available literature, and reference data.

Some assessment methods incorporate peer review into the development process, while other methods do not undergo review. Clairain (2002) describes a detailed procedure for distributing assessment methods to independent peer reviewers and incorporating reviewer recommendations into the assessment approach.

1.2.4.2. Assessment method application and protocol. An assessment method protocol determines the type of data collected, the type of output the method generates, the time necessary to complete an assessment, the type of skills and training required, and whether on-site or off-site information is required. Within ecological assessments both quantitative and qualitative data streams are common. Quantitative data includes repeatable measurements and estimates that generate numerical values (Berkowitz et al., 2011). Wetland assessment protocols often require quantification of the amount of ground cover, the shrub density, or the average tree diameter within a defined area (Smith and Klimas, 2002). Common quantitative measures applied in stream assessments include measurements of width:depth ratios and measurements of substrate particle size (Rosgen, 1996).

Qualitative variables employ narrative statements describing an ecosystem impact or condition based on a visual estimate of ecosystem structure or the presence/absence of ecological stressors. Although qualitative approaches increase rapidity, they often lack sensitivity (Klimas, 2008). Many assessment methods contain a combination of quantitative and qualitative elements.

The assessment protocol also dictates the types of output generated by each method. Assessment methods produce numerical (i.e., continuous) or categorical output types. Numerical outputs include scoring mechanisms in which method outputs appear on a continuous range. For example, many assessment approaches vary scoring between 0.0 and 1.0, while others apply a scale between 1.0 and 10 (Smith and Klimas, 2002). Categorical outputs most often describe a wetland or stream as “Good”, “Fair”, or “Poor.” In other cases, assessment approaches describe an ecosystem functional or conditional status as “High”, “Medium”, or “Low.” Fennessy et al. (2004) suggests that categorizing assessment components dampens variability resulting in a more robust method. However, others argue that utilizing categories increases user bias, potentially

decreasing accuracy (Klimas, 2008). Additionally, in cases where multiple ecosystem functions or conditions are evaluated, some methods combine individual functional or conditional scores into a single value (Fennessy et al., 2007). This approach simplifies the comparison between ecosystems. However, combining scores limits the ability to target individual conditional or functional elements, increases uncertainty, and makes method validation (i.e., determining method accuracy by applying independent measures) more difficult (Fennessy et al., 2004).

Assessment protocols also impact the efficiency and practicability of methods through time requirements and the need for onsite visits. Approaches necessitating a large number of field measurements or time consuming analysis are beyond the scope of a rapid assessment approach, which should require a half day or less of field work and a half day or less of office preparation and analysis (Smith et al., 1995; Berkowitz et al., 2011).

1.2.4.3. Geographic extent. Assessment approaches also differ in the geographic extent covered by the method. The geographic extent of assessment methods includes approaches designed based on ecological or geophysical regions (e.g., watershed, ecoregion, Land Resource Region) and approaches restricted to a geopolitical area (e.g., state, county).

1.2.4.4. Assessment method modification. Recent debate has resulted from the modification of assessment methods and the application of modified approaches to expanded geographic regions or ecosystem types. The debate stems from the potential application of new requirements for permit applicants, potential changes to mitigation requirements, or a lack of familiarity with assessment methods. In some cases a decreased level of documentation is utilized when assessment modification occurs. For example, initial assessment method development may include an interagency, interdisciplinary team of ecological experts, field data collection, method protocol testing, peer review, and other steps documenting the decision

making process. Similar approaches are not always clearly applied to modified assessment methods or the decision making process is not well documented. However, opportunities exist for modification and geographical expansion of current assessment methods. For example, the assessment approach developed by Noble et al. (2010) for headwater streams in western WV and eastern KY is undergoing expansion into surrounding areas. The expansion process is based on collection of quantitative data, testing of field data collection protocols, and well documented adjustment of the assessment method based on data.

1.3. Methods

The data presented below was derived from two sources including 1) available published literature and 2) a survey of natural resource professionals. The current work builds upon the works of Bartoldus (1999) who identified 40 assessment methods developed prior to 1998 (a subset of which were rapid) and Fennessy et al. (2007) who provided data for 16 rapid assessment methods developed before 2003. In order to account for recently developed rapid wetland and stream assessment approaches, a nationwide survey was sent to natural resource professionals employed by the US Army Corps of Engineers (USACE) during the summer of 2012 (Berkowitz and Wilder, In Preparation). Survey results and findings of the literature review were combined, resulting in a dataset containing 37 different wetland rapid assessment methods and 25 different stream rapid assessment methods currently in use (Tables 1.2 and 1.3). Due to multiple rapid assessments in use within a single state or geographical region, the data presented in the results and discussion section, may exceed the number of assessments examined. For example, because the Ohio Rapid Assessment Method (ORAM; Mack, 2001) is applied in Ohio, Florida, New York, and Pennsylvania, multiple respondents provided data for that method. Results are organized in four sections addressing 1) assessment method development, 2)

assessment protocol, 3) geographical extent and 4) method modification. The current dataset is not intended to include all potential wetland and stream rapid assessment methods in use, but seeks to provide an overview of the type of approaches commonly applied.

1.4. Results

1.4.1. Assessment method development

As outlined above, assessment method development includes the individuals involved in drafting the assessment approach, determining the goals of the approach, how ecosystem types are addressed within the approach, the basis for assessment scaling, and what level of technical peer review was required prior to assessment implementation. Results indicate that interagency groups contributed to development of 50% and 56% of assessment methods in wetlands and streams respectively (Figure 1.4). Some rapid assessment approaches in use were developed by academia, state agencies, or by an independent agency. Assessment method developers can ensure that assessment results coincide with specific resource management obligations. For example, the HGM approach was developed specifically to address the needs and requirements of the USACE regulatory program, including permitting and mitigation activities (Brinson, 1993; Ainslie, 1994). The inclusion of multiple stakeholders into assessment development encourages buy-in and application of the assessment approach across large geographic regions (Clarian, 2002).

Rapid assessment methods are designed to determine the level of ecosystem function, condition, or a mixture of function and condition (Figure 1.5). In wetlands, 24% of the methods assessment approaches utilized a measure of function, and 24% measure condition. 39% of assessment methods contained both functional and conditional elements.

Table 1.2. Wetland rapid assessment methods currently in use, estimated application area, and source.

Method	Region in use	Source
Anchorage wetlands assessment method	AK	Municipality of Anchorage (1996)
California rapid assessment method	CA	CWMW (2013)
Charleston method (2003)	MS, AR	Regulatory Branch (2002)
Charleston method (2010)	SC	Regulatory Branch (2010c)
Functional assessment of wetlands-Homer	AK	City of Homer (2006)
Calculating compensatory mitigation in wetlands of Washington	WA	Hruby (2012)
Delaware comprehensive assessment protocol	PA	Jacobs (2003)
Floristic quality index	WY, MT	Hauer et al. (2002b)
Functional assessment of Colorado wetlands	NM, CO	Johnson et al. (2011)
Functional assessment of Colorado Wetlands	CO	Johnson et al. (2011)
Modified HGM (interim HGM)	TX	Regulatory Branch (2010)
Highway methodology supplement (Descriptive Approach)	New England	Regulatory Branch (1999)
Hydrogeomorphic functional assessment - prairie potholes	WY, MT	Hauer et al. (2002)
Hydrogeomorphic functional assessment - vernal pools	CA	Bauder et al. (2009)
Hydrogeomorphic functional assessment - Yazoo Basin	MS	Smith and Klimas (2002)
Hydrogeomorphic functional assessment - Eastern Kentucky	KY	Noble et al. (2010)
Massachusetts Coastal Zone Management Method	MA	Hicks and Carlisle (1998)
MN routine assessment method for evaluating wetland functions	MN	Minnesota Board of Water and Soil Resources (2010)
Montana wetland assessment method	MT	PBS&J (2008)
NC wetland assessment method	NC	NC Wetland Functional Assessment Team (2010)
Modified Charleston method	LA	USACE New Orleans District (2012)
Ohio rapid assessment method	OH, PA, FL, NY	Mack (2001)
Pennsylvania function based aquatic resource protocol	PA	State of Pennsylvania (In Preparation)
Penn state stressor checklist	PA	Brooks et al. (2002)
Ratio method	TN	RIBITS (2013)
Standard operating procedure - Virginia	VA	USACE and VA DEQ (2004)

Table 1.2 continued

Method	Region in use	Source
Standard operating procedure - Georgia	GA	Regulatory Branch (2004)
Tennessee Valley Authority modified ORAM	TN	Awl (2004)
Texas rapid assessment method	TX	Regulatory Branch (2010b)
Uniform mitigation assessment methodology	FL	UMAM (2012)
Ratio method	AL	RIBITS (2013)
Washington functional assessment method	WA	Hruby et al. (1999)
Washington State wetland rating system	WA	Hruby (2004)
Wetland ecosystem services protocol for southeast Alaska	AK	Adamus (2010)
Wetland rapid assessment procedure - Alabama	AL	RIBITS (2013)
Wetland rapid assessment procedure - Florida	FL	SFWMD (1997)
Wisconsin rapid assessment method	WI	WI DNR (1992)

In streams, only 5% of assessment methods measured ecosystem function, while 24% were conditional measures and 45% combined both functional and conditional elements. As described above, conditions are defined as the relative ability of an aquatic resource to support and maintain a community of organisms having a species composition, diversity, and functional organization comparable to reference aquatic resources in the region (Federal Register, 2008).

While ecosystem functions include the physical, chemical, and biological processes that occur in ecosystems (Smith et al., 1995), several regulatory statutes specify the need to determine changes in function, resulting in the development of approaches (including HGM) that provide assessments of ecosystem function. In recent years the number of conditional approaches has increased, in part due to the perception that conditional assessment methods are more rapid than functional assessment approaches and because many conditional methods merge all scores into a single result (Wardrop et al., 2007; Fennessy et al., 2004).

Table 1.3. Stream rapid assessment methods currently in use, estimated application area, and source.

Assessment	Region in use	Source
2004 Standard operating procedure for compensatory mitigation	GA	Regulatory Branch (2004)
2010 Charleston stream assessment	MS	USACE Charleston District (2010)
California rapid assessment method for wetlands - riverine wetlands	CA	CWMW (2013)
Compensatory stream mitigation SOP and guidelines - Alabama	AL	Regulatory Branch (2012)
Compensatory stream mitigation SOP and guidelines - New York	NY	Regulatory Branch (In Preparation)
Field evaluation manual for Ohio's primary headwater streams	MN	Anderson (2009)
Illinois stream mitigation guidance	IL	Illinois DNR (2010)
Kansas stream mitigation guidance (KSMG)	KS	Mulder et al. (2010)
Little Rock stream method	AR	Regulatory Branch (2011b)
Louisville District stream assessment protocol (LDSAP)	KY	Kentucky DEP (2008)
Missouri stream mitigation method (MSMM)	MO	Missouri DNR (2007)
MnPCA stream habitat assessment	MN	Minnesota PCA (2002)
Mt NRCS riparian assessment	MT	USDA (2012)
NC stream assessment method	NC	NC Stream Functional Assessment Team (2013)
New Mexico rapid assessment method	NM, CO	Muldavin et al. (2011)
Pacific Northwest streamflow duration method	OR, WA, ID	Nadeau (2011)
Qualitative habitat evaluation index (QHEI)	PA	Rankin (1989)
Rapid bioassessment protocols for use in streams and rivers	USA	Barbour et al. (1999)
Rapid stream riparian assessment	AZ, NM, UT, CO	Stevens et al. (2005)
Stream standard operating procedure - Texas	TX	Regulatory Branch (2011)
Tennessee stream mitigation guidelines	TN	Tennessee DEC (2004)
Texas rapid assessment method (TXRAM)	TX	Regulatory Branch (2010b)
Unified stream methodology (USM)	VA	USACE and VA DEQ (2007)
Waterway assessment method for Anchorage	AK	Municipality of Anchorage (1996b)
West Virginia and Kentucky Hydrogeomorphic headwater stream assessment protocol	KY, WV, TN, OH,	Noble et al. (2010)

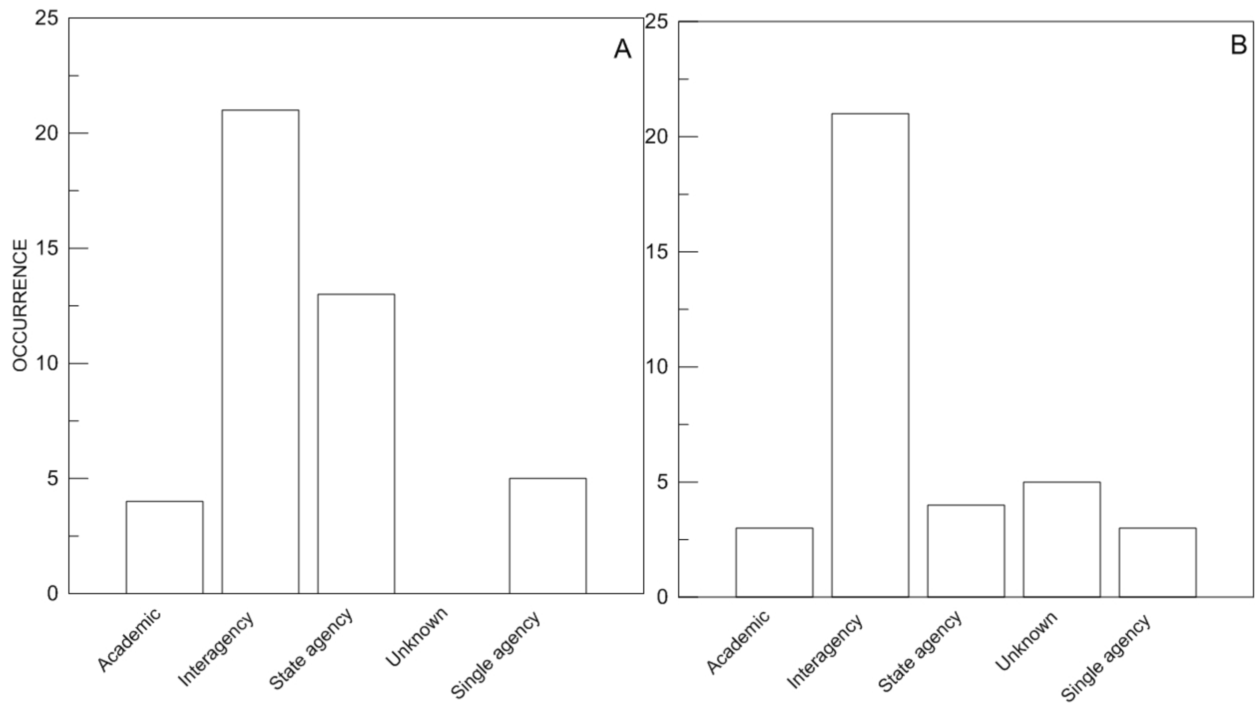

Figure 1.4. Summary of assessment method development participants in A) wetlands and B) streams.

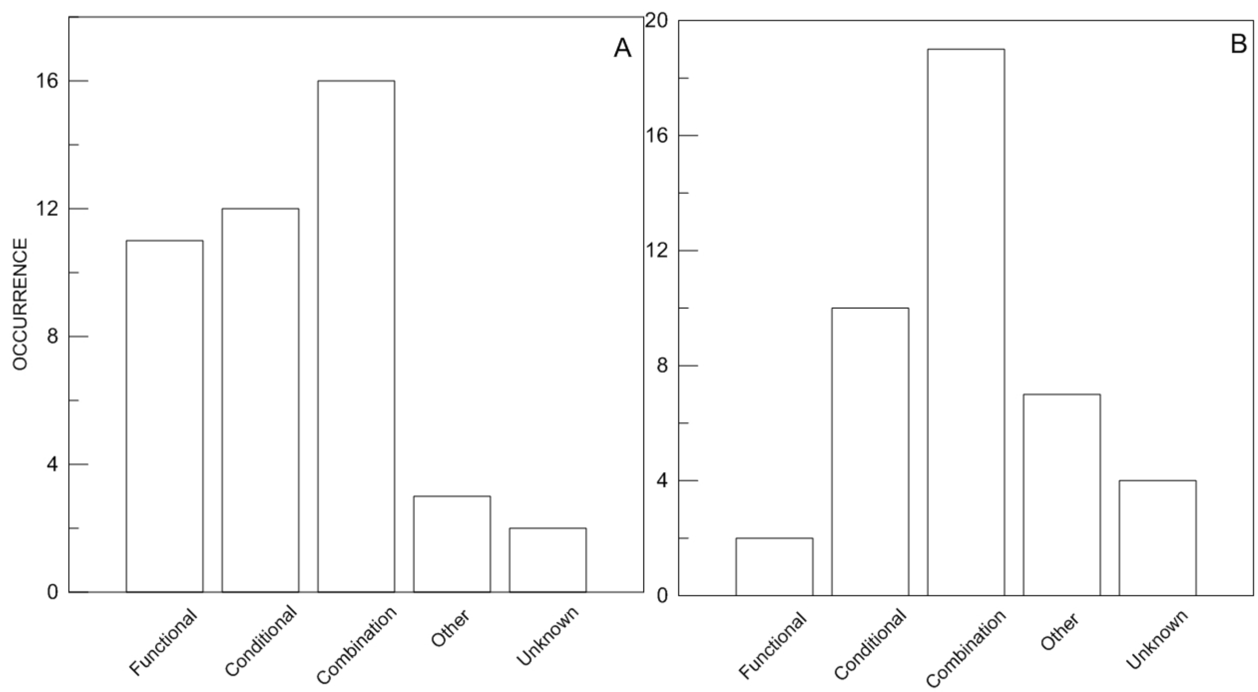

Figure 1.5. Summary of assessment methods addressing ecosystem function, condition, or a combination in A) wetlands and B) streams.

Classifying ecosystems by vegetation type, hydrologic regime, or other means focuses the scope of an assessment method, decreasing assessment variability while increasing accuracy and repeatability (Karr and Chu, 2006; Wharton, 1978; Rowe et al., 2009). Within the current dataset 67% and 73% of the assessment methods utilized applied ecosystem classification in wetlands and streams respectively (Figure 1.6). In wetlands, 35% of assessment methods applied an HGM classification (e.g., riverine, slope, depressional; Brinson, 1993). An equal number utilized a combination of classification approaches applying a mixture of vegetation, hydrology, and landscape position characteristics. The undefined category “Wetland type” was observed in 19% of cases, while the remaining methods utilized classifications based on vegetation or the USFWS system (e.g., palustrine shrub-scrub) developed by Cowardin et al. (1979).

In streams, classification schemes based on flow regimes such as perennial, intermittent, or ephemeral were applied in 57% of cases. Classification relying upon the Rosgen approach applied in 10% of assessments, which includes elements of slope, channel morphology, and sinuosity (Rosgen, 1996). Ten percent of rapid stream assessment methods applied a combination of classification approaches.

In addition to classification, assessment method calibration seeks to increase the accuracy and validity of an assessment approach. Available literature refers to developing a “standard of comparison” or “scoring” the assessment method (Sutula et al., 2006). Assessment methods must be calibrated to determine which ecosystem components prove beneficial and which components represent damaging characteristics or stressors (Fennessy et al., 2007). Smith et al. (1995; 2013) provides a discussion of assessment calibration, including the application of reference data, best professional judgment (BPJ), and other strategies.

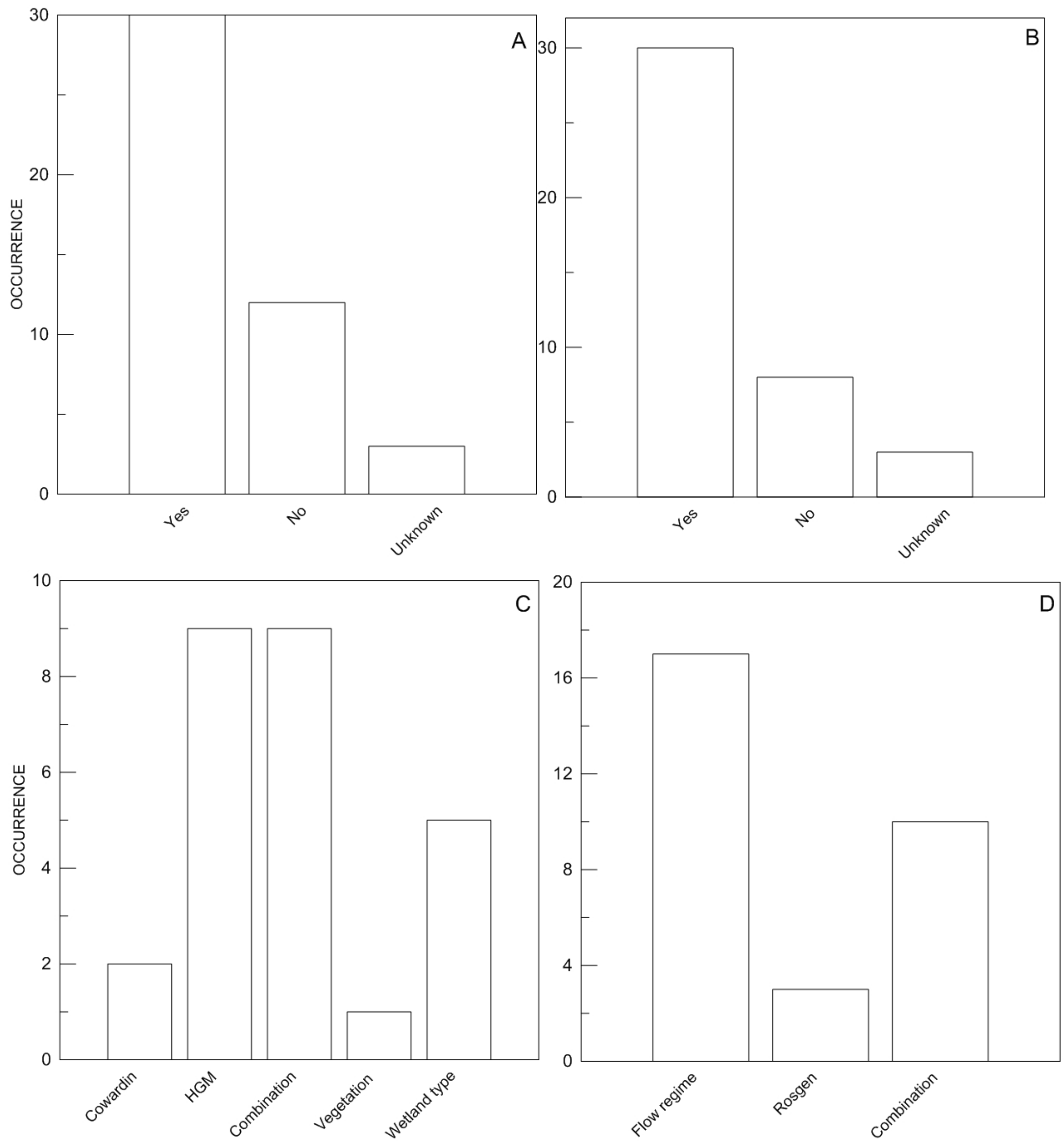

Figure 1.6. Summary of assessment methods addressing whether ecosystem classification was utilized in A) wetlands and B) streams and what type of classification was employed in C) wetlands and D) streams.

Literature review and survey results indicate that 22% and 12% of assessment methods rely on BPJ for calibration in wetlands and streams respectively (Figure 1.7). Additionally, 7%

of wetland assessments and 10% of stream assessment methods calibrated based upon a combination of BPJ and available literature sources. Wardrop et al. (2007) reports similar findings, describing a conditional assessment approach in which the criteria for condition categories were based on the literature or BPJ. In wetlands, 5% of approaches calibrated results with reference data only. Results demonstrated that reference data was applied in the calibration process to some degree in 54% and 59% of assessments conducted in wetlands and streams respectively. The majority of assessment methods preformed a calibration that utilized some reference data in combination with BPJ, and/or literature.

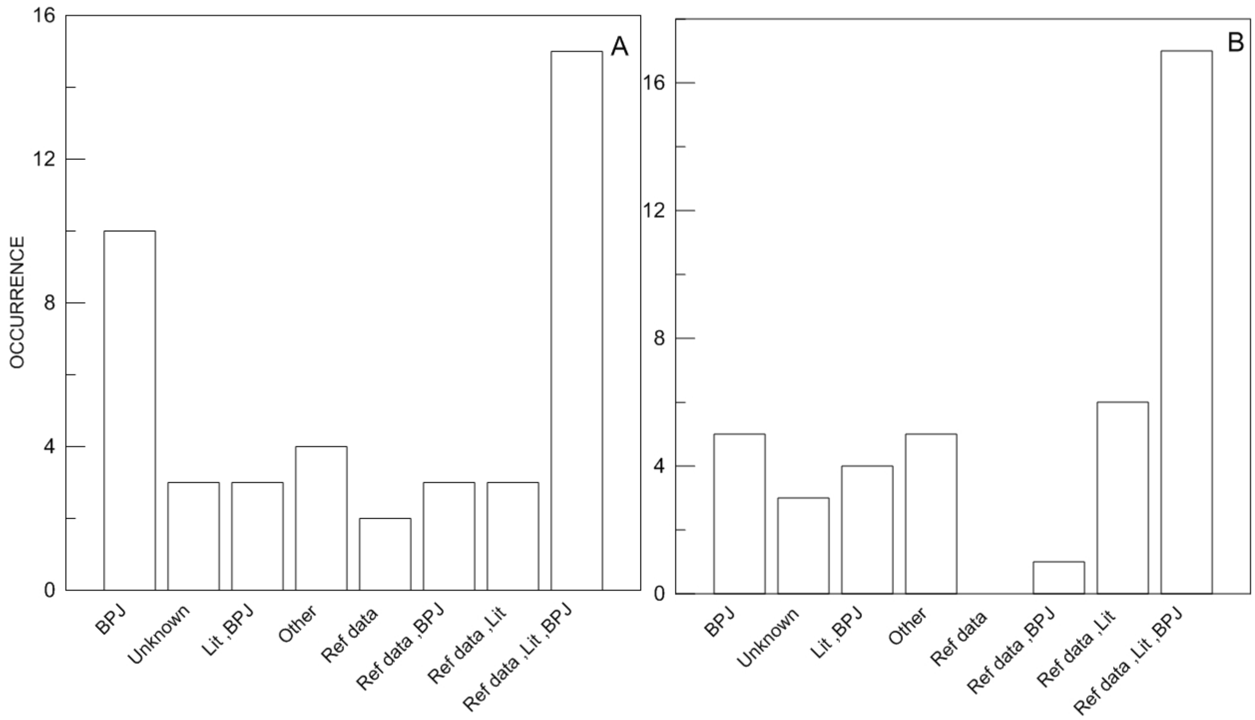

Figure 1.7. Summary of assessment methods calibration (i.e., standard of comparison) approach in A) wetlands and B) streams. Abbreviations include: Best Professional Judgment (BPJ), Reference data (REF), Literature review (Lit).

Incorporating peer review into the method development process provides technical feedback, expands the number of reviewers to ensure the technical validity and utility of the assessment methods, obtains recommendations for additional literature, and identifies potential

gaps in the assessment approach (Clairain, 2002). Peer review also encourages assessment method user buy-in and promotes defensibility (Smith et al., 2013). The current study reports 52% and 46% of the assessment methods examined received peer review in wetlands and streams respectively (Figure 1.8). No peer review was reported in 21% and 29% of cases and it remains unclear if peer review occurred in 26% and 24% of assessments addressing wetland and streams respectively.

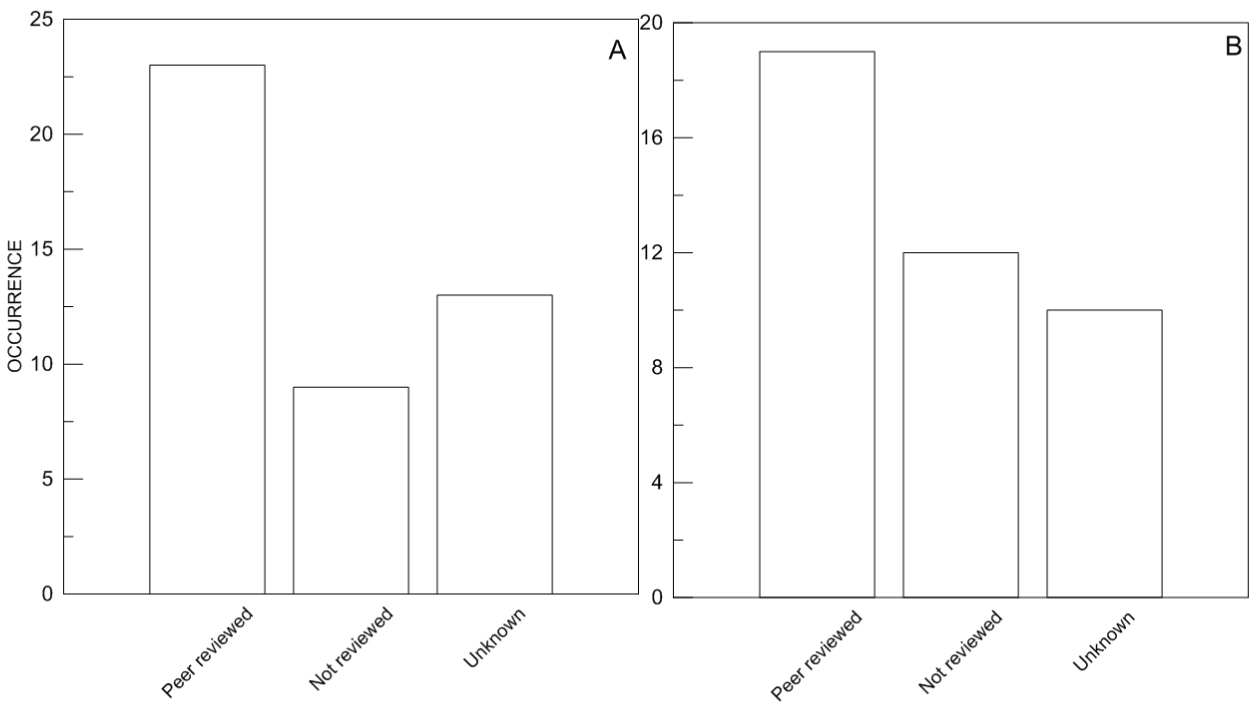

Figure 1.8. Summary of assessment methods receiving peer review in A) wetlands and B) streams.

1.4.2. Assessment method protocol

The objectivity of assessment results has been shown to impact assessment methods outcomes, with more objective measures associated with decreased variability (Berkowitz et al., 2011). Literature and survey results show 20% and 17% of the assessment methods applied objective measures in wetlands and streams respectively, while more assessments methods

utilized subjective measures (41% and 24% in wetlands and streams respectively; Figure 1.9). A combination of objective and subjective measures was utilized in 34% and 54% of assessment methods addressing wetlands and streams respectively.

Whether quantitative or qualitative data collection occurs as part of an assessment protocol is related to objectivity. For example, measuring the diameter of each tree within a defined area yields accurate and quantitative, repeatable results as shown in Berkowitz et al. (2011). Qualitative data utilized in assessment methods generally relies on narrative statements or the perceived presence or absence of ecological stressors (Fennessy et al., 2007; Wardrop et al., 2007). A direct measurement of tree diameter is objective, while narrative categories remain subjective with responses depending on user experience, perception, and training. As a result, Klimas (2008) notes that although qualitative approaches are designed to be rapid and repeatable, they lack sensitivity and potentially decrease accuracy. The current dataset demonstrates that 61% and 71% of rapid assessments utilized quantitative data in wetlands and streams respectively (Figure 1.10). Qualitative data was collected in 24% and 17% of approaches examining wetlands and streams respectively.

Offsite and onsite data are commonly used in ecological assessments (Smith and Klimas, 2002). Quantitative and qualitative data can both be generated onsite or using office based resources (e.g., GIS, aerial photos). However, the majority of rapid assessment methods require a field visit with 80% and 83% of assessment methods incorporating onsite data in wetlands and streams respectively (Figure 1.11). The need for a site visit adds to the amount of time required to complete the assessment, but allows for evaluation of both direct numerical measurements (e.g., tree diameter) and narrative statements or observation of ecological stressors (Wardrop et al., 2007).

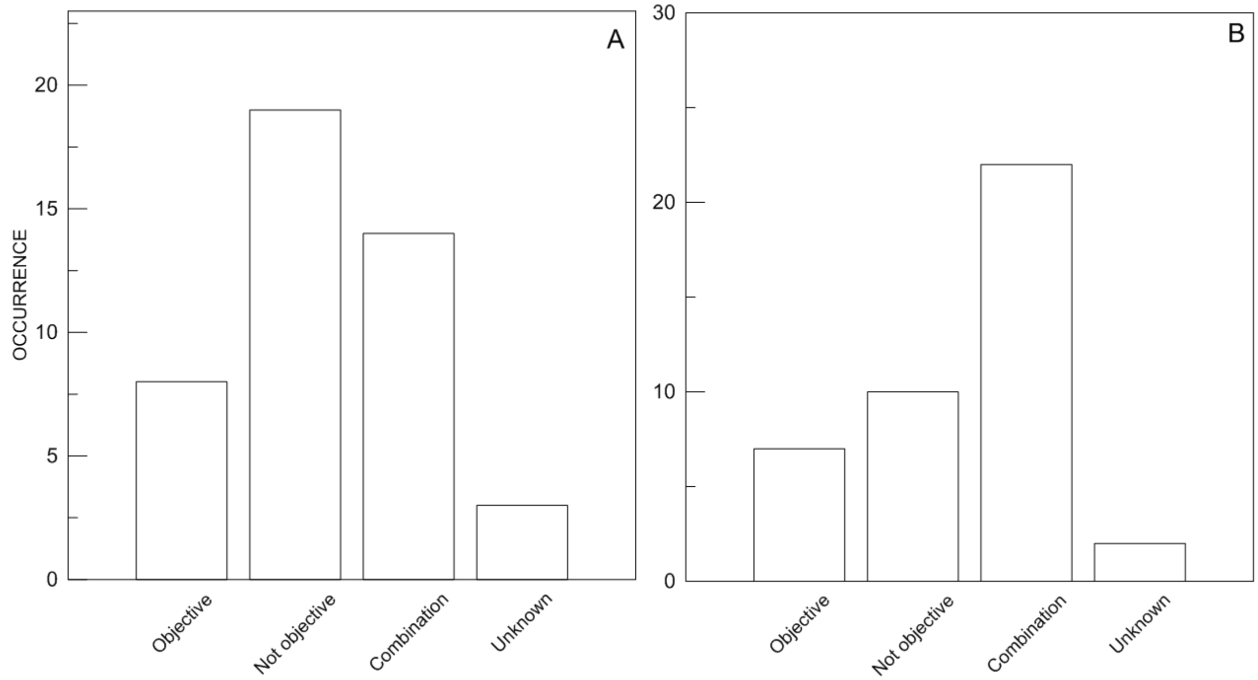

Figure 1.9. Summary of assessment method objectivity in A) wetlands and B) streams.

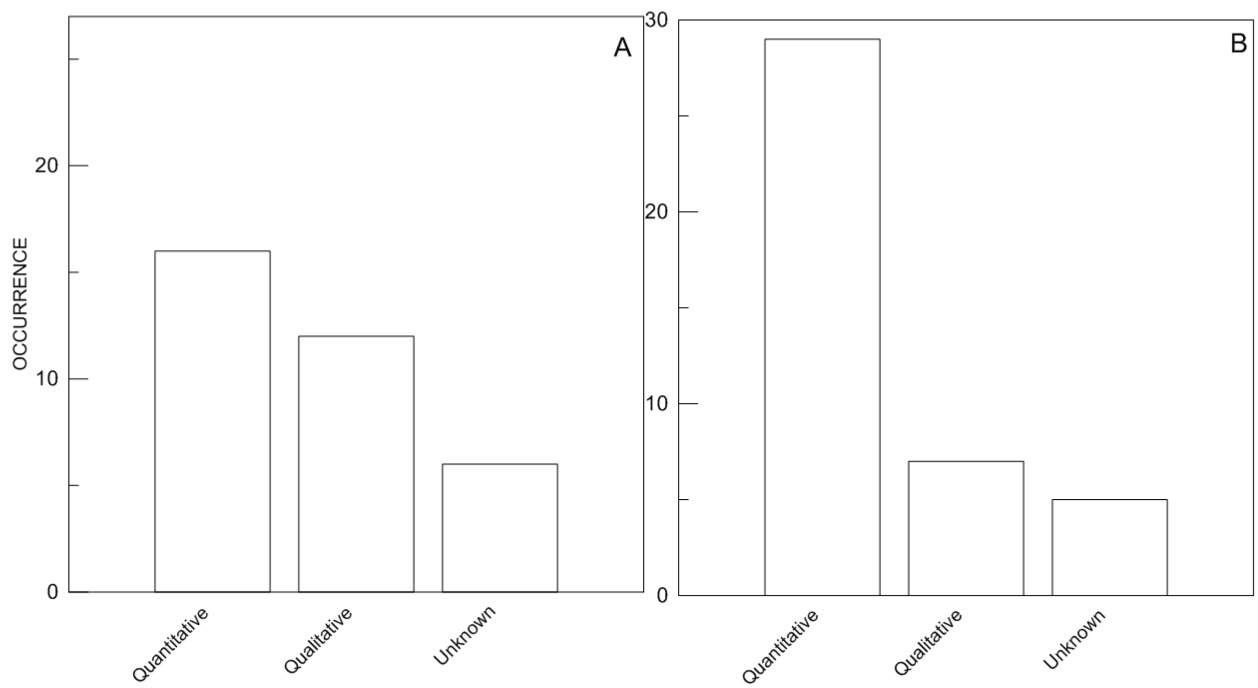

Figure 1.10. Summary of measurement type required by assessment methods in A) wetlands and B) streams.

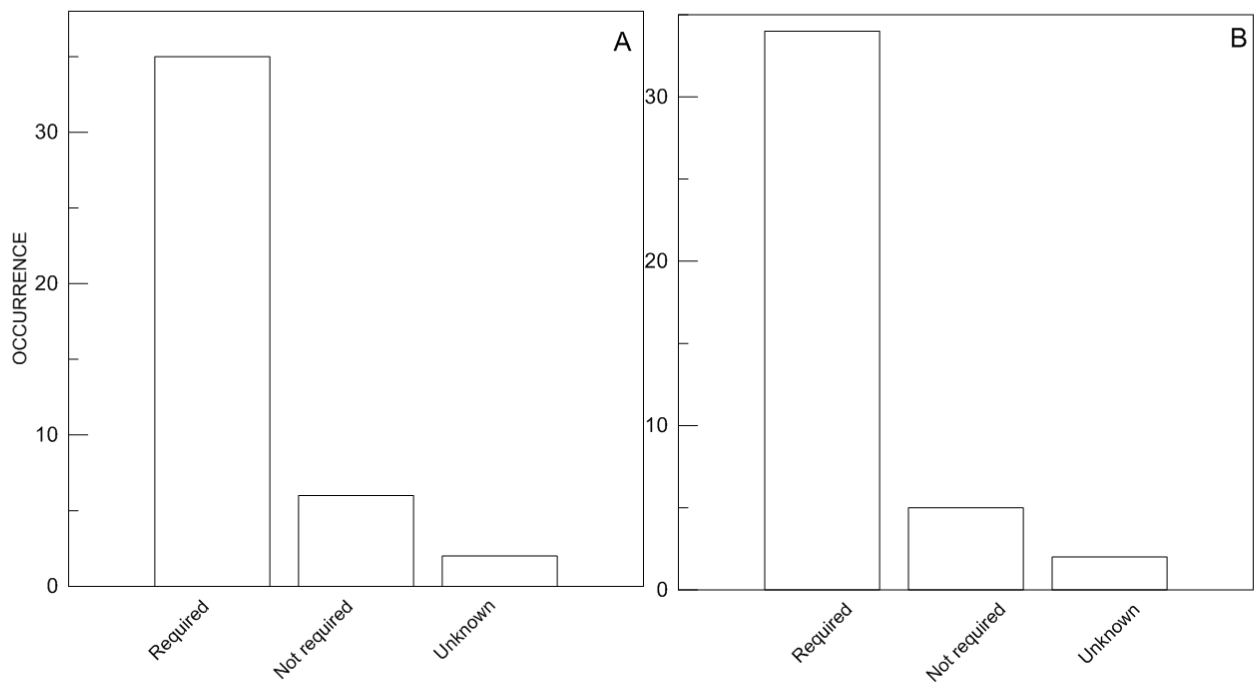

Figure 1.11. Summary of assessment field methods requiring a field visit in A) wetlands and B) streams

In order to promote usability and remain practicable, assessments are designed to be rapid. Need for rapid assessments methodologies is well documented and has been defined as one-half day or less field data collection and one half day of data analysis (van Dam et al., 1998; Mack et al., 2000; Smith et al., 1995; USEPA, 2004; Figure 1.12). The following section summarizes the data by examining the cumulative number of rapid assessment methods that can be completed within a given time. In wetlands 41% of assessment methods required 1 hour or less to complete, while a total of 68% of assessment methods capable of being completed in 4 hours or less, and a total of 80% of assessment methods capable of being completed in less than one day. Results indicated that 10% of methods required more than 1 day and as a result may not meet the definition of a rapid assessment. In streams, 41% of respondents identified assessment methods requiring 1 hour or less to complete, while a total of 66% of assessment methods could be completed in 4 hours or less, and a total of 78% of assessment methods could be completed in

less than one day. In streams, survey results indicated that 2% of methods required more than 1 day and as a result may not meet the definition of a rapid assessment.

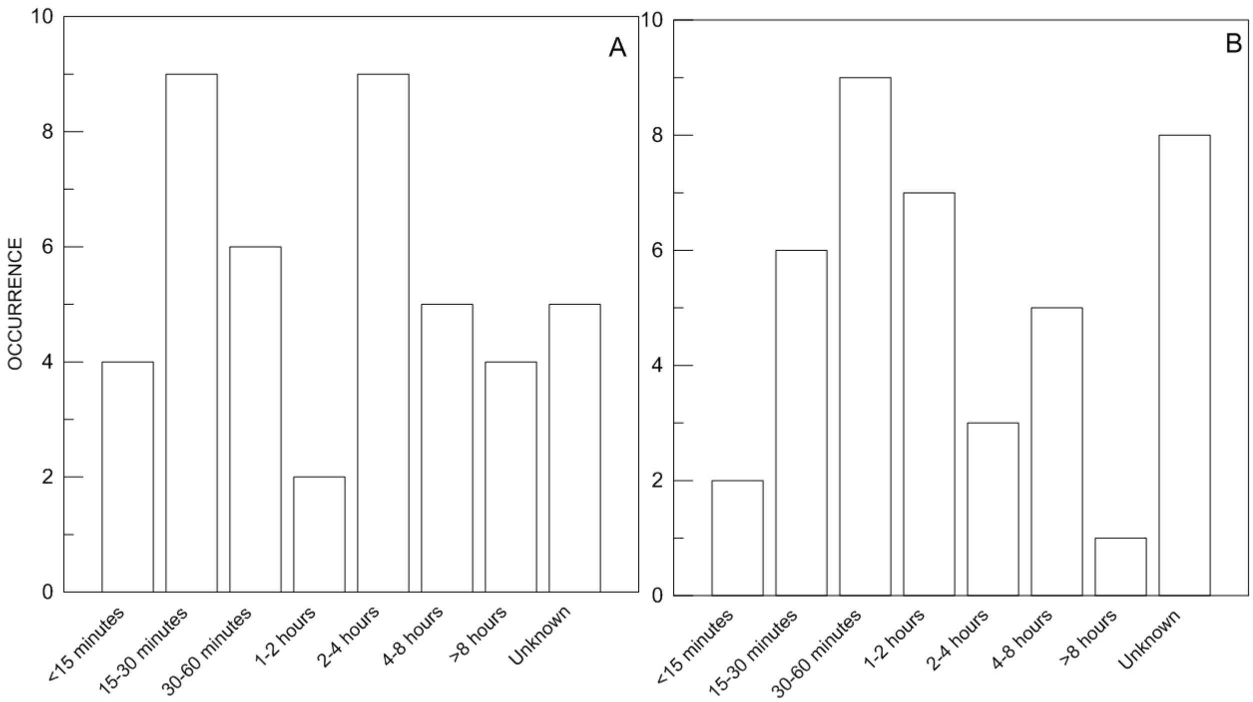

Figure 1.12. Summary of time required to complete assessment methods in A) wetlands and B) streams.

In addition to the time required to complete each assessment, the majority of approaches examined require some degree of training prior to application of the assessment methods (Figure 1.13). Further, 54% and 66% of assessment methods required special skill sets to complete the assessment method in wetlands and streams respectively (data not shown). In wetlands, most rapid assessment methods required familiarity with wetland delineation techniques, GIS or remote sensing skills, and plant identification expertise. In streams, responses focused on familiarity with stream dynamics, flow regimes, and the identification of plants, fish, and benthic invertebrates.

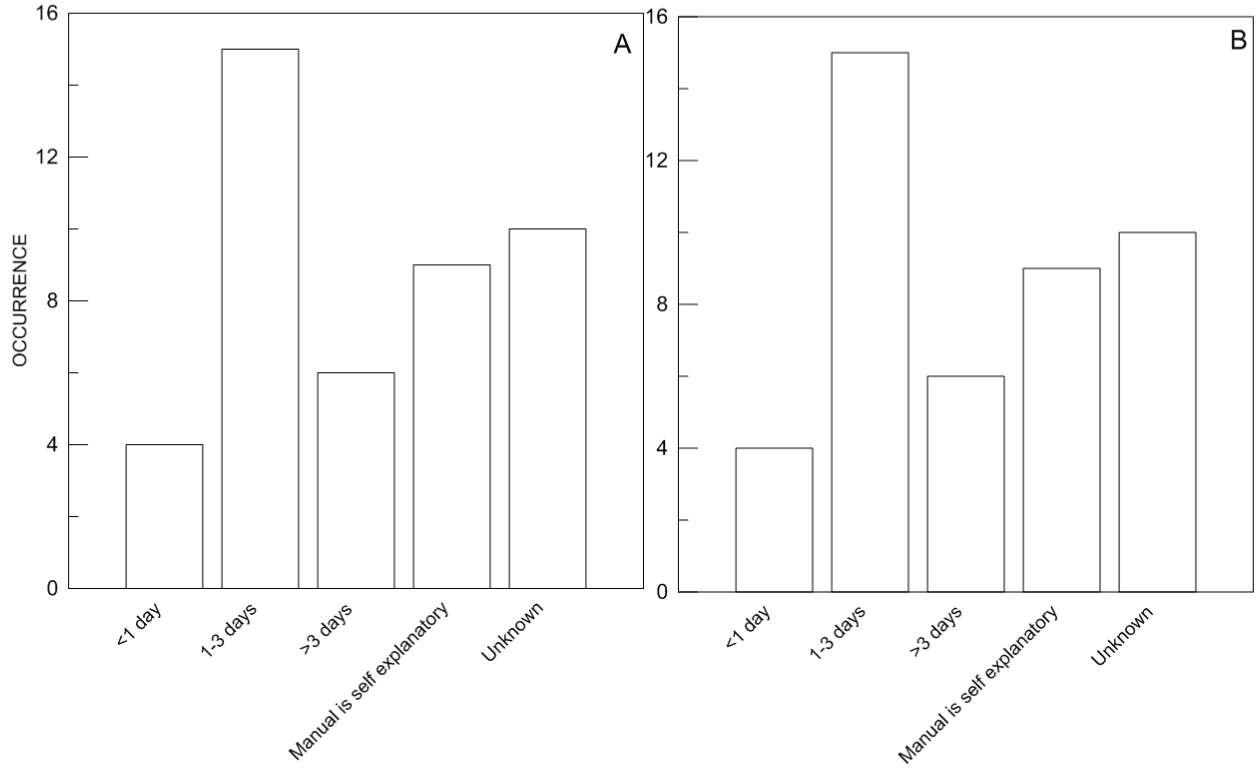

Figure 1.13. Summary of training time required to complete assessment methods in A) wetlands and B) streams.

1.4.3. Geographic extent

The current dataset is not intended to provide a comprehensive map outlining all wetland and stream rapid assessment methods in use, but presents an overview of many of the approaches commonly applied. As a result, Figures 1.14 and 1.15 depict the geographic extent of the wetland and stream assessment methods reported within available literature and survey responses. Within the current dataset, 30 states apply a wetland assessment, although the methods in use may not cover the entire geographic extent of the state or apply to all wetland classes within the state. Notably, more than one rapid assessment approach was reported in 12 states. Stream results show similar patterns, with 26 states applying a rapid assessment method in at least some portion of the state or subset of stream classes; 5 states exhibited more than one rapid assessment approach.   Both wetland and stream results display a paucity of methods across the Midwestern

and Arid West portions of the United States, potentially due to the high degree of agricultural manipulation in the central portion of the nation and the ephemeral hydrology associated with western states (Wakeley, 2002; USACE, 2005b).

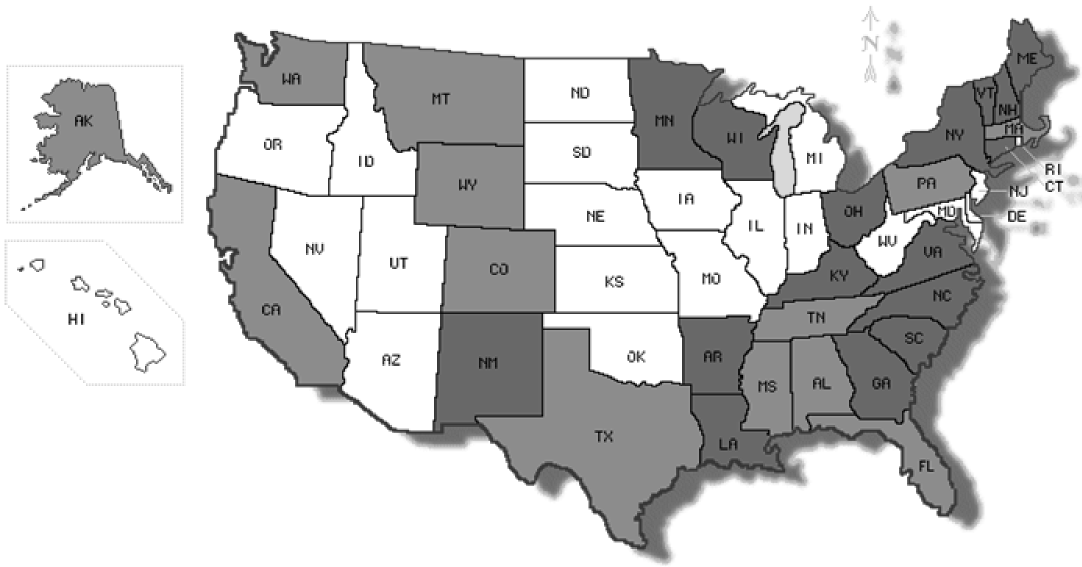

Figure 1.14. Regional extent of wetland rapid assessment methods examined. The dark shading indicates that one assessment method is in use within the state; light shading indicates more than one assessment is in use within the state.

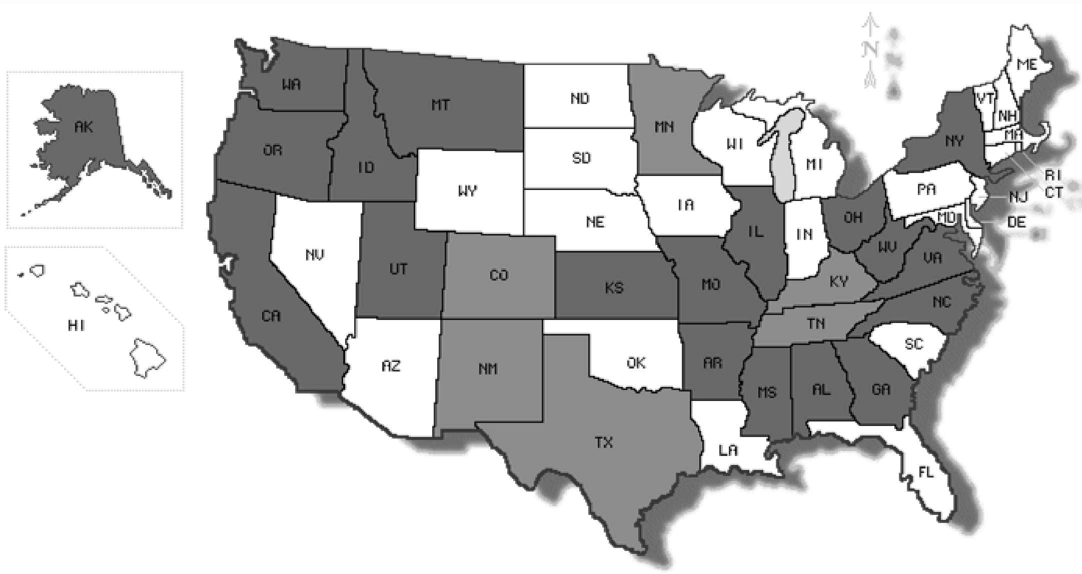

Figure 1.15. Regional extent of stream rapid assessment methods examined. The dark shading indicates that one assessment method is in use within the state; light shading indicates more than one assessment is in use within the state.

1.4.4. Assessment method modification

A number of assessment methods have undergone geographic expansion or modifications designed to address additional ecosystem types. For example, Klimas et al. (2011) expanded the classification of wetlands across much of the Mississippi Alluvial Valley region. Also, Wilder et al. (2012) expanded the geographical extent and wetland types addressed within the southeastern US. This was accomplished by combining existing assessment methods (Noble et al., 2007; 2011) with newly collected and calibrated data. In both cases, the modification process was well documented. However, the modification of several assessment methods based upon best professional judgment with limited documentation has resulted in recent debate.

Within the current dataset, a large percentage of assessment methods in use showed signs of modification, however the level of modification applied and the amount of supporting documentation available remains unclear (Figure 1.16). The types of modifications utilized include expansion into new geographic areas or ecosystem types, re-calibration of assessment variables and equations, or revision of assessment sampling protocols. For example, the Ohio Rapid Assessment Method (developed for application on Ohio) has been applied in Pennsylvania, Florida, and New York (Mack, 2001). Several assessment approaches recalibrate existing methods based upon internal discussions, input from partner agencies, and public comment. This includes reinterpreting the valuation of assessment method variables or changing the equations that calculate an assessment score (USACE New Orleans District, 2012). Further, assessment protocols have undergone modification including the addition or removal of assessment variables or the clarification of qualitative, categorical descriptions (Regulatory Branch, 2010c). The modification of assessment methods can result in improvements in validity and applicability, as observed in assessment approaches that include peer review, field testing,

and in some cases release of an interim version prior to final publication (Noble et al., 2010). The current dataset suggests that recent debate has resulted from the lack of documentation providing evidence of the decision making process used to modify existing methods.

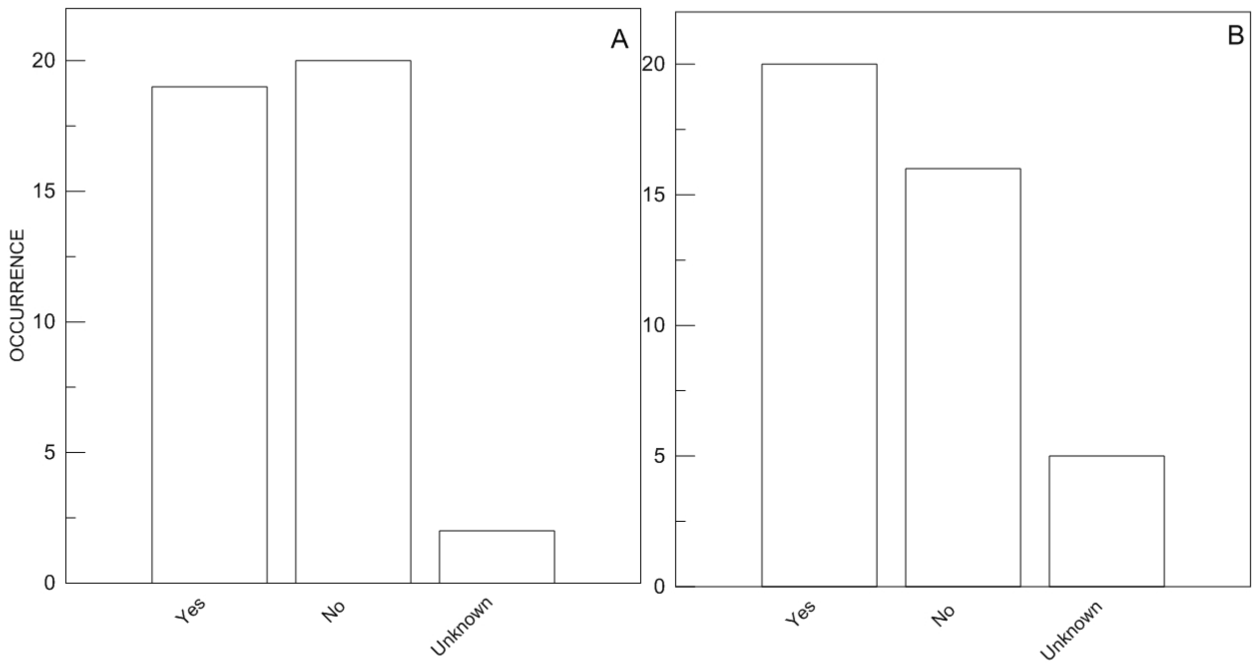

Figure 1.16. Summary assessment methods that have been modified in A) wetlands and B) streams.

1.5. Discussion

The following trends were observed in both wetland and stream assessments. Assessment methods developed by interagency teams were more likely to employ ecosystem classification, calibration based on some degree of reference data, and peer review. This suggests that interagency teams develop methodologies that limit variability through classification, base method outcomes on collected data, and document the decision making process through assessment review.

Ecosystem classification limits the observed variability within functional and conditional assessment methods, and results suggest that methods employing a classification scheme were

considered more objective. In wetlands 67% of the assessments applying classification were reported to be objective, while in streams 83% of approaches using classification were considered objective by survey respondents. Additionally, assessment methods that applied ecosystem classification were also more likely to utilize quantitative data. In wetlands and stream respectively, 64% and 73% of assessment methods applying ecosystem classification utilized quantitative data. These findings suggest that the combination of classification and quantitative data decreases variability while increasing the repeatability and validity of assessment methods.

Assessment methods based on some degree of reference data collection were considered more objective than approaches based on BPJ and/or available literature. In wetlands, 67% of methods considered objective applied reference data, while 72% of stream assessments considered objective included reference data. The application of reference data increases validity and defensibility by allowing for documentation of the relationships between assessment variables and ecological function or condition (Smith et al., 1995). Assessment methods based upon the collection of reference data allow for verifying data submitted as part of the permitting, monitoring or mitigation efforts, and the use of reference data allows for assessment method validation using independent measures of ecological condition or function (Fennessy et al., 2007).

In order for assessment methods to remain practical, they must be rapid and the requirement of a field visit increases assessment length. A field site visit is often required to accurately conduct the classification and collection of quantitative data previously discussed. In wetlands, assessment methods not requiring a field visit required an average of 2 hours to complete, assessment approaches requiring a field visit based on narrative statements (i.e., qualitative data) required an average of 2.2 hours, and methods requiring a field visit with some

quantitative data collection averaged between 3.8 and 5.5 hours to complete. Results observed in stream assessments were similar although more variable, with methods lacking a field visit requiring an average of 3.2 hours, narrative (i.e., qualitative) data requiring an average of 2.9 hours, and methods requiring a field visit with some quantitative data collection required between 2.5 and 8 hours to complete.

Overall study results indicate that natural resource professionals apply a large number of assessment methods exhibiting a wide range of variability. For example, rapid assessment approaches encompass measurements of both ecological function and condition, some methods apply classification while others do not, and method calibration is based on a variety of factors including reference data, BPJ, and available literature. Peer review is included in the development of some but not all approaches. A range of perceived user objectivity also exists within rapid assessment methods, with approaches requiring quantitative, qualitative, and a combination of measurement types. Further, the inclusion of a field site visit, special skills, equipment, and training all effect the time requirements and applicability of rapid assessment methods. Finally, a large number of assessment methods have undergone modification, resulting in recent debate. The data examining wetland and streams assessments provides an opportunity to examine current practices and make recommendations for improving rapid ecological assessment approaches.

1.6 Recommendations

1.6.1. Development of an Ecological Assessment Technical Standard

The application of technical standards has been applied to a number of natural resource management initiatives including the identification of wetlands, wetland hydrologic criteria, and hydric soils (Environmental Laboratory, 1987; USACE, 2005; NTCHS, 2007). Developing

technically sound and scientifically defensible approaches for measuring the function, condition and performance of ecosystems remains important in the context of environmental planning, permitting, impact assessment, and calculation of mitigation requirements (Zedler, 1996). As outlined in the sections above, a variety of rapid assessment methodologies underwent development and implementation; including HGM (various locations), FACWet (Colorado), the Charleston (South Carolina) and modified-Charleston (Louisiana) methods, and others (Brinson et al., 1998; Smith and Klimas, 2002; Johnson et al., 2011; USACE Charleston District, 2010c; USACE New Orleans District, 2012). The application of some rapid assessment methods resulted in scrutiny questioning the development process utilized and the technical validity of several methodologies. Much of the debate surrounding rapid assessment approaches focuses on 1) documentation of development strategy and decision making, 2) repeatability and consistency of assessment results, and 3) calibration of rapid assessment outcomes. Despite the variability outlined within the current study, sufficient common ground exists allowing for refinement and improvement of existing methods to achieve more scientifically based and technically defensible results.

A technical standard would provide a framework for rapid ecosystem assessment method development/implementation and a path for improving currently employed methods**.** The core aspects of the framework focus on 1) use of a diverse development team with clear assessment goals, 2) ecosystem classification and geographical extent, 3) collection of repeatable and quantitative data, 4) scaling of the assessment method based on reference data, and 5) rapid application. In addition to developing a technical standard for the development of new rapid assessment applications, it is important to address methodologies already in use. Although many of the rapid assessment methods utilized today fail to adequately address wetland classification,

quantitative data, and reference based scaling, opportunities exist to improve technical validity and defensibility. Therefore, the technical standard should include strategies and approaches designed to increase confidence in existing methods while decreasing uncertainty. A technical standard for rapid assessment approaches promotes a level of technical validity and transparency in ecosystem monitoring, impact assessment, and calculation of mitigation requirements.

1.6.2. Training

The study results suggest a number of areas in which rapid assessment method developers and users require training regarding assessment methods and concepts (Figure 1.16). For example, 7% of assessment documents did not specify if classification was part of the assessment method, and 10% of approaches failed to identify whether that the assessment method was designed to measure ecosystem function or condition. Further, over 50% of methods examined require specialized skill sets, equipment, and/or training. Both certified collegiate and professional training courses can be conducted locally, regionally, or on a national basis in order to improve the utilization of rapid assessment approaches.

1.6.3. Expand geographical and ecosystem coverage

Figures 1.14 and 1.15 outline the extent of assessment method reported within the current dataset and suggest that many areas lack ecological rapid assessments for wetland and stream types and/or lack geographical coverage. The expansion of methods into larger areas and more ecosystem classes allows for a more uniform approach to assessment and the determination of mitigation requirements, monitoring, and other regulatory efforts. Geographic expansion should include both broadening of existing methods where appropriate and the development of new assessment approaches. In conjunction with the proposal to develop an ecological

assessment technical standard, future expansion would promote valid and defensible assessment results.

1.7. Summary

A large number of rapid ecological rapid assessment approaches remain in use throughout the United States, exhibiting a wide variety of characteristics. However, existing assessment methods can be categorized based on small number of key components including: method development team, ecosystem classification, method calibration, geographic extent, time requirements, and the type of data collected and generated. The current dataset suggests that assessment approaches resulting in the most repeatable, objective, and technically sound outcomes were developed using interdisciplinary groups, classification, quantitative data, and peer review. As a result, recommendations are provided to promote science-based approaches through development of a technical standard for rapid ecological assessment, additional training, and guidance for modifying or expanding the geographical reach of existing methods.

1.8. Hypothesis

The research presented in this dissertation seeks to evaluate and improve the validity of rapid ecological assessment methods within both wetland and stream environments. Studies focus on validating rapid functional assessment approaches through the use of biogeochemical measures, hydrologic monitoring, and water quality parameters. We also investigate the capacity to develop application tools including restoration recovery trajectories for assessing ecosystem performance within project relevant timeframes. Further, a technical standard for rapid ecological assessment is developed to promote science-based approaches in ecological assessment.

We hypothesize that direct and proxy measures of biogeochemical function will support results generated within the rapid assessment approach. Further, measuring biogeochemical functions requires the coupling of abundance measures (e.g., soil nutrient concentrations) with processing or transport mechanisms (e.g., microbial activity, flood frequency). Also, ecosystem performance curves based on subset of rapid assessment parameters will aid in the development of restoration milestones. Finally, a technical standard for rapid ecological assessment methods will promote technically sound, science-based assessment approaches and outcomes.

1.9. Synopsis of Chapters

In Chapter 2, the effectiveness of a rapid assessment method developed for use in bottomland hardwood wetlands is evaluated. Biogeochemical and hydrologic measures are compared with results of a rapid assessment method examining nutrient cycling, organic C export, and water quality functions. Additionally, we explore the need to couple abundance measures and transport mechanisms when examining biogeochemical functions.

Having established the validity of the rapid assessment method evaluated in Chapter 2, Chapter 3 develops a framework for identifying restoration trajectory metrics within project-relevant timescales. Four out of 17 examined variables yield positive restoration trajectories within a few years to 20 years. Remaining variables provide limited useful information within critical early years following reforestation due to the time required for measurable changes to occur. As a result, assessment components are classified into three categories of rapid response, response, and stable variables. Development of early restoration milestones and performance standards should focus on rapid response variables.

In Chapter 4, a rapid assessment method developed for headwater streams is investigated in order to expand and verify the results outlined in Chapter 2. Rapid assessment method

evaluation utilizes proxy measures of nutrient inputs, processing, and transport applying leaf litter traps, leaf litter decomposition, and stream loading equations respectively. Water quality parameters and the outcomes of an economic analysis comparing rapid and traditional assessment approaches are also examined. Additionally, we develop a conceptual model outlining the relationship between biogeochemical cycling, disturbance regime, and ecological recovery.

Based on the wide array of rapid assessment methods examined within this literature review and the outcomes presented in Chapters 2 through 4, Chapter 5 outlines a technical standard for the development and application of rapid assessment approaches. The technical standard provides a description of nine independent, testable components as well as providing guidance on how existing rapid assessment approaches can be improved. The implications of all research chapters are discussed in Chapter 6.

CHAPTER 2: LINKING WETLAND FUNCTIONAL RAPID ASSESSMENT METHODS WITH QUANTITATIVE HYDROLOGICAL AND BIOGEOCHEMICAL MEASUREMENTS ACROSS A RESTORATION CHRONOSEQUENCE

2.1. Abstract

The need for practical, repeatable, and technically sound ecosystem assessment methods remains essential to natural resource management. Rapid assessment methodologies determining ecosystem condition and function continue expansion, especially within wetlands. However, few studies determine the validity of rapid assessment approaches by applying quantitative parameters, especially with respect to biogeochemical functions. Functional measurements require extensive sampling and analytical expertise, beyond financial and time constraints of most restoration projects. Further, measuring biogeochemical ecosystem functions requires the coupling of abundance measures (e.g., soil nutrient concentrations) with processing or transport mechanisms (e.g., microbial activity, flood frequency). This work assessed nutrient cycling, organic C export, and water quality improvement functions applied to > 300 km^2 of restored bottomland hardwood forests located in the Mississippi River Valley, USA. Assessment parameters (e.g., sapling shrub density, organic soil horizon thickness) and biogeochemical measures (e.g., microbial biomass C, potentially mineralizable N) were determined at 45 reforested areas and 21 control locations representing an 80 yr restoration chronosequence. Significantly higher rapid assessment outcomes were associated with increased ecosystem functionality (p = 0.001 - 0.029). These findings suggest that rapid assessment tools serve as reliable proxies for measurements of nutrient and biogeochemical cycling; validating the procedure examined. Assessment scores were also associated with increased restoration stand age (p_s < 0.001) supporting further development of similar rapid assessments utilizing ecosystem

classification, qualitative data collection, and scaling based on reference data. The wide variety of rapid assessments in use underscores the need for validation with biogeochemical and hydrological measurements.

2.2. Introduction

Wetlands and aquatic ecosystems provide a number of well established biological, chemical, and hydrologic functions linked to ecosystem services that prove beneficial to society (Novitski et al., 1996; Smith et al., 1995). Continued human development pressure has resulted in regulation and permitting procedures for activities impacting wetlands and associated functions. (Cole, 2006; Ainslie, 1994). A variety of ecosystem assessment strategies were developed with the goal of improving wetland management, and recent trends in wetland conditional and functional evaluation focus on rapid assessment methods (Stein et al., 2009; 2009b). Fennessy et al. (2007) identified over 40 rapid assessment protocols and development of additional methodologies continues (Johnson et al., 2011; Wilder et al., 2012). The increased use of rapid assessments methodologies is due to the need for techniques that are sensitive to ecosystem impacts, are easily attained in a short period of time (one-half day or less field data collection), and are insensitive to seasonality (van Dam et al., 1998; Fennessy et al., 1998; 2004; Mack et al., 2000; Berkowitz et al., 2011).

The Hydrogeomorphic (HGM) approach represents a suite of rapid assessment procedures designed to evaluate ecological function. HGM techniques typically include geomorphic, vegetative, and structural measurements that have been applied to numerous wetland and stream types and form the basis for the methods presented in the current work (Brinson 1993, 1995; Rowe et al., 2009). In the HGM approach, easily attainable measurements of ecosystem structure (e.g., tree diameter, ground cover) are combined using simple multimetric equations to produce

Functional Capacity Index (FCI) scores ranging from zero (a lack of wetland function) to 1.0 (fully functional) (Smith et al., 1995).

The use of HGM techniques maintains several advantages over other rapid assessment approaches by requiring 3 key factors 1) ecosystem classification, 2) collection of quantitative data, and 3) scaling based on reference data (Clairain, 2002). The HGM approach has been approved for use by several US federal resource management agencies, continues expansion into additional geographic areas and ecosystem types, and has been upheld in several recent US court decisions as a legally defensible and acceptable methodology to assess resource impairment (Federal Register, 1997; Noble et al., 2010; Ovec v Corps re Reylas, 2012). Bauder et al. (2009) describes the HGM approach as more accurate than other rapid assessment methods, and Cole (2006), although critical of rapid assessments in general, describes HGM techniques as the best available method for rapidly assessing wetland function.

The accuracy and efficacy of rapid assessment methods can be strengthened through what Wakeley and Smith (2001) define as “validation” or testing rapid assessment outcome accuracy by using comparisons with field or laboratory measures of ecosystem function. Therefore, several published studies have attempted assessment method validation by comparing calculated FCI scores and ecosystem functional proxies. For example, Bohonak and Bauder (2011) describe an HGM method developed for vernal pool regions in California, USA that significantly, positively correlated rapid assessment outcomes for hydrologic and habitat functions with proxy measures including inundation period and the presence of indicator plant species. (Spearman rank correlation coefficients, r_s = 0.44-0.76). Hill et al. (2006) examined hydrologic functions for one depressional wetland in Tennessee, USA by developing a simulated hydrologic model. Significant relationships were found between rapid assessment results and hydrologic model

predictions following some alteration of the FCI empirical formulas (r^2 = 0.82-0.84). A similar study reported agreement between rapid assessments and simulated hydrologic modeling results within a depressional prairie pothole wetland located in North Dakota, USA (Pohll and Tracy 2000). Finally, Stein et al. (2009) compared measures of biological integrity and bird species richness to a rapid assessment of estuarine and riverine wetland condition in California, USA and reported significant relationships (r_s = 0.30-0.64).

While several studies link rapid assessment outcomes with hydrologic or habitat functions, there is a paucity of data comparing rapid assessment with proxies of biogeochemical cycling. Franklin et al. (2009) investigated six sites located within riverine wetlands in Tennessee, USA and reported a significant correlation between the FCI score generated for nutrient cycling function and leaf fall phosphorus concentrations. This one study represents the current extent of rapid assessment validation for biogeochemical function. Also, it remains unclear if determining higher material inputs, such as leaf nutrient inputs from litter fall, adequately addresses the question of ecosystem functions including nutrient cycling. Litter fall nutrient concentration must be linked with a microbial processing or physical transport mechanism carrying out the *cycling* of nutrients function. To further this discussion, we couple nutrient concentrations, microbial activity, and wetland hydrology as processing and transport mechanisms capable of driving the function of interest (Figure 2.1). For example, a wetlands ability to immobilize nutrients and other compounds imported to the wetland via flooding requires 1) material transport and 2) immobilization.

Determining changes in ecosystem function following restoration allows resource managers to quantify restoration success (Whigham, 1999; Kentula, 2000). Several studies suggest changes in wetland function based on stand age and forest succession. For example,

Faulkner et al. (2010) indicates that newly restored forested wetlands provide limited carbon storage and water quality improvement functions compared to natural systems, but are expected to show functional increases with time. Hunter and Faulkner (2001) and Hunter et al. (2008) report increases in denitrification potential and other biogeochemical functional proxies when comparing restored and natural forested wetlands. As a result, the current work examines changes in wetland functions and rapid assessment outcomes across the restoration chronosequence.

This study seeks to 1) link rapid assessment functional indices with measures of biogeochemistry and hydrology, 2) couple nutrient or microbial concentrations with transport and processing mechanisms providing direct measures of ecosystem functionality and 3) examine changes in three biogeochemical cycling functions with forest succession across a large dataset spanning an 80 yr restoration chronosequence.

2.3. Methods

2.3.1. Study sites

Forty-five reforested sites ranging from 1-20 yr post restoration plots located within management areas administered by the US Army Corps of Engineers, Forest Service, and various state agencies were used in this study. The study area included sample plots within the Yazoo Basin in Mississippi with one site located nearby in Louisiana, USA (Figure 2.2). Site selection included consideration for areas with open public access, restoration projects contributing to development of a restoration chronosequence, previous land use of 100% agricultural, and proximity to the region for which the rapid assessment protocol was developed. Restoration activities included tree seedling planting and did not include hydrologic modification. Species planted on restoration sites varied depending on topography and flood

regime but included a mixture of water oak (*Quercus nigra*), willow oak (*Quercus phellos*), Nuttall oak (*Quercus nuttallii*), Shumard oak (*Quercus shumardii*), green ash (*Fraxinus pennsylvanica*), pecan (*Carya illinoensis*), and bald cypress (*Taxodium distichum*).

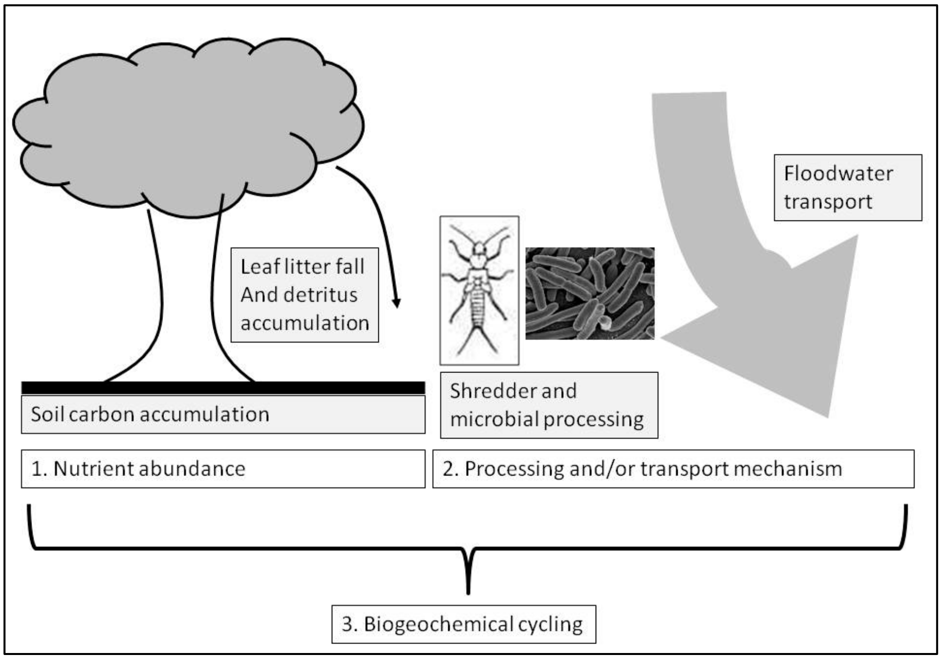

Figure 2.1. Conceptual diagram demonstrating the need to couple abundance measures with processing/transport mechanisms when evaluating biogeochemical function.

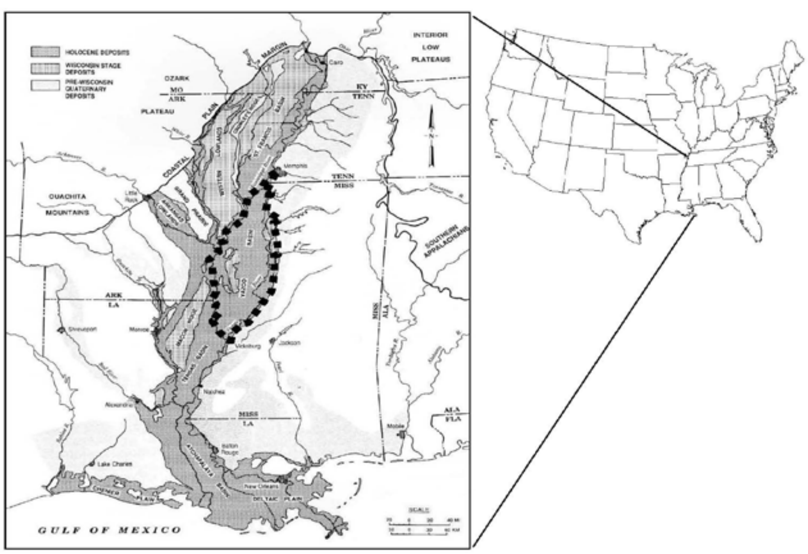

Figure 2.2. Study area highlighting the Lower Mississippi Valley. The Yazoo basin is outlined in black. (Saucier, 1994).

Additionally, twenty-one control sampling plots were located within the Delta National Forest and in surrounding areas. Control plots exhibited second growth stands exceeding 80 yr since known impact, thereby representing the least disturbed bottomland hardwood forest wetlands in the region. Although Smith and Klimas (2002) address a number of wetland subclasses (e.g., connected depression, flats) within the study area, all data presented within the current study was collected in areas classified as riverine backwater wetlands only. As a result, both reforested and mature sample areas receive hydrologic inputs from direct precipitation and backwater flooding at an estimated frequency of 5 yrs or less. Backwater flooding is defined by Smith and Klimas (2002) as inundation resulting from impeded drainage, usually due to high water in downstream systems. Typical backwater flooding scenarios result when streams in flood stage prevent effective drainage within the tributary network; low-lying areas associated with those tributaries become saturated or inundated.

Sites locations occur within meander belts 2 and 3 of the Mississippi river floodplain (Saucier, 1994). Soil series throughout the study area included Sharkey very-fine, smectitic, thermic chromic epiaquerts, Dowling very-fine, smectitic, nonacid, thermic vertic endoaquepts, Perry very-fine, smectitic, thermic chromic epiaquerts, and Alligator very-fine, smectitic, thermic chromic dystraquerts; poorly drained clays with small inclusions of somewhat poorly drained Commerce fine silty, mixed, superactive, nonacid thermal fluvaquentic endoaquepts. All observed soil series phases were between 0-2 percent slope (Soil Survey Staff, 2011).

2.3.2. Rapid assessment variable collection

This study evaluated nine rapid assessment parameters representing three biogeochemical functions (nutrient cycling, export of organic C, and water quality improvements) addressed within the Regional Guidebook for Applying the HGM Approach to Assessing Wetland

Functions of Selected Regional Wetland Subclasses, Yazoo Basin, Lower Mississippi River Alluvial Valley (Table 2. 1). Smith and Klimas (2002) provide detailed sampling instructions for each of the variables examined. Data collection occurred during the spring and early summer of 2011. The nine HGM variables underwent conversion into FCI scores via application of multimetric equations (Table 2. 1).

2.3.3 Validation variable collection

Measures of ecosystem biogeochemical function require both the processing and transport of elements and compounds. For example in order to examine export of organic C function, investigators must examine C processing within an ecosystem and the transport mechanism capable of exporting C to down gradient locations. As a result biogeochemical functional measures must include both concentration analysis as well as a means of processing and transport. Soil collection consisted of 10 cm deep triplicate soil cores obtained proximal to the sampling location of each rapid assessment determination. Triplicate cores underwent homogenization and subsequent refrigeration at 4 ^{o}C until analyses. All results are presented on a dry-weight basis. Weight percent moisture contents were determined by drying subsamples in a forced-air oven at 70 ^{o}C until constant weight. Total carbon (C) and total nitrogen (N) content was determined on dried, ground subsamples analyzed with a Costech Elemental Combustion System (Valencia, CA; Kahn, 1998).

Organic matter content determination followed the loss on ignition (LOI) method; dried ground samples underwent combustion at 550°C in a muffle furnace for 4 hours (Sparks, 1996; Nelson and Sommers, 1996). Microbial biomass C (MBC) utilized a 24-hour chloroform fumigation followed by K_2SO_4 extraction and combustion analysis on a Shimadzu TOC-V series C analyzer.

Table 2.1. Summary of rapid assessment variables, description, sampling technique, and rapid assessment functions. Modified from Smith and Klimas (2002).

Assessment variable	**Description and symbol**	**Sampling technique**
1. Flood frequency	Frequency of overbank or backwater flooding (V_{FREQ})	Measured from flood frequency map/stream gauge data
2. Cation exchange capacity	Cation exchange capacity change due to soil disturbance (V_{CEC})	Estimated based on soil type
3. Tree basal area	Basal area per hectare; proportional to tree biomass (V_{TBH})	Diameter of all trees > 7.6 cm diameter in circular 0.04 ha plot
4. Snag density	Density of standing dead woody stems (V_{SNAG})	Count of all snags > 7.6 cm diameter in a circular 0.04 ha plot
5. Woody debris biomass	Volume of woody debris biomass per hectare (V_{WD})	Count of nonliving stems along a 3.7 m transect
6. Shrub-sapling density	Density of saplings and shrubs per hectare (V_{SSD})	Count of all woody stems within two 0.004 ha plots
7. Ground vegetation cover	Percent cover of herbaceous and woody vegetation (V_{GVC})	Estimated percentage of ground covered with vegetation plots
8. O horizon biomass	Mass of organic matter in the O horizon (V_{OHOR})	Measured O horizon thickness
9. A horizon biomass	Mass of organic matter in the A horizon (A_{OHOR})	Measured A horizon thickness
Rapid assessment function	**Multimetric equation**	**Function description**
1. Cycle nutrients	$FCI = \dfrac{\left[\dfrac{(V_{TBA}+V_{SSD}+V_{GVC})}{3} + \dfrac{(V_{OHOR}+V_{AHOR}+V_{WD}+V_{SNAG})}{4}\right]}{2}$	Ability to convert nutrients from organic to inorganic forms through biogeochemistry
2. Export organic C	$FCI = V_{FREQ} \times \dfrac{\left[\dfrac{(V_{OHOR}+V_{WD}+V_{SNAG})}{3}\right] + \left[\dfrac{V_{TBA}+V_{SSD}+V_{GVC}}{3}\right]}{2}$	Capacity to export dissolved and particulate organic C to downstream systems
3. Improve water quality	$FCI = V_{FREQ} \times \left[\dfrac{(V_{CEC}+V_{OHOR}+V_{AHOR})}{3}\right]$	Ability to remove or temporarily immobilize nutrients and other compounds

Microbial biomass C was determined by subtracting the extractable total organic C (TOC; Shimadzu TOC-V series C analyzer) in the triplicate controls from the triplicate chloroform-treated samples as presented by Vance et al. (1987) with modifications (White and Reddy 2001; Malecki-Brown and White, 2009). Exchangeable NH_4^+ was determined following Mulvaney (1996) with the following modification: 25 mls of KCl was added to ~ 2 g moist soil in a 30-ml centrifuge tube and shaken for 30 min. These served as the time zero controls for the potentially mineralizable N (PMN) rates. The PMN rates were determined by adding 10 mL of distilled, deionized water to ~5 g of moist soil in glass serum bottles evacuated with 99.99% O_2-free N_2 gas (White and Reddy, 2001). Samples for PMN analysis were incubated for 10 days at 40°C, extracted with 2M KCl and compared to exchangeable NH_4^+ values (time-zero controls). Extracts were preserved at pH < 2 and analyzed using a Seal Analytical AQ2 Automated Discrete Analyzer (Mequon, WI) following EPA method 350.1 (USEPA,1993; Van Zomeren et al., 2011).

Additionally, direct monitoring of near-surface hydrology utilized slotted groundwater wells installed to 50 cm below the soil surface. Within each study area one groundwater well was located at the center of the study plot in which rapid assessment data was collected (Smith and Klimas 2002). The use of a single groundwater well establishes the site hydrology within the immediate area represented by the rapid assessment method. Water table level recordings were taken twice daily using automated Insitu Level TROLL 500 dataloggers (Ft. Collins, CO). Well construction, installation, and data analysis followed Sprecher (2000) and US Army Corps of Engineers (2005) specifications. The National Research Council (1995) defined high water tables as occurring within 30 cm of the surface, ensuring saturation or inundation within the majority of the root zone. This represents the area considered critical for wetland soil

biogeochemical functioning due to the onset of anaerobic conditions within a zone exhibiting high concentrations of organic matter and soil microbes (Chorover et al., 2007; Segers, 1998). As a result, water table data are expressed as the percentage of high-water table days (i.e. the incidence of water tables >30 cm) occurring during the monitoring period (May 2010 – July 2011; Berkowitz and Sallee, 2011).

2.3.4 Data analysis

Following testing for normality (Shapiro-Wilk test), Pearson Product Moment Correlations (Pearson coefficients) were generated by comparing measures of soil chemistry biogeochemical function and hydrology with rapid assessment results. The rapid assessment outcomes (i.e., FCI scores) for the nutrient cycling function were compared to soil C, N, MBC, and PMN values; which represent the wetland nutrient pool and the potential nutrient processing capacity within the study area. FCI scores for the export of organic C function were compared to soil TOC, LOI, and wetland hydrology; accounting for the abundance of organic C sources and the flooding mechanism responsible for C transport to down gradient locations. Finally, FCI scores for the improve water quality function were compared to the hydrology, MBC and PMN values; examining the capacity of study areas to receive and process nutrient loads coming into the wetland. Significance values were evaluated at the $\alpha = 0.05$ level. Linear regression analysis resulted in the calculation of r^2 (SPSS IBM, Inc. Version 20), however ecological restoration data often deviates from linear patterns within decadal time scales (Battaglia et al., 2002). Comparing between rapid assessment FCI scores and the aforementioned measures provides a methodology for validating the HGM approach, where positive relationships promote increased confidence with respect to the validity of rapid assessment outcomes in restored and control wetland areas.

Additionally, changes in FCI scores corresponding to stand age underwent evaluation across the 80 yr restoration chronosequence to determine if the rapid assessment distinguished between restored areas of different ages. Older restored wetlands and control plots are expected to provide increased biogeochemical measures and wetland functions compared to agricultural and recently reforested hardwood wetlands (Whiting and Chanton, 2001; Bruland and Richardson, 2006; Faulkner et al., 2010). Due to a lack of normality within the restoration chronosequence dataset the Spearman Rank Order Correlation (r_s) was applied to all chronosequence data ($\alpha = 0.05$).

2.4. Results and Discussion

2.4.1 Nutrient cycling

Nutrient cycling within wetland ecosystems includes transformation of nutrients between organic and inorganic pools; representing an important function of wetland biogeochemistry (Ovington, 1965; Pomeroy, 1970). The majority of conversions between labile and recalcitrant nutrient pools result from alterations of soil organic matter and the cycling of nutrients through the food chain via plant uptake, and processing by means of the microbial loop (Vogt et al., 1986; Fennessy et al., 2008). As a result, soil C and N concentrations represent a measure of potential nutrient cycling. Total C was positively, significantly correlated to nutrient cycling FCI scores [$r = 0.781$; $p = 0.002$; Figure 2.3(A)]. Total N results were also positively, significantly correlated with FCI scores ($r = 0.641$; $p = 0.017$) lending support to the hypothesis that the rapid assessment approach distinguishes between soil nutrient regimes [Figure 2.3(B)].

The fact that rapid assessment outcomes relate to soil nutrient concentrations suggests that increased soil nutrient concentrations indicate high nutrient cycling functionality. However, soil nutrient concentrations examined alone fail to directly address the question of nutrient

cycling. Additional metrics are needed to link soil nutrient concentrations and the capacity for cycling of the nutrients within a wetland (Figure 2.1). Introducing measures of microbial pool size and activity in conjunction with nutrient concentrations provides evidence that 1) nutrient concentrations correspond to rapid assessment FCI scores and 2) wetlands contain the necessary microbial pool to accomplish nutrient cycling functions.

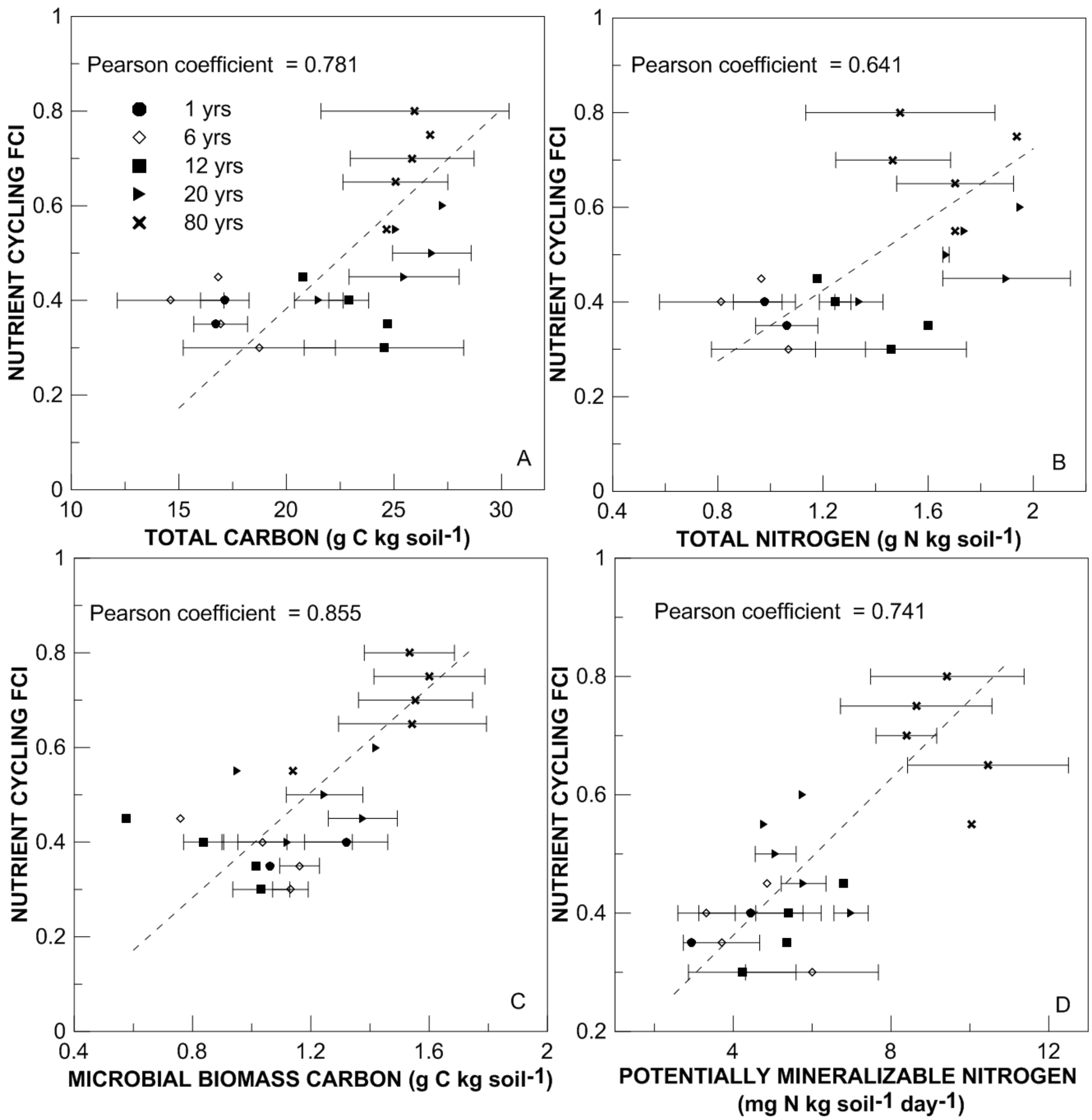

Figure 2.3. Comparison between measures of soil chemistry and biogeochemical function to nutrient cycling rapid assessment outcomes (functional capacity index scores). Measures include: A) soil C, B) soil N, C) MBC, and D) PMN. Error bars represent one standard deviation.

MBC represents the subset of the greater C pool responsible for driving nutrient bioavailability to plants and higher trophic levels by the conversion of organic nutrient forms to more labile inorganic forms (White and Reddy, 2001). Therefore, higher MBC concentrations suggest a higher capacity to convert or cycle nutrients. The MBC was significantly, positively correlated with FCI scores [r = 0.855, $p < 0.001$; Figure 2.3(C)]. While the size of the microbial pool has proven useful in this and other studies, it is also important to measure the activity of the microbial pool since much of the pool can be inactive at any one time. Therefore, we compared the PMN rate as this is a direct measure of the microbial pool's transformation of organic N to inorganic N, a critical nutrient cycling process supporting plant growth. Therefore, PMN provides an indicator of labile N availability; which drives vegetative growth (White and Reddy 2000) with wetlands containing higher rates of PMN have a higher capacity to release bioavailable N (Reddy and DeLaune, 2008). The PMN rate was significantly correlated with rapid assessment outcomes across the wetlands sampled [r = 0.851, p = < 0.001; Figure 2.3(D)].

2.4.2 Export of organic C

The export of organic C function describes the capacity of a wetland to transfer dissolved and particulate organic C and associated nutrients out of the wetland system, providing a source of nutrients and other materials to down-gradient areas (Smith and Klimas 2002). The riverine and backwater wetlands found throughout the study area display high productivity and connectivity across large segments of the lower Mississippi Valley region, and remain important sources of the organic C for aquatic food webs and biogeochemical cycles in adjacent habitats (Elwood et al., 1983; Sedell et al., 1989). While dissolved organic C occupies the base of the microbial food web, driving biogeochemical function (Edwards, 1987, Edwards and Meyers, 1986), particulate organic C maintains populations of many shredders and filter feeding

organisms (Vannote et al., 1980). From a landscape perspective, watersheds with a high concentration of riverine and backwater wetlands export more organic C than watersheds with fewer wetlands (Mulholland and Kuenzler, 1979; Johnston et al., 1990).

As seen in the nutrient cycling data presented above, developing comprehensive measures of wetland functions requires validating rapid assessment metrics both with 1) high and low levels of abundance (e.g., nutrient concentration) and 2) a mechanism by which the ecosystem function is accomplished (e.g., microbial transformation of soil nutrients). Similarly, evaluating the export of organic C function applies measures of organic material abundance using TOC and organic matter as measured by LOI. The mechanism for exporting the C is determined by onsite hydrology, which provides a means by which physical export occurs. TOC significantly correlated to higher export organic C FCI scores [r = 0.797, $p < 0.001$; Figure 2.4(A)]. Organic matter content (LOI) also increased linearly with increasing export organic C FCI scores [r = 0.804, $p < 0.001$; Figure 2.4(B)].

Measurements of onsite hydrology are required to demonstrate that not only do sites with high rapid assessment scores exhibit increased soil TOC and organic matter content, but there is a capacity to export those materials to adjacent habitats via fluctuating water tables and backwater flooding. The abundance of high water table events (%) correlates well with generated FCI outcomes [r = 0.804, $p < 0.001$; Figure 2.4(C)] demonstrating that in addition to predicting higher nutrient and organic matter contents, sites with high FCI scores are also capable of moving materials down-gradient.

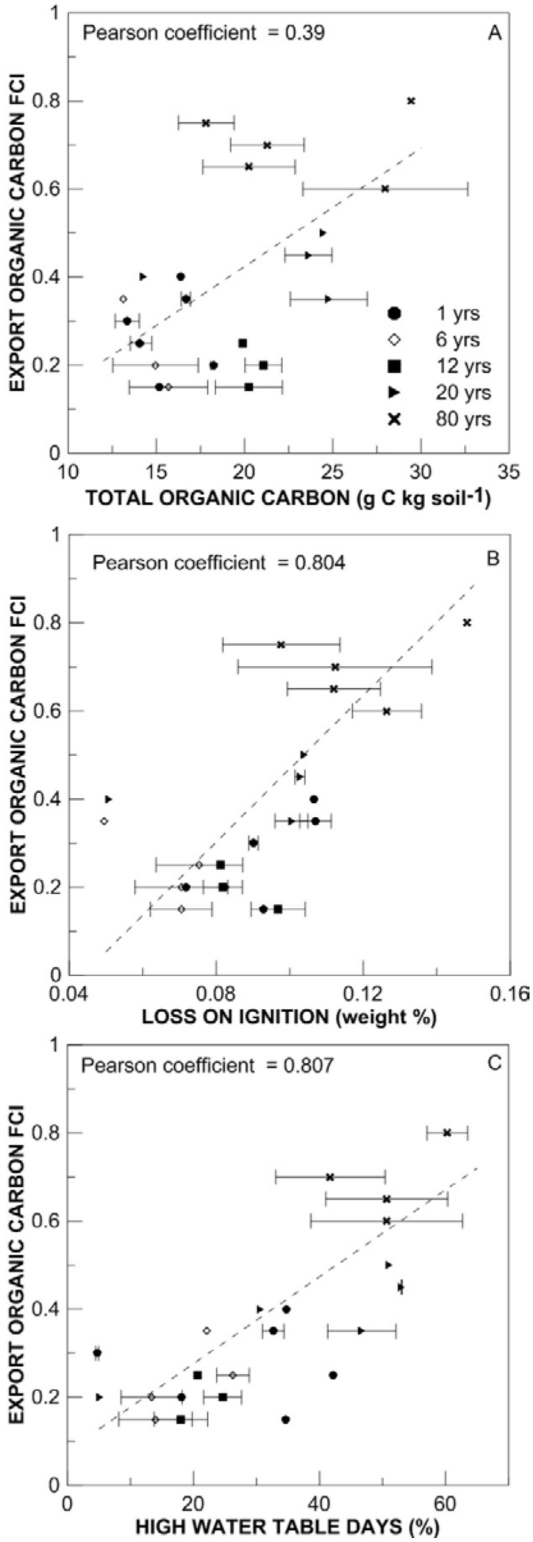

Figure 2.4. Comparison between measures of soil chemistry, biogeochemical function, and site hydrology to export organic C rapid assessment outcomes (functional capacity index scores). Measures include: A) TOC, B) organic matter as determined by LOI, and C) onsite hydrology (% of study period with high water table). Error bars represent one standard deviation.

2.4.3 Improve water quality

The third function of interest is the capacity of the wetlands to improve water quality, described by Smith and Klimas (2002) as the removal of elements and compounds. This function includes the ability of a wetland to permanently remove or temporarily immobilize nutrients, metals, and other elements and compounds that are imported to the wetland from various sources, primarily via flooding. The capability of wetlands to intercept materials transported from terrestrial environments via floodwaters is a well established and an important ecosystem service (Peterjohn and Correll, 1984; Cooper et al., 1986; 1987). Removal of materials includes biogeochemical processes such as complexation, chemical precipitation, adsorption, denitrification, immobilization, and other processes (Faulkner and Richardson, 1989, Reddy and DeLaune, 2008). Retention pathways include the sorption of nutrients (e.g., NH_4^+), pesticides, metals, and other substances to charged soil surface particles, particularly clays, which are abundant within the study area (Soil Survey Staff, 2011).

Additionally, many temporary removal mechanisms depend on the oxidation-reduction state of wetland soils (Reddy and DeLaune, 2008). For example, many phosphate compounds undergo sequestration by Fe and Mn oxides under oxidized conditions, while reduced sulfides bind with metal cations (e.g., Fe, Pb, Cu) forming insoluble, unreactive sulfides compounds that remain stable under strongly reduced environments (Khalid et al., 1978; Holford and Patrick, 1979). Regardless of the pathway that leads to retention, the water quality improvement function requires that 1) elements and compounds are imported through flooding and other means and 2) materials are sequestered through physical and biogeochemical processes.

Onsite hydrology data correlates well with generated FCI outcomes for the water quality improvement function [$r = 0.839$, $p = 0.005$; Figure 2.5(A)] demonstrating that sites with high

FCI scores are capable of importing elements and compounds into the wetlands, increasing the potential for water quality improvements to occur. Therefore, sample locations receiving higher rapid assessment scores exhibited increased exposure to floodwaters transporting elements and compounds. The PMN rates correlated to FCI scores [$r = 0.721$, $p = 0.029$; Figure 2.5(B)] indicating that areas receiving higher FCI scores display the capacity to provide increased nutrient availability for removal via plant uptake, denitrification, and other processes (Hanson et al., 1994). MBC concentrations also represent the capacity of a soil to convert nutrients between organic and inorganic pools, affecting water quality within wetlands and in the overlying water column. MBC values correlated with generated FCI outcomes [$r = 0.769$, $p = 0.009$; Figure 2.5(C)] further supporting the application of rapid assessment outcomes in determining a wetlands ability to improve water quality.

2.4.4 Changes in functions across the restoration chronosequence

Increases in FCI scores correlated with increasing restoration stand age for nutrient cycling ($r_s = 0.83$; $p < 0.001$), export organic C ($r_s = 0.78$; $p < 0.001$), and improve water quality functions ($r_s = 0.75$; $p < 0.001$). Functional assessment scores increased 17-23% between the time of restoration and 20 yr of forest growth. After 20 yr restored areas functional scores remain 16-44% below the levels observed at control sites. Results demonstrate that all three functional measures increase with forest succession and appear to follow a trajectory toward increased functionality. The lack of restored forested wetland sites between 20 yr and control (~ 80 yr) stems from the time of first implementation of many restoration projects beginning during the 1990s. This represents a gap in available information regarding forested wetland restoration within the study area.

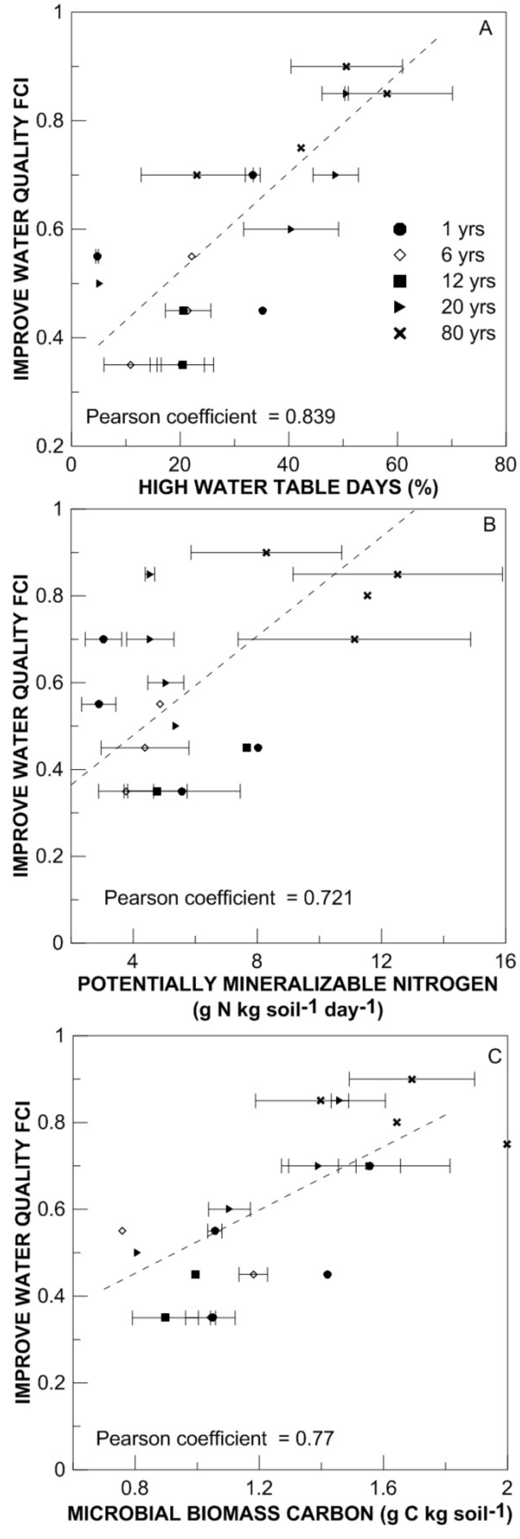

Figure 2.5. Comparison between measures of biogeochemical function and site hydrology to water quality improvement rapid assessment scores (functional capacity index scores). Measures include: A) onsite hydrology (% of study period with high water table), B) PMN, and C) MBC. Error bars represent one standard deviation.

2.5. Conclusions

The rapid assessment approach examined proved effective in discriminating between wetlands exhibiting a range of reforestation ages across three biogeochemical functions (nutrient cycling, export of organic C, water quality improvements). Measures or proxies of biogeochemical cycling significantly correlated with rapid assessment outcomes, providing evidence that the assessment method supplied valid, reliable outcomes while satisfying the requirements of a rapid approach. The rapid assessment method applied in the current study utilized wetland classification, quantitative data collection, and scaling based on reference data. Results suggest that HGM and similar assessment approaches incorporating these key elements provide a valuable tool for resource managers, the scientific community, and the public. However, the wide variety of rapid assessments in use increases the necessity for evaluation and validation with measures of ecosystem function or services. Further, efforts to validate rapid assessment procedures require utilization of both nutrient and microbial concentration measures and transfer/processing mechanisms in order to adequately address wetland functionality, especially with respect to biogeochemical functions. The study demonstrated that measurements of restored wetland functionality increased with forest succession, but did not achieve the level of functionality observed at control sites. As the growth and evolution of rapid assessment approaches continues, the need for accepted technical criteria addressing rapid assessment development and validation procedures will also continue to expand.

CHAPTER 3: DEVELOPMENT OF RESTORATION TRAJECTORY METRICS IN REFORESTED BOTTOMLAND HARDWOOD FORESTS APPLYING A RAPID ASSESSMENT APPROACH

3.1. Abstract

Large scale bottomland hardwood wetland restoration and reforestation efforts continue to expand throughout the Lower Mississippi Valley. Monitoring of restoration performance and the development of restoration trajectories pose challenges to resource managers and remain problematic due to 1) temporal patterns in forest succession, 2) budget constraints and short project monitoring timeframes, 3) disparity in the extent of pre-restoration hydrologic and landscape manipulations, and 4) lack of coherent restoration performance standards. The current work establishes a framework for identifying restoration trajectory metrics within project-relevant timescales. The study examined 17 variables commonly applied in rapid assessments. Four variables yielded positive restoration trajectories within a few years to 20 years. These include shrub-sapling density, ground vegetation cover, and development of organic and A soil horizons. Remaining variables including flood frequency and tree density provide limited useful information within critical early years following reforestation due to the time required for measurable changes to occur. As a result, assessment components are classified into three categories of rapid response, response, and stable variables. Restoring entities should maximize stable variables (e.g., afforestation species composition) during project implementation through site selection and planting techniques; while development of restoration milestones should focus on rapid response variables. Data collected at mature bottomland hardwood control sites displays the non-linearity of trajectory curves over decadal time scales.

3.2. Introduction

A variety of factors including settlement expansion, agriculture and forestry, and flood control decreased wetland acreages within the Lower Mississippi Valley (LMV) by 74% by 1982; with only 2.8 of an original 10 million ha remaining today (Gardiner and Oliver, 2005; The Nature Conservancy, 1992; King et al., 2006). LMV wetland loss rates exceed all other portions of the United States, creating an area of concern in terms of both wetland acreage and wetland functional losses (Hefner and Brown, 1995). During the 1970s and 1980s public and private organizations recognized the negative impacts of wetland functional degradation and began promoting wetland restoration designed to repair damaged and degraded ecosystems within the region (US Congress, 1985; Haynes et al., 1995; Hobbs and Cramer, 2008). In response, an estimated 275,000 ha of bottomland hardwood forest LMV has undergone reforestation, including over 20,000 acres under the jurisdiction of the U.S. Army Corps of Engineers (USACE, 1989; Allen et al., 2000; King et al., 2006; King and Keeland, 1999). Recently, the science and practice of ecological restoration has evolved to focus on maximizing ecological functionality within current biotic and abiotic constraints (Harris et al., 2006; Jackson and Hobbs, 2009).

Despite increases in wetland acreage resulting from large-scale restoration projects, no consensus exists regarding performance standards or early successional trajectory curves in forested systems (Thom, 1997; Ruiz-Jaen and Aide, 2005; Hughes et al., 2005). Recent work suggests measures of performance focus on vegetation composition, ecosystem processes, species diversity, and structural benchmarks (Gardiner et al., 2004; Wilkins et al., 2003; Hamel, 2003; Allen, 1997). However, calibration of appropriate methods for determining restoration

performance continues to lack clarity, specifically within the first few years following restoration (Steyer et al., 2003).

The time frames associated with forested wetland restoration complicate the establishment of performance standards (Hobbs and Harris, 2001; Kusler, 1986). Bottomland hardwood ecosystems require multiple decades to reach maturity, while regulatory agencies typically require less than a decade (commonly <5 years) of permit applicant sponsored post-project monitoring to determine restoration performance (Clewell and Lea, 1990; Landin and Webb, 1986). The temporal variability associated with ecosystem restoration remains problematic as few studies establish a restoration chronosequence exhibiting restored forest dynamics and functionality over time (Spencer et al., 2001).

In addition to the problems posed by forest successional changes, restoration trajectory is also influenced by the extent of site manipulation associated with restoring activities. For example many sites undergo plantings of ecologically desirable species (Stanturf and Gardiner, 2000; Humphrey et al., 2004), while other areas are subject to natural regeneration following clear-cutting or abandonment of previously farmed fields (Spencer et al., 2001; Battaglia et al., 2002). The amount of on-site preparation and changes to site hydrology and topography influence restoration outcomes, however the lack of an equal starting point for restoration complicates establishing performance standards. Often, responsible parties and agency staff are limited by budgetary and time constraints for post-restoration monitoring, compliance activities, and remediation of low quality restoration efforts.

The Hydrogeomorphic (HGM) Approach and other rapid assessment techniques examine wetland components to assess ecosystem function or condition (Brinson, 1993; Brinson et al., 1994; Stein et al., 2009). HGM has been widely applied because it specifically focuses on

requirements of the Clean Water Act and has been utilized to monitor many wetland ecosystem types (Brinson and Rheinhardt, 1996; Klimas et al., 2004; Humphrey et al., 2004). HGM collects data on a number of structural ecosystem components and applies multimetric equations to develop an index of wetland function or condition; providing a practical basis for evaluating wetland areas.

Kentula et al. (1992) and Zedler (1996) identified the need for establishment of performance standards or criteria for ecological restoration and mitigation projects. Further, Smith and Klimas (2002) and Klimas et al. (2004) examined expected recovery patterns within selected wetland assessment variables. The current work builds upon the available literature by 1) identifying rapid assessment variables that respond quickly following restoration, 2) developing statistically significant early stage restoration performance standards for reforested wetlands, and 3) providing examples of potential applications for restoration trajectories.

3.3. Methods

3.3.1 Study area

Study area selection was based on criteria including: 1) restoration project implemented within project relevant timescales (<20 years), 2) construction of a restoration chronosequence, 3) previous land use of 100% agricultural with no hydrologic restoration occurring onsite, and 4) located proximal to the region addressed by the assessment method developed for use in the study area. In order to minimize potentially confounding effects due to topographic location and hydrology, all selected study areas classified as riverine backwater wetlands as defined in Smith and Klimas (2002). Forty-five reforested sites ranging from 1-20 years post planting were examined during the study. The study area included sample plots located within the Yazoo Basin in Mississippi with one site located nearby in Louisiana (Table 3.1; Figure 3.1).

Table 3.1. Summary of site characteristics: location, area reforested, number of independent forests sampled, age, and condition

County, State	Area replanted (ha)	Forests sampled	Age (years)	Condition
Bolivar, MS	344	5	1	Restored
Ouachita, LA	1212	5	1	Restored
Bolivar, MS	1011	5	6-7	Restored
Quitman, MS	217	5	6-7	Restored
Washington, MS	140	5	6-7	Restored
Washington, MS	210	5	11-12	Restored
Washington, MS	186	5	11-12	Restored
Yazoo, MS	3499	10	20	Restored
Yazoo, MS	-	5	>80	Control
Sharkey, MS	-	21	>80	Control
Total	6819	71		

Study area age was determined by the dates of reforestation activities and historical documentation. Restoration activities utilized seedling planting and did not include hydrologic modification such as alterations to existing water control structures (e.g., ditches or levees). Planted species included a mixture of water oak (*Quercus nigra*), willow oak (*Quercus phellos*), Nuttall oak (*Quercus texana*), Shumard oak (*Quercus shumardii*), green ash (*Fraxinus pennsylvanica*), pecan (*Carya illinoensis*), and bald cypress (*Taxodium distichum*).

Twenty-six mature control sampling plots were also examined within the Delta National Forest. Control sites exhibited second growth forests >80 years old and represent the least disturbed forested wetlands in the region. Sample areas receive hydrologic inputs from precipitation and backwater flooding and occur within meander belts 2 and 3 of the Mississippi river floodplain (Saucier, 1994). Soils throughout the study area were characterized by Sharkey, Dowling, Perry, and Alligator poorly drained clay soils with small inclusions of somewhat poorly drained Commerce silty clay loam. All observed soil series phases were between 0-2 percent slope (Soil Survey Staff, 2011).

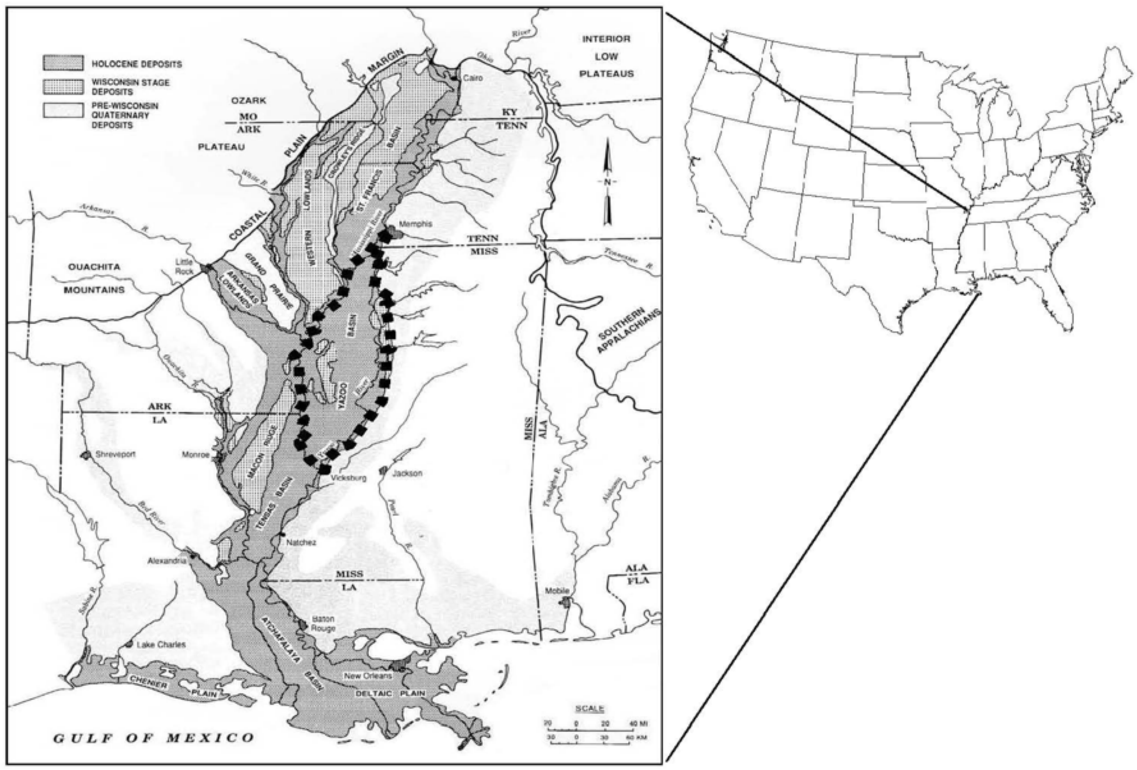

Figure 3.1. Study area highlighting the LMV. The Yazoo basin is outlined in black. (Saucier, 1994).

3.3.2. Selection of variables and data collection

The selection of variables was based upon the assessment protocols outlined in Smith and Klimas (2002) who developed an HGM guidebook specifically calibrated within the study area. The potential application of HGM variables as measures of restoration trajectory provides several advantages including: 1) data collection protocols are rapid (Berkowitz et al., 2011) and 2) utilize sampling measurements and protocols that resource professionals are familiar with (i.e., determination of tree diameter at breast height; Mack, 2007; Stander and Ehrenfeld, 2009). Further, the protocols provided in Smith and Klimas (2002) are currently applied as part of

ongoing monitoring efforts, providing an available source of data with the potential to produce science-based, applicable tools for developing restoration trajectories and performance standards.

Smith and Klimas (2002) identify seventeen variables commonly applied in wetland assessments. Variables included off-site and on-site measurements. Off-site variables evaluated flood regime, restoration site configuration, and the characteristics of adjacent properties. On-site variables included examination of soil characteristics, vegetative composition and vigor, and the degree of site disturbance (Table 3.2). Smith and Klimas (2002) provide detailed sampling instructions for each of the variables examined. Data collection occurred during the spring and early summer of 2011.

Additionally, measurements of onsite hydrology, river stage, precipitation, and soil carbon were collected. Within each study area, triplicate soil cores (10 cm deep) were homogenized and maintained at 4°C until total organic carbon was measured as loss on ignition of dried ground samples at 550°C in a muffle furnace for 4 hours (Sparks, 1996). Climate data reports daily precipitation values collected at the Vicksburg/Tallulah Primary Local Climatological Data Site (National Weather Service, 2012). River stage was determined within the center of the study area utilizing the Big Sunflower River gauge located at Holly Bluff, MS (USACE, 2012). Direct monitoring of near-surface hydrology utilized slotted groundwater wells installed 50 cm below the soil surface. One groundwater well was located at the center of each study area, establishing the hydrology within the immediate area represented by the HGM assessment. Water table level recordings were taken twice daily using Insitu Level TROLL 500 dataloggers (Ft. Collins, CO). Well construction, installation, and data analysis followed US Army Corps of Engineers (2005) specifications. The National Research Council (1995) defined high water tables as occurring within 30 cm of the surface, ensuring saturation or inundation

within the majority of the root zone. This represents the area considered critical for wetland functioning (Chorover et al., 2007). As a result, water table data are expressed as the number of high-water table days (i.e. the incidence of water tables >30 cm) occurring during the monitoring period (Berkowitz and Sallee, 2011).

3.3.3. Data analysis

Forest ages were combined into two year increments because planting periods vary between November and June. For example, forests restored 11 and 12 years ago were grouped together. One vegetation sample plot and associated sampling transects were located within each forest as outlined in Smith and Klimas (2002). Each sampled forest was treated as an independent sample; results report average values based upon forest age. Pearson Product Moment Correlations compare variables with restoration forest age. Where strong correlations (critical value $r > 0.418$, $p < 0.01$, n = 45) were observed within the first 20 years following restoration, significance between forest ages was determined by applying one-way ANOVA following testing for normality (Shapiro-Wilk test) and homogeneity of variance (Levene Statistic). The non-parametric Krustal-Wallis test was applied in cases where data was not normally distributed. Multiple comparisons analysis was conducted using Tukey HSD and LSD tests. Significance levels were evaluated at $\alpha = 0.05$ (SPSS, 2011).

3.4. Results and Discussion

Shrub-sapling density, ground vegetation cover, O horizon and A horizon thickness represent the only 4 of 17 variables measured within the assessment protocol that display significant correlations with restoration forest age (Table 3.3). The fact that several variables correlated with stand age shortly after restoration suggests a potential utility in evaluating linkages between variable outcomes and restoration site performance.

Table 3.2. Summary of rapid assessment variables, description, and sampling technique. Response variables (†) and rapid response variables (‡) are identified.

Rapid assessment variable	Description	Sampling technique
1. Core Area	Portion of wetland lying within 100m buffer	Measured from aerial photo/GIS layer
2. Habitat connections	Proportion of the wetland perimeter connected to suitable habitat	Measured from aerial photo/GIS layer
3. Wetland tract	Contiguous wetland area adjacent to the wetland	Measured from aerial photo/GIS layer
4. Flood frequency	Frequency of overbank or backwater flooding	Measured from flood frequency map/stream gauge data
5. Cation exchange capacity	Cation exchange capacity change due to soil disturbance	Estimated based on soil type
6. Soil integrity	Proportion of the wetland exhibiting altered soils	Estimated based on amount of soil disturbance visible
7. Micro-depressional ponding	Percentage of small topographic depressions and vernal pool features	Estimated based on percent of depressions within sample area
8. Tree basal area†	Basal area per hectare; proportional to tree biomass	Measured DBH of all trees > 7.6 cm in diameter within circular 0.04 ha plot
9. Tree density†	Number of trees per hectare	Count of all trees > 7.6 cm in diameter within circular 0.04 ha plot
10. Snag density†	Density of standing dead woody stems	Count of all snags > 7.6 cm in diameter within circular 0.04 ha plot
11. Tree composition	Species composition of the tallest stratum	Percent concurrence with measured tree quality index within the uppermost stratum
12. Woody debris biomass†	Volume of woody debris biomass per hectare	Count of nonliving stems along a 3.7 m transect
13. Log biomass†	Volume of log biomass per hectare	Count of logs along a 15 m transect
14. Shrub-sapling density‡	Density of saplings and shrubs per hectare	Count of all woody stems within two 0.004 ha plots
15. Ground vegetation cover‡	Percent cover of herbaceous and woody vegetation	Visually estimated percentage of ground covered with herbaceous and woody vegetation within four 1 m^2 plots
16. O horizon biomass‡	Mass of organic matter in the O horizon	Measured O horizon thickness
17. A horizon biomass‡	Organic matter accumulation in the A horizon	Measured A horizon thickness

As a result, these four variables were selected for additional analysis as potential indicators of restoration trajectory and performance within the first years following reforestation. Shrub-sapling density and ground vegetation cover were normally distributed ($F(3,45) = 32.6$ and $F(3,45) = 12.55$ respectively) while O horizon and A horizon thickness were not normally distributed ($F(3,45) = 76.4$ and $F(3,45) = 32.6$ respectively). In all cases when comparing variable outcomes to forest age, significant differences ($p < 0.01$) were observed at the $\alpha = 0.05$ level.

Table 3.3. Pearson Correlation outputs comparing rapid assessment variables to restoration forestage within 12[a]-20 years. With n=45, *r* > 0.418 is significant to *p* < 0.01[‡]

Variable	r	Variable	r
1. Core area	0.30	10. Snag density	0.01
2. Habitat connections	0.22	11. Tree composition	0.07
3. Wetland tract	0.24	12. Woody debris biomass	0.30
4. Flood frequency	0.32	13. Log biomass	0.24
5. Cation exchange capacity	0.01	14. Shrub-sapling density[a‡]	0.59
6. Soil integrity	0.01	15. Ground vegetation cover[‡]	0.64
7. Micro-depressional ponding	0.13	16. O horizon thickness[‡]	0.85
8. Tree basal area	0.03	17.A horizon thickness[‡]	0.89
9. Tree density	0.07		

All four variable outputs increase during years 0-12 following planting (Figure 3.1). Examining the entire restoration chronosequence, the soil A and O horizon variables show a distinctly different pattern than shrub-sapling density and ground vegetation variables. The depth of both O and A horizon variables increases throughout the restoration chronosequence. Because the figure displays both soil horizon depth and forest age, the slope between depth and age represents the rate of soil horizon change. The two soil horizon variables display a decrease in the rate of soil horizonation as restoration forests develop toward maturity. The findings agree with the findings of Groninger et al. (2000) who examined soil horizonation based on published

soil surveys and other off-site data. Results from the current study show that upper soil layer development continues following reforestation and suggests an asymptotic curve arises over longer time periods (>20 years). The construction of mid range and long term trajectory curves remains difficult because a paucity of data exists for restored agricultural wetland areas with intermediate ages between 20 years and maturity; due to the fact that restoration activities within the LMV only began in recent decades (King and Keeland 1999).

The observed increases in A and O horizon development prove measurable using rapid assessment techniques. The accumulation of soil organic matter within near-surface horizons has been linked with wetland hydrology (Reddy and DeLaune, 2008). Results of the current study demonstrate that hydrology within the study area remains driven by both precipitation and backwater flooding resulting from increased river stage [Figure 3.3(A)]. Further, increased total organic carbon concentrations occurred in areas exhibiting a large number of high water table events [Figure 3.3(B)]. Findings suggest that soil horizon development represents an appropriate and useful indicator of restoration trajectory and performance within project relevant timescales. The other two variables of interest, shrub-sapling density and ground vegetation cover, display a different pattern with forest age. The variables increase initially after restoration, followed by a sharp decline 15-20 years post reforestation (Figure 3.2). The timing of the change in variable corresponds with the development of tree succession and associated canopy closure approximately 15-20 years following restoration planting [Figure 3.4(A)]. Smith and Klimas (2002) predicted that variable responses would follow the observed patterns and display recovery curves of similar shape, although they provide no statistical data.

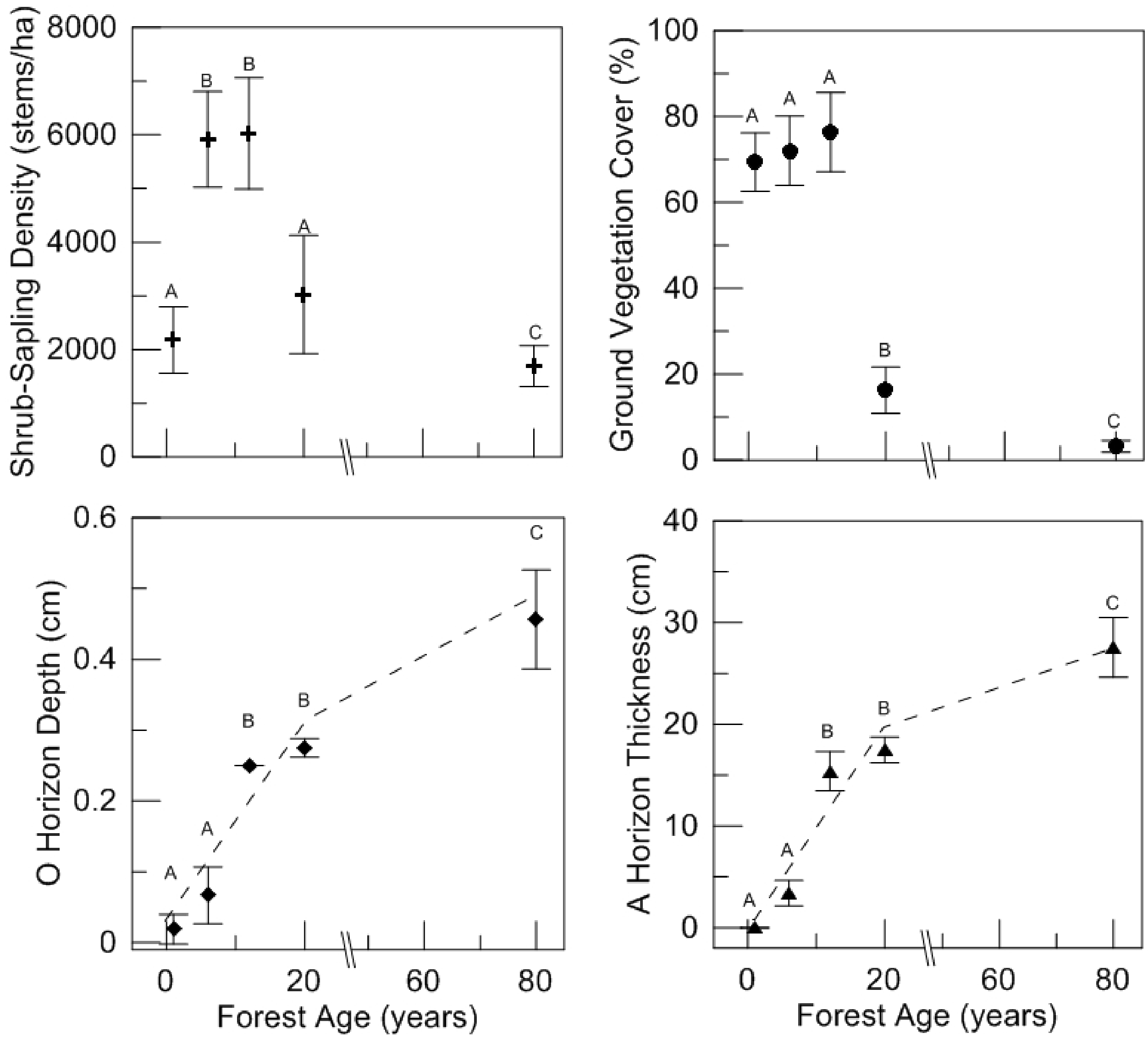

Figure 3.2. Observed changes in variable output with increasing forest age following reforestation and in mature bottomland hardwood forests (>80 years old). Error bars represent 1 standard deviation from the mean. Letters represent significant differences in multiple comparisons analysis between variable groups as determined by Tukey HSD and LSD tests. Note the broken axis between 20 and 40 years. Dashed lines represent the rate of O and A soil horizonation.

Additionally, other studies report a 20 year canopy closure threshold for planted oak species (*Quercus spp.*) in the LMV (Twedt and Portwood, 1997; Allen et al., 2000; Williams et al., 1997). Following the closure of the canopy, the onset of ground shading initiates the

observed decreases in shrub-sapling density and percent ground vegetation cover via light limitation (Allen, 1997).

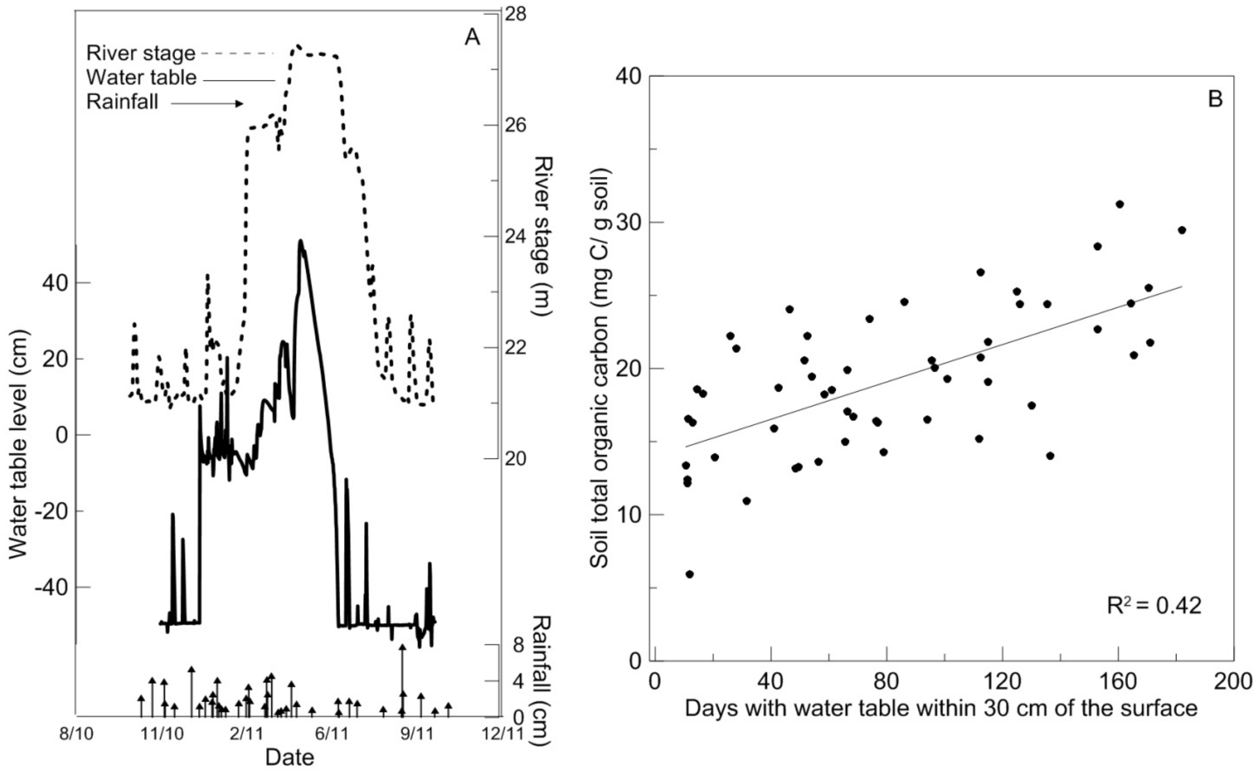

Figure 3.3. A) Rainfall, river stage, and ground water table observed at a representative study area. Note that restored backwater riverine wetland hydrology responds to a combination of precipitation and river stage. B) Correlation demonstrating the relationship between soil organic carbon concentration and study area hydrology as measured by the occurrence of high ground water tables (i.e., water level within 30 cm of the soil surface). Soil organic carbon and O horizon thickness increase with the occurrence of high water tables.

Results support the application of shrub-sapling density and ground vegetation cover as useful indicators of restoration trajectory and performance in the early years following reforestation. Also, the observed decreases in both variables following tree canopy closure suggest an additional benchmark for intermediate stage (mid-successional) restoration standards following approximately 20 years of restoration as observed in the break in slope within log transformed data [Figure 3.4(B)].

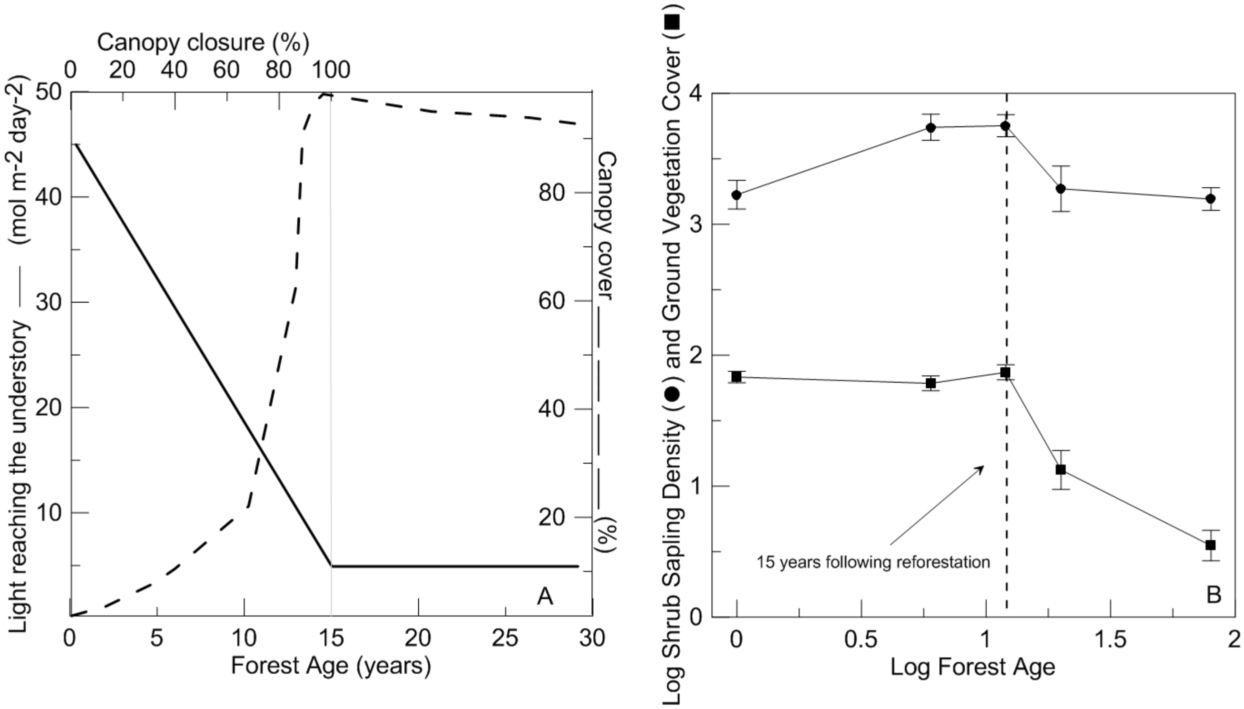

Figure 3.4. A) Canopy cover (dashed line) leads to light limitation (solid line), resulting in the shading out of understory plants occurring between 15 and 20 years following reforestation. Adapted from Bigelow et al. (2011) and Summers (2010). B) Log transformed data displaying threshold effect of forest age as observed in ground vegetation cover and sapling-shrub density data. Note the apex and subsequent decline in occurring at 15 years following reforestation.

The data presented above identifies four rapid assessment variables that respond quickly following restoration plantings. In order to develop early restoration trajectories and performance standards, efforts should focus on variables capable of determining whether a restoration project is on a pathway toward the desired outcome. Results examining sapling and shrub density shows a significant increase during the first 6-7 years following restoration, while the development of soil horizons required 11-12 years before measurable impacts were observed. Resource managers should incorporate specific, numerical increases in sapling and shrub density as a restoration milestone within mitigation requirements in the first years after restoration, followed by soil horizonation milestones in longer-term monitoring requirements. For example, the current study reports that sapling-shrub density should double over 6-7 years of restoration. Developing early

performance standards allows restoring entities to take corrective actions if needed (e.g., replanting or additional site modification) within project monitoring timeframes. Restoration performance standards remain unique for different ecosystems and regions, with the reported data applying to agricultural areas within the LMV undergoing bottomland hardwood wetland reforestation.

Thirteen variables evaluated as part of the rapid assessment failed to respond rapidly following forested wetland replanting. However, these variables play an important role in determining overall site condition in both young and mature ecosystems. The variables examined (Table 3.2) classify into three main categories: 1) rapid response variables with a high potential to change in the first years following reforestation, 2) response variables requiring additional time (e.g., >15 years) to display a measureable effect, and 3) stable variables that remain fixed over time. The current study establishes four rapid response variables. Response variables including tree density and basal area increase over time following 20 years of reforestation growth as suggested by Smith and Klimas (2002). On the other hand, stable variables such as flood frequency and the size of the wetland tract are not likely to change within project timescales.

Establishing three variable categories helps guide the development of restoration milestones and performance standards. However the fact that many variables remain essentially constant following restoration requires that managers maximize these variables through appropriate selection of restoration sites. For example, restoring entities should strive to create connected tracts of wetland area, plant appropriate vegetation, and consider both geomorphic position and landscape alterations affecting a given restoration project (Smith et al., 2008).

In situations where project goals include determining ecosystem conditional/functional change over time, resource managers should focus on variables that respond within the timescale of interest. When developing restoration trajectory curves and defining restoration performance milestones, additional emphasis must be placed on the subset of assessment variables that display rapid response and address both practical and ecological concerns. The combination of rapid response, response, and stable assessment variables characterize overall site conditions at longer timescales and all variables should be incorporated into restoration milestones as appropriate.

3.5. Conclusions

Determining the performance of restoration projects remains problematic due to the time required for forested wetlands to reach maturity, limited monitoring requirements, and a lack of coherent performance standards. Identifying measurable rapid assessment variables enables resource managers to establish early restoration milestones that examine the likely trajectory of a reforested area within project relevant timescales. Four rapid assessment variables showed strong correlations within recently reforested agricultural areas. Soil O and A horizon increased throughout the restoration chronosequence, providing direct relationships with forest age. Shrub-sapling density and ground vegetation cover increased in young restoration sites, followed by decreasing variable output with the onset of canopy closure, thus providing performance standards in both early and intermediate age forests. Assessment variables showing a rapid response following reforestation define early restoration trajectories and performance, allowing for corrective action within project relevant monitoring periods. When developing restoration trajectory curves and determining restoration performance milestones, emphasis should be placed on a subset of assessment variables that respond quickly and address both practical and

ecological concerns. Additionally, variables that respond slowly or remain stable over project timescales should be maximized through site selection and reforestation techniques.

CHAPTER 4: INVESTIGATION OF BIOGEOCHEMICAL FUNCTIONAL PROXIES IN HEADWATER STREAMS ACROSS A RANGE OF CHANNEL AND CATCHMENT ALTERATIONS

4.1. Abstract

Historically, headwater streams received limited protection and were subjected to extensive landscape alteration from logging, farming, mining, and development activities. Despite these alterations, these streams provide multiple essential ecological functions. This study examines proxy measures of biogeochemical function across a range of catchment alteration by tracking nutrient cycling (i.e., inputs, processing, and stream loading) with leaf litter fall, leaf litter bag decomposition, stream loading, and water quality parameters. Nutrient input and processing remained highest in older second growth forests (the least altered areas within the study region), while recently altered sample locations transported higher loads of nutrients, sediments, and specific conductivity. Biogeochemical cycling proxies of C and N input and processing significantly, positively correlated with the results of a rapid assessment approach (Pearson coefficient = 0.67–0.81; p = 0.002-0.016). Additionally, stream loading equations demonstrate that N and P transport, sediment, specific conductivity, and changes in temperature negatively correlated with rapid assessment scores (Pearson coefficient = 0.56-0.81; p = 0.002-0.048). Significant differences in nutrient processing, stream loading, water quality, and rapid assessment results were also observed between headwater streams located in recently altered (e.g., mined) and older second growth forested catchments (Mann-Whitney U = 24; p = 0.01-0.024). These findings demonstrate that biogeochemical cycling is reduced in altered headwater catchments, and indicate that rapid assessment scores respond to a combination of alteration type and recovery time. An analysis examining the time and economic requirements of

biogeochemical proxy measurements highlights the benefits of rapid assessment methods in evaluating biogeochemical functions.

4.2. Introduction

Headwater streams represent a major component of the riverine landscape, frequently accounting for half of total stream length in a catchment (Leopold et al., 1964). Historically, the abundance or headwater streams across the landscape, combined with the perception that 'dry' streams provide few ecological functions and benefits resulted in a lack of legal protection, assessments of ecological function, and resource management despite extensive degradation from logging, farming, mining, and development activities (Roy et al., 2009). However, recent studies confirm that headwater systems provide a number of ecological functions and benefits including organic matter transport, sediment capture, temperature regulation, aquatic and riparian habitat, and hydrologic regulation (Doppelt, 1993; Meyer and Wallace, 2001; Meyer et al., 2007; Wipfli et al., 2007). Few investigations examine biogeochemical functions in headwater streams. Noble et al. (2010) defined biogeochemical cycling functions as follows:

> The ability of the high gradient headwater stream ecosystem to retain and transform inorganic materials needed for biological processes into organic forms and to oxidize those organic molecules back into elemental forms through respiration and decomposition. Thus, biogeochemical cycling includes the activities of producers, consumers, and decomposers.

The cost, expertise, and time required to measure biogeochemical cycling functions remains beyond the scope of many management projects, resulting in the application of biogeochemical proxy measures (Stein et al., 2009). Based upon the definition above, three proxy measures of biogeochemical cycling were selected for investigation: 1) input of nutrients and organic materials into the headwater streams through leaf litter fall, 2) the processing of nutrients and other materials via leaf litter decomposition, and 3) the transport of nutrient and

other material loads by stream discharge. The following paragraphs outline the importance and use of these biogeochemical cycling proxies.

Leaf fall and decomposition represent major biologic contributors to forest and stream nutrient pools as documented extensively in the scientific literature (Bormann and Likens, 1967; 1995; Band et al., 2001; Webster and Benfield, 1986; Allan, 1995; Gessner et al., 1999). Ferrari (1999) reported that leaf litter fall accounted for 69 percent of total nitrogen contributions in forests, providing the dominant energy source to headwater streams which incorporate organic matter from terrestrial environments in the stream channel (Nelson and Scott, 1962; Hynes, 1963; Minshall, 1967; Cummins et al., 1973; Fisher and Likens, 1973; McDowell and Likens, 1988; Qualls, 2004; 2005). Investigating leaf litter fall and decomposition provides important insight into nutrient sources, and transformations occurring through biogeochemical cycling (Benfield, 1996; Aerts, 1997). The leaching and decomposition (i.e., processing) of leaf litter provides the basis of stream biogeochemical cycling and trophic transfer, and many studies employ leaf litter traps and decomposition bags (Petersen and Cummins, 1974; Benfield, 1996; Vitousek et al., 1994). Because leaf litter fall and processing occur via physical, chemical, and biological processes (Meyer, 1980), litter fall and litter bag processing studies represent appropriate proxy measures of biogeochemical cycling.

Anthropogenic and natural disturbances affect stream biota and ecosystem functioning directly or indirectly, impacting leaf litter availability, decomposition, water quality, and biogeochemical cycling (Gulis and Suberkropp, 2003). For example, Hagen et al. (2006) investigated leaf litter decomposition rates across a gradient of agricultural impacts, reporting that leaf breakdown rates were related to landuse category. Meyer (1980) compared leaf bag decomposition rates under high and low sedimentation regimes, and found that leaf breakdown

was lowest in areas subject to increased sedimentation. Atkinson and Cairns (2001) examined leaf decomposition rates within a chronosequence of restored ecosystems, indicating that leaf decomposition rates were higher in older, more mature restored locations. Based on the results of these studies and others, we hypothesize that nutrient inputs and processing should remain high in older, less altered areas and decrease in recently altered landscapes subject to surface mining, agriculture, and recent logging (Figure 4.1). Common alterations within the study area include agriculture, forestry impacts, suburban development, and surface mining (Hagen et al., 2006; Palmer et al., 2010). The majority of the study area has undergone historic alteration due to land clearing for farming, silviculture, and mining activities (Trimble, 1977; Adams et al., 2012). As a result, mature second growth forests represent the least altered landscape conditions across much of the region.

Several studies link natural and anthropogenic disturbance regimes to a change in the loading of materials within headwater stream systems (Lowrance et al., 1984; Osborne and Kovacic, 1993; Richards et al., 1996). Generally, findings suggest that higher levels of agriculture, urban, forestry and other alterations within headwater catchments increase nutrient and sediment loading to downstream environments (Gurtz et al., 1980; Likens et al., 1970). For example, Houser et al. (2006) reported relationships between the amount of alteration within a headwater catchment and amount of carbon, phosphorus, and nitrogen discharged from the system ($R^2 = 0.32 – 0.79$). As a result, examining the loading of elements and water quality measures in headwater stream ecosystems provides a proxy measure of biogeochemical cycling, and nutrient loading is hypothesized to increase with increasing levels of alteration and more recent alteration (Figure 4.1).

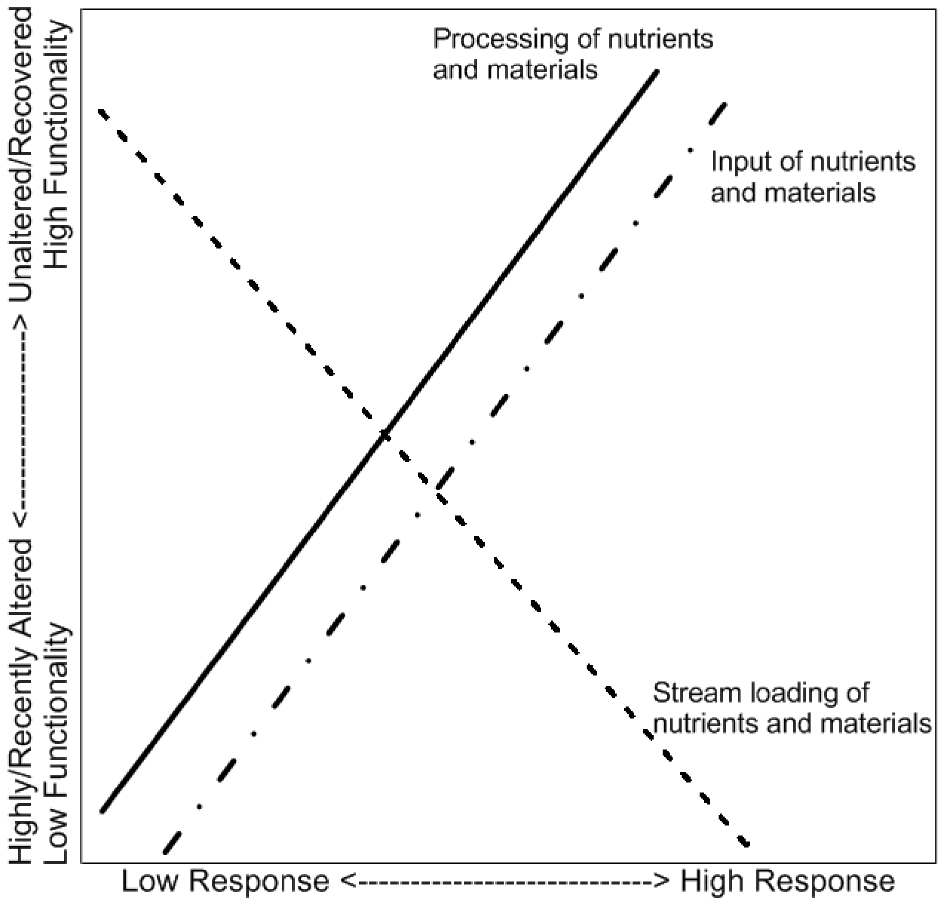

Figure 4.1. A conceptual model depicting biogeochemical cycling in headwater streams determined by material inputs, processing, and loading across a range of catchment alterations.

Rapid assessment approaches allow natural resource managers to quickly and inexpensively make and document decisions concerning potential impacts of development projects and to quantify elements of restoration projects (Whigham, 1999; Kentula, 2000). In the rapid assessment approach, easily attainable measurements are combined using simple multimetric equations to produce Functional Capacity Index (FCI) scores ranging from zero (a lack of function) to 1.0 (fully functional) (Smith et al., 1995). Recently, rapid assessment techniques have been developed for headwater streams because most traditional stream evaluation methods (i.e. benthic macroinvertebrate and water chemistry sampling) are constrained to the narrow windows of time when water is present in the channel, making them impractical for year-round

application (Mack et al., 2000; Berkowitz et al., 2011). Rapid assessment approaches utilize geomorphic, vegetation, and structural measurements that do not depend on the presence of water in ephemeral and intermittent stream channels, which remain dry during extended periods (Brinson, 1993, 1995; Rowe et al., 2009; Noble et al., 2010).

Although several studies compare selected functions with rapid assessment scores, few investigate biogeochemical functions or rapid assessments designed for use in streams (Stein et al., 2009; Bohonak and Bauder, 2011; Pohll and Tracy, 2000; Franklin et al., 2009; Berkowitz and White, In Press). The current study investigates whether rapid assessment outcomes provide reliable and practical tools for estimating biogeochemical cycling across a range of headwater stream catchment alterations.

While the application of rapid assessment approaches can result in a loss of analytical precision, economic constraints dictate the need for rapid and practicable methodologies (Turner, 1991; Stein et al., 2009). As a result, proxy measures of ecosystem function should incorporate economic considerations when possible (Turner et al., 2000). Therefore, the current study seeks to 1) examine biogeochemical function using proxy measures across a number of catchment alterations, 2) compare rapid assessment results with proxy biogeochemical measurements, and 3) compare the cost requirements of the two approaches outlined above.

4.3. Methods

4.3.1. Study sites

We collected data in West Virginia, USA, from March 2011 – July 2012 (Figure 4.2). All ten study sites met the definition of headwater streams described in Noble et al. (2010) with channel slopes exceeding 5% and water flowing during, and shortly after precipitation events. Groundwater may provide some water to the stream channel, however during dry periods,

headwater streams often lack flowing surface water (Federal Register, 2007). Study sites captured the various types of alteration observed in study area including impacts from agriculture, forestry, recreation and urban development, and surface mining (Hagen et al., 2006; Palmer et al., 2010). Because the vast majority of the study area exhibits historic alterations (e.g., land clearing for farming, silviculture, and mining) mature second growth forests represent the least altered landscape conditions across much of the region (Trimble, 1977; Adams et al., 2012; Table 4.1).

Table 4.1. Headwater stream catchment condition, characteristics, and biogeochemistry rapid assessment score.

Site	Stream/catchment alteration	Catchment area (ha)	Elevation (m)	Time since last alteration (yr)	Biogeochemistry rapid assessment score (FCI)
1	Forestry and recreation	1.03	265	25	0.89
2	Previously logged second growth forest	6.53	239	82	0.93
3	Previously logged second growth forest	4.13	248	94	0.97
4	Mining, constructed channel	9.13	247	13	0.72
5	Previously logged second growth forest	12.9	228	109	0.91
6	Mining, constructed channel	1.39	372	17	0.61
7	Agriculture pastureland	4.37	251	67*	0.62
8	Forestry	8.90	780	30	0.65
9	Previously logged second growth forest	1.28	726	103	0.9
10	Mining, constructed channel	3.12	274	12	0.29

*The agricultural pastureland is continuously grazed by cattle, but based on tree ring data, land clearing occurred 67 years ago.

Second growth, mature forested catchments contained intact channels exhibiting little erosion, a variety of particle substrate sizes, low stream particle embeddedness, and abundant large woody debris within the channel. Recent forestry and agricultural activities within the study area result in headwater channels characterized by increased sedimentation, erosion, and particle embeddedness as well as alteration to riparian areas (Hagen et al., 2006). Surface

mining activities have recently become common within the study area within the last three decades, including mountain top removal mining and valley fill (Palmer et al., 2010). These severe landscape alterations have resulted in the construction of engineered headwater stream channels characterized by large substrate particle substrates (i.e. rip-rap), little large woody debris, and sparse vegetation in riparian areas. The time period since the last watershed alteration was estimated based upon historical records and tree cores taken from riparian areas directly adjacent to each of the ten headwater stream reaches.

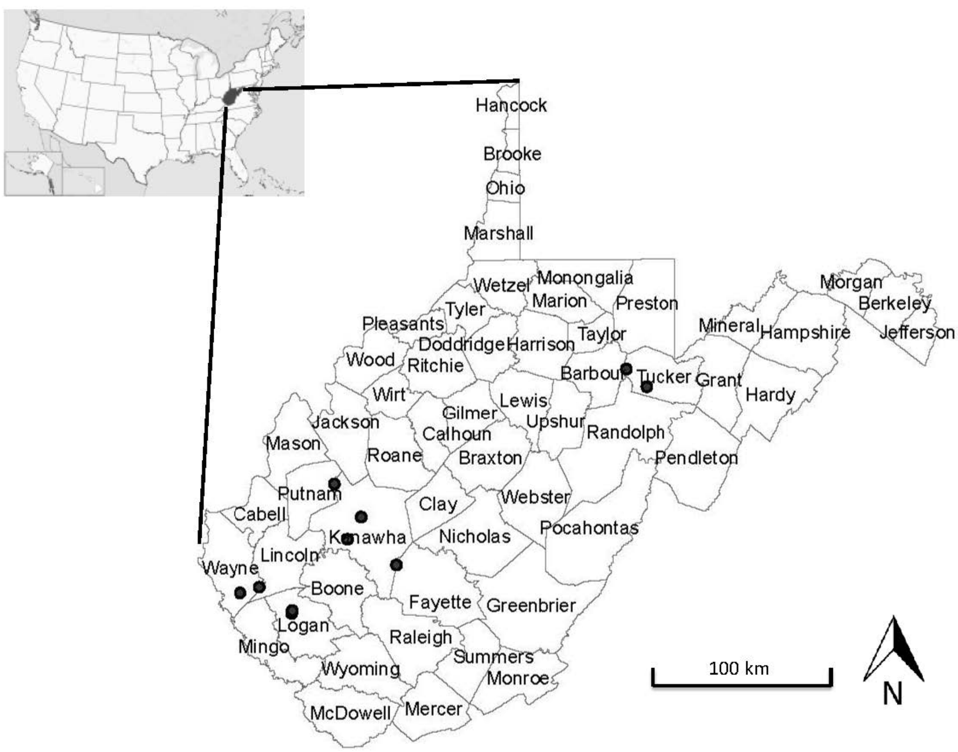

Figure 4.2. Map of the headwater stream study locations in West Virginia, USA.

4.3.2. Leaf litter fall and stream nutrient inputs, decomposition, and processing

Four large leaf traps (4.10 m^2) located adjacent to each headwater stream were sampled weekly from October 2011 – January 2012. Leaf material was collected, dried, and weighed. Samples underwent homogenization prior to analysis and litter was subsampled for total

moisture content (105 °C), ground, and analyzed for total nitrogen (N) and total carbon (C) concentration via combustion on an Elementar Vario Macro Carbon Nitrogen analyzer (900 °C) (Kahn, 1998; Klute, 1986). Total C and N inputs were calculated for the entire study period.

Leaf litter decomposition bags consisting of 1-cm mesh PVC hardware cloth and measuring 23 x 23 cm were constructed. All material used to fill the litter bags was collected at one sample location soon after abscission and thoroughly mixed to ensure an equal distribution of leaf species composition and nutrient content within bags across sample sites, thereby providing a consistent substrate from which to compare decomposition rates, (Gulis and Suberkropp, 2003; Hough and Cole, 2009). A variety of species were present: *Liriodentron tulipifera*, *Platanus occidentalis*, *Magnolia trepetala*, *Magnolia acuminata*, *Acer saccharum*, *Acer rubrum*, *Fagus grandifolia*, *Betula lenta*, *Carya cordiformis*, *Nyssa sylvatica*, *Tilia americana*, *Aesculus flava*, *Quercus rubra*, *Quercus prinus*, *Quercus alba*, and *Quercus velutina*.

Collected leaf litter underwent drying until constant weight was reached. Each bag received 30 g of dried leaf litter. Eight control litter bags underwent C and N analysis prior to deployment, facilitating the calculation of nutrient processing over the study period. Leaf litter bag placement consisted of distributing eight bags located in the riparian/buffer zone adjacent to each study location. Litter bags were placed on the ground surface and were not subject to inundation from stream water. The collection of replicate leaf litter bags occurred following 6 months (Harmon et al., 1999). Following collection, the leaf litter remaining in each bag was analyzed for C and N concentrations as described above. The decrease in leaf litter mass was determined by the change in dry weight after collection. Leaf litter C and N processing was measured by summing the total amount of C or N removed from replicate bags at each sampling location and dividing by the total amount of nutrients originally deployed at each sample site.

Several studies (Petersen and Cummins, 1974; Karberg et al., 2008) demonstrate that decreases in leaf litter nutrients cannot be completely attributed to decomposition (i.e., fail to differentiate between losses due to leaching, conversion to carbon dioxide through decomposition, and removal of leaf fragments by invertebrates). As a result, the data represents loss of leaf litter material or processing, providing a useful basis for comparison across the landscape alterations examined.

4.3.3. Stream loading and water quality

Surface water sampling occurred March 2011 – July 2012. Due to the ephemeral hydrology of the study sites opportunistic sampling took place whenever possible, resulting in the collection of approximately 30 water samples from each study site. Total ammonia was measured colorimetrically [Hach 8030; Detection Limit (DL) = 0.02 mg/L]. Nitrate and nitrite analysis utilized colorimetry following cadmium reduction (EPA 353.2; DL = 0.1 mg/L). Total inorganic nitrogen (TIN) was calculated by summing total ammonia, nitrate, and nitrite. Dissolved organic carbon (DOC) was analyzed using heated persulfate oxidation (SM5310C; DL = 0.50 mg/L). Total phosphorus (TP) analysis utilized the ascorbic acid method following digestion (AWWA 4500-P; DL = 0.01). Total suspended solids (TSS) were analyzed gravimetrically using pre-weighed glass fiber filters (SM 2540D; DL = 2.0 mg/L). Soil temperature measurements were recorded every 8 hours using TidbiT data loggers (Onset Corporation; Bourne, MA) placed 10 cm below the soil surface. Daily temperature ranges were averaged for the entire study period.

Headwater stream discharge measurements utilized Plasti-Fab extra-large 60 degree trapezoidal flumes (Tualatin, OR). Each flume included a recessed notch supporting an Aqua Troll 200 (Ft. Collins, CO) vented pressure transducer measuring water stream level and specific

conductivity every 15 min. Discharge calculations applied the depth-discharge equations supplied by flume manufacturers accommodating 0.0054 - 42.92 L/sec flows (Walkowiak, 2008). The calculation of stream loading has been applied to studies of stream water quality and biogeochemical cycling providing a method to estimate the total output of a component from a catchment (Vanderbilt et al., 2003; Campbell et al., 2004). Loading measures provide a more comprehensive measure of ecological cycling than traditional concentration measurements (Horowitz, 2008). Loading data accounts for catchment area and sample period, interpolating between instantaneous sample events to estimate stream loading when concentration data collection occurs infrequently (Hope et al., 1997; 1997b). The loading calculations applied the 'Method 5' equations of Verhoff et al. (1980) and Walling and Webb (1985) as recommended by Littlewood (1992) when continuous discharge data are available (Equation 1):

$$\text{Stream load} = [K\cdot Qr \cdot \textstyle\sum_{n}^{i=1}[CiQi]/\left(\sum_{n}^{i=1} Qi\right)]/A \qquad (1)$$

where K represents the period of record, Q_i is the instantaneous discharge at the time of sampling, Ci is the instantaneous analyte concentration, Qr is the mean discharge for the period of record, n is the number of samples, and A represents the headwater catchment area. Note that the stream loading equation accounts for study duration and provides outputs as abundance per area; when comparing to values published in other studies, results are converted to abundance per area per time.

4.3.4. Biogeochemical functional rapid assessment FCI score

This study combined five rapid assessment variables (embeddedness, detritus, tree diameter, large woody debris abundance, and watershed land use) representing the Biogeochemical Cycling function addressed within the Draft Regional Guidebook for the Functional Assessment of High-gradient Ephemeral and Intermittent Headwater Streams in

Western West Virginia and Eastern Kentucky (Table 4.2). Noble et al. (2010) provides detailed sampling instructions for each of the variables examined. The five variables underwent conversion into FCI scores via application of a multimetric equation (Table 4.2).

4.3.5. Economic analysis

Estimates of time and cost requirements to complete the biogeochemistry rapid assessment approach were based on data published by Berkowitz et al. (2011) who reported the need for only 1 site visit consisting of three hrs of field time and three hours of analysis. Cost estimates for completion of the nutrient input, processing, and stream loading parameters described above were based on the average number of water samples collected (30 visits to each sample site) and the average time required to sample stream water, download field data loggers, and conduct periodic maintenance on equipment. All labor costs were estimated at the rate of a current graduate research assistant earning $30,000 USD per year (i.e., $14 USD per hour). Equipment cost estimates were based on actual costs of the instrumentation described above. The cost of sample analysis (including laboratory time and supplies) is estimated based upon university cost center pricing. In order to ensure that the cost estimate remains conservative, we do not estimate the labor cost associated with equipment installation, site selection, or analysis and post processing of study results.

4.3.6. Statistical analysis

Statistical procedures included normality testing applying the Shapiro-Wilk test followed by Pearson Product Moment Correlation and linear regression analysis. In cases where data deviated from normal distributions, Spearmans Rank Order Correlation was applied.

Table 4.2. Summary of rapid assessment variables, description and rationale for selection, and the Biogeochemistry rapid assessment equation applied. Modified from Noble et al. (2010).

Assessment variable	Description and rationale for selection
1. Stream particle embeddedness (V_{EMBED})	Embeddedness estimates the degree to which coarse substrates are covered, surrounded, or buried by fine sediments, indicating erosion and movement of fines (Chang, 2006).
2. Stream and riparian large wood (V_{LWD})	Abundance of large wood within stream and riparian area. Large wood influences the movement, storage, and addition of organic matter into streams (Hilderbrand et al., 1997) and is a source of particulate organic matter (Fischenich and Morrow, 2000).
3. Stream and riparian area detritus cover ($V_{DETRITUS}$)	Abundance of detrital material covering the riparian and stream channel surface. Generation of dissolved nutrients from detritus is a result of chemical leaching of soluble compounds, nutrients released during microbial breakdown, and release occurring during invertebrate feeding on decaying detritus (Meyer and O'Hop, 1983). Nutrient cycling and availability decrease when detritus is absent from the stream system (Wallace et al., 1997).
4. Riparian tree diameter at breast height (V_{TDBH})	Size and abundance of trees in the riparian zone. Trees adjacent to headwater streams affect lighting, temperature, nutrient cycling, promoting stream stability and water quality (Hedman et al., 1996; Osborne and Kovacic, 1993). Leaf litter provides a major energy base for low-order headwater streams (Benfield et al., 1991).
5. Watershed landuse (V_{WLUSE})	Area weighted average of land-use classifications occurring within the headwater catchment. Land-use affects runoff quantity and quality (Jones et al., 2001; Rheinhardt et al., 2009). With catchment alteration, more surface water travels downstream increasing sediment, nutrient loading, and impacting water quality (Simmons et al., 2008; DeFries and Eshleman, 2004).

$$FCI = \left\{ V_{EMBED} \times \left[\frac{\left(\frac{V_{LWD} + V_{DETRITUS} + V_{TDBH}}{3} \right) + V_{WLUSE}}{2} \right] \right\}^{1/2}$$

The non-parametric Independent Sample Mann-Whitney U test examined differences between older second growth forested and recently altered study sites subject to agricultural, surface mining and recent logging impacts (Table 4.1). All significance was evaluated at α =

0.05. (SPSS IBM, Inc. Version 20). Within the water quality dataset three outliers were removed based on the results of Grubbs' test ($p < 0.05$) (Iglewicz and Hoaglin, 1993). Within the inorganic nitrogen loading data, one highly altered area subject to mining and valley filling, displayed levels >12 times higher than all other study sites. Within the total phosphorus loading data an altered area subject to ongoing agricultural inputs including cattle directly accessing the stream above the water sampling location, displayed levels >74 times higher than all other study sites. Within the maximum specific conductivity data, one highly altered area subject to mining and valley filling, displayed conductivity values >4 times higher than all other study sites. Error propagation within the rapid method was examined at three study sites based on multiple comparison data collected by independent field teams as outlined in Berkowitz et al. (2011). The analysis resulted in generation of vertical error bars for three of the study sites examined.

4.4. Results and discussion

4.4.1. Leaf litter fall and stream nutrient inputs

Leaf litter fall N inputs ranged from 0.29 to 3.87 g N/m^2. N inputs were highest in areas with older second growth forests; lower N inputs occurred in areas where recent disturbance resulted in stream catchment displaying decreased or complete lack of tree canopy. Comparisons of leaf litter N input with rapid assessment biogeochemical cycling FCI scores yielded significant positive correlations [r = 0.78, n = 10, p = 0.008; Figure 4.3(A)]. Leaf litter C inputs ranged from 15.2 to 163 g C/m^2. These results compare with data compiled by Petersen and Cummins (1974) who report daily organic matter inputs to small streams between 0.97 g/m^2/day and 4.2 g/m^2/day. Our study sites averaged inputs of 1.57 g/m^2/day (range = 1.26 – 1.81 g/m^2/day) for forested locations. Comparisons of leaf litter C input with results of the rapid assessment biogeochemical cycling FCI scores yielded significant positive correlations [r = 0.81,

n = 10, p = 0.005; Figure 4.3(B)]. Leaf litter fall results support the hypothesis that recent and severely altered locations receive smaller total nutrient inputs (Figure 4.1). Although the decrease in nutrient inputs with alteration and low forest cover is intuitive, the availability of nutrients and other materials for processing, transformation and export remains an important aspect of biogeochemical functions in headwater streams (Noble et al., 2010). Since leaf litter fall represents the major source of materials to headwater streams, the observed decreases limit biogeochemical cycling in recently altered catchments compared to older forested locations.

4.4.2. Leaf litter bag decomposition and processing

Leaf litter mass decomposition over the six month sampling period ranged from 23.1 to 61.7%. These results compare well with the findings of Gosz et al. (1973) who found approximately 30% leaf litter (*Acer saccarum*) decomposed following 6 months of exposure in a New Hampshire forest. Additionally, Gingerich and Anderson (2011) report mass decomposition (*Typha latifolia*) of 45.4 - 56% over a period of one year in litter bags in West Virginia headwater riparian areas. Comparisons of leaf litter mass decomposition with results of the rapid assessment biogeochemical cycling FCI scores yielded significant positive correlations ($r = 0.64$, $n = 10$, $p = 0.024$; data not shown). Results demonstrate that an average of 27% more leaf litter processing occurred within second growth forested catchments, compared to headwater streams subject to recent agriculture, surface mining, and logging activities ($U = 24$, $p = 0.01$). These results agree with the findings of Simmons et al. (2008) who also reported decreased decomposition occurring at altered locations, including catchments affected by surface mining. Leaf litter bag N processing ranged from 3.2 to 27.4%. Simons and Seastedt (1999) reported up to 38% N release over 225 days for *Populus deltoides* riparian litter. Comparisons of leaf N

processing with results of the rapid assessment biogeochemical cycling FCI scores yielded significant positive correlations [r = 0.81, n = 10, p = 0.002; Figure 4.3(C)].

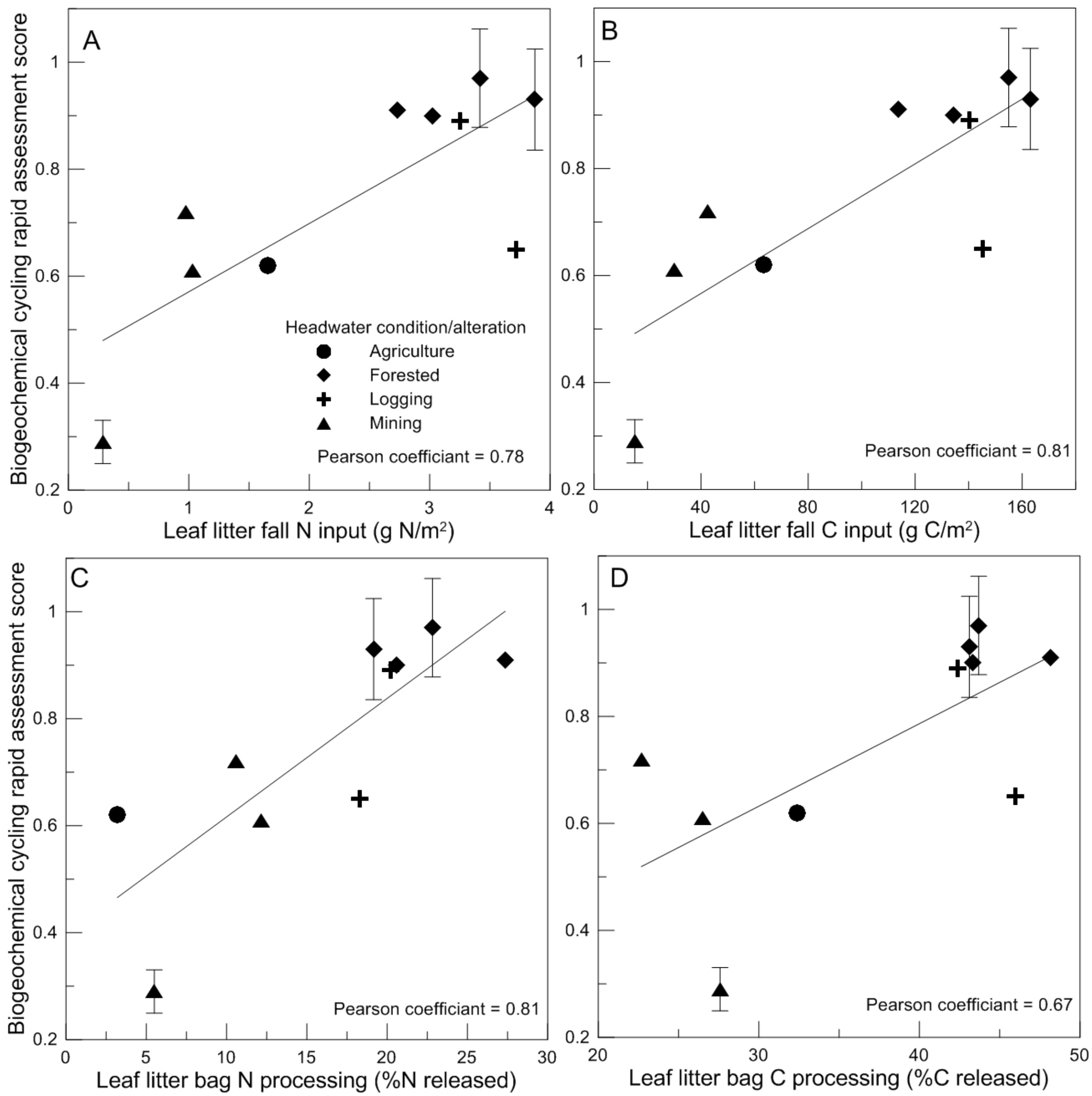

Figure 4.3. Comparison of A) leaf litter fall N and B) C input and C) leaf litter bag N and D) C processing to biogeochemical rapid assessment score across a range of headwater stream alteration. Regression lines represent all data points regardless of headwater condition. Vertical error bars display error propagation within the rapid assessment equation for three selected study sites (Berkowitz et al., 2011).

Similarly, leaf litter bag C processing ranged from 22.7 to 48.1% with an average of 37.6%. As discussed previously, the litter bag decomposition study does not address the fate of processed C, with possible outcomes including respiration as CO_2, removal as particulate organic matter, or leaching as DOC (Petersen and Cummins, 1974). Comparisons of leaf litter C processing with results of the rapid assessment biogeochemical cycling FCI scores yielded significant positive correlations [$r = 0.67$, $n = 10$, $p = 0.016$; Figure 4.3(D)].

Results from the leaf litter bag decomposition study demonstrate that although equal amounts of leaf litter was placed at each study site, second growth forested areas processed more materials than recently altered areas. Leaf litter N and C processing was significantly higher in second growth forested study areas compared to altered areas subject to recent agriculture, surface mining, and logging activities ($U = 24$, $p = 0.01$). Results support the conceptual model; not only do recently altered headwater streams and associated riparian areas receive fewer nutrients and other materials, they also process less nutrients (Figure 4.1). The decrease in leaf litter processing and nutrient release suggests a shift in the size and composition of the drivers of biogeochemical cycling (e.g., invertebrate, microbial, and fungal decomposer communities). Similar results were reported within the study area by Simmons et al. (2008) documenting decreases in soil nutrients and shifts in microbial communities, affecting biogeochemical functions following land alteration including surface mining activities. The observed agreement between the biogeochemical proxy data presented above and the biogeochemical cycling rapid assessment function demonstrates that the rapid approach represents a useful measure of nutrient inputs and processing across a number of landscape alterations.

4.4.3. Stream loading and water quality

In addition to the input and processing factors previously described, stream loading in headwaters streams depends on biogeochemical processes controlling solute concentration, providing nutrients and other materials to surface waters, and potentially impacting water quality. Landscape alterations including surface mining, agriculture, and forestry activities affect biogeochemical cycles by changing vegetative composition, microbial activity, temperature and moisture regimes, and other factors (Kreutzweiser et al., 2008; Simmons et al., 2008). Landuse alterations also influence hydrologic linkages resulting in increased sedimentation, erosion, flashiness, and introduction of nutrient laden particles; factors impacting water quality (Lindberg et al., 2011). In the current study, inorganic N loading ranged from 0.92 – 75.3 kg/ha with an average of 9.8 kg/ha [Figure 4.4(A)]. Observed values correspond with the findings presented by Groffman et al. (2004) and Simmons et al. (2008) who reported values between 0.11 and 5.32 kg N/ha/yr in forested watersheds. This study had values ranging from 0.37 to 3.1 kg N/ha/yr for second growth forested watersheds with recently altered areas reaching loading values up to 50 kg N/ha/yr.

Comparisons of TIN loading with predicted results of the rapid assessment biogeochemical cycling FCI score yielded significant negative correlations ($r = -0.70$, $n = 9$, $p = 0.019$). Simmons et al. (2008) showed a different relationship, reporting a decrease in N loading to surface waters following clearcutting and surface mining; suggesting that N-limitation within the disturbed watershed studied was limiting N mobility. However, several studies demonstrate increased in N loading in recently altered catchments similar to what is reported here.

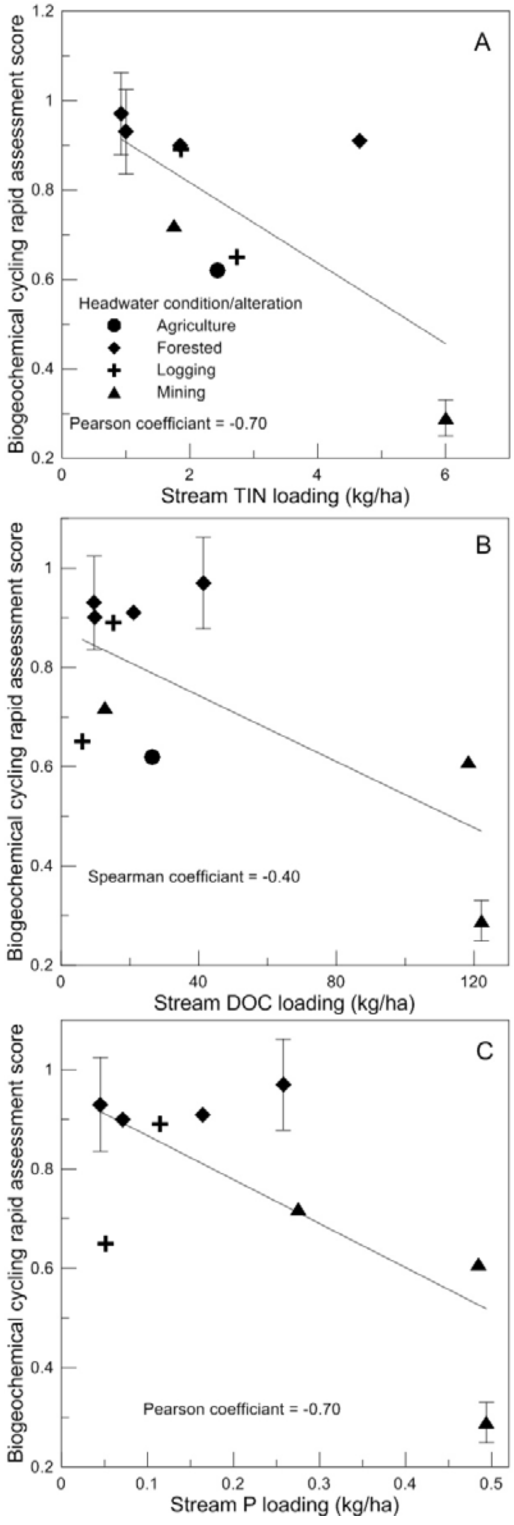

Figure 4.4. Comparison of A) inorganic N, B) DOC, and C) P loading in headwater streams exhibiting a range of catchment alteration and comparison to biogeochemical rapid assessment score. Regression lines represent all data points regardless of headwater condition. Vertical error bars display error propagation within the rapid assessment equation for three selected study sites (Berkowitz et al., 2011).

Schmidt et al. (1996) showed that alterations due to logging leads to elevated N availability (e.g., increased N mineralization and nitrification), while others document elevated N loading to receiving waters following landscape alterations (Lamontagne et al., 2000; Carignan et al., 2000; Steedman, 2000). The current study did not measure atmospheric contributions of N, however Gilliam et al. (1996) reports total N deposition of 15 kg N/ha/yr across the study area with larger deposition rates observed at higher elevations. The current dataset does not show a relationship between elevation and TIN loading, suggesting that regional variations in atmospheric N deposition are not significantly driving the observed relationships between catchment alteration and TIN loading (Table 4.1; Figure 4.4).

DOC loading ranged from 6.2 – 122 kg/ha. Data were not normally distributed. Comparisons of DOC loads with predicted biogeochemical cycling FCI scores were not significantly correlated (r_s = -0.39, n = 10, p = 0.130), and the observed relationship was dominated by two highly altered areas subject to valley filling and previous or ongoing mining activities [Figure 4.4(B)]. Despite the lack of statistical significance, a three-fold increase in DOC loading was observed, on average, between forested and highly altered areas. These findings agree with the results of several studies reporting a two to five-fold increase in C export following recent landscape alterations including logging activities (Carignan et al., 2000; Steedman, 2000; Lamontagne et al., 2000). Simmons et al. (2008) also reported increases in C loading following surface mining, exporting an average of 2.5 times more C than adjacent forested watersheds. Increased DOC loading can be attributed to changes in moisture regimes, increased soil and stream channel temperatures (Figure 4.5), reduced soil C storage, and increased leaching of debris resultant from forestry, mining, or agricultural activities.

Total P loading ranged from 0.045 – 36.8 kg/ha. TP loading significantly differed between second growth forested catchments and areas subject to recent agricultural, logging and surface mining activities ($U = 24$, $p = 0.024$). Comparisons of TP loadings with predicted results of the biogeochemical cycling FCI score yielded significant negative correlation [$r = -0.66$, $n = 9$, $p = 0.026$; Figure 4.4(C)]. TP loading rates measured in the current study averaged 92 g/ha/yr in second growth forested catchments and 318 g/ha/yr in mined areas; Simmons et al. (2008) reported similar results with TP loading increasing from 80 g/ha/yr to 156 g/ha/yr in forested and mined watersheds respectively. Because TP loading is driven by the weathering of primary minerals and the mineralization of organic P, landscape alterations that disturb mineral substrates and introduce organic debris into headwater catchments can lead to increased TP loading to surface waters (Kreutzweiser et al., 2008). Surface mining, logging, and agricultural (e.g., livestock) operations also result in exposed mineral surfaces that can facilitate TP loading to streams through mineral weathering and the transport TP associated with sediment particles.

Total suspended solids flux ranged from 22.2 – 375 kg/ha (Figure 4.5). Comparisons of TSS loading with predicted results of the biogeochemical cycling rapid assessment FCI score yield significant negative correlation [$r = -0.56$, $n = 10$, $p = 0.048$; Figure 4.5(A)] and TSS values were significantly higher in recently altered areas ($U = 24$, $p = 0.01$). Similar increases in sedimentation with alterations due to mining are described by Pond et al. (2008) who reported elevated sediment deposition in headwater catchments containing surface mining activities; Simmons et al. (2008) demonstrated a three-fold increase in daily sediment concentration between mined and unaltered watersheds. Waters (1995) discusses the impacts of other landscape alterations (e.g., farming) on sedimentation rates and the effects on water quality including P loading to streams. TP loading was correlated with TSS loading ($r = 0.84$; data not

shown), suggesting that the measurement of TSS may serve as an rapid and cost effective estimate of P loading within the study area.

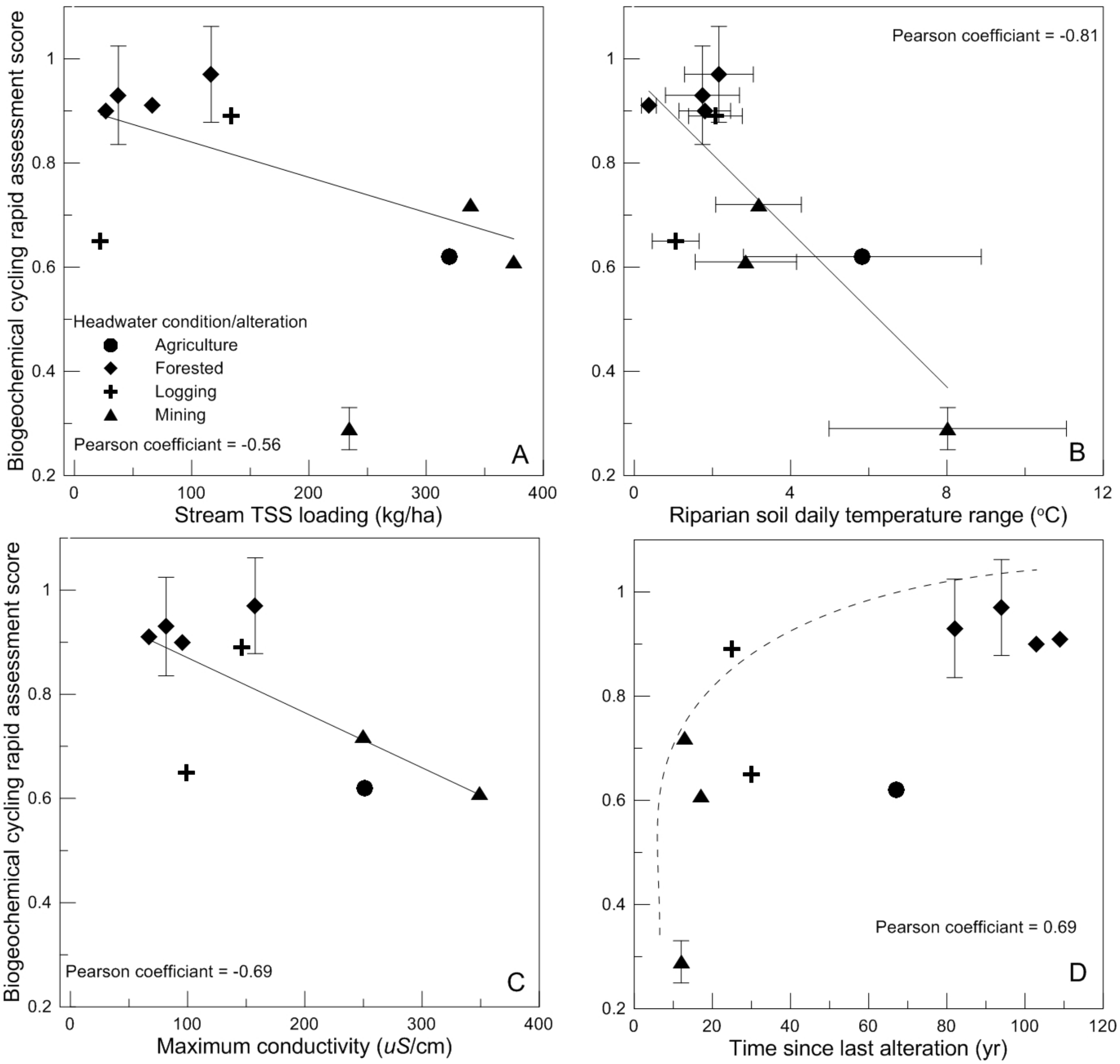

Figure 4.5. Comparison of A) TSS loading, B) riparian soil temperature range, C) maximum conductivity, and D) the time since the last alteration occurred in headwater streams exhibiting a range of catchment alteration to biogeochemical rapid assessment score. Regression lines represent all data points regardless of headwater condition. Horizontal error bars represent one standard deviation within temperature data. Vertical error bars display error propagation within the rapid assessment equation for three selected study sites (Berkowitz et al., 2011).

Mean daily temperature ranges observed within riparian soils (10 cm depth) at each study site varied between 0.37 and 8.0 ^{o}C [Figure 4.5(B)]. Temperature range data negatively correlated with the biogeochemical cycling FCI score ($r = -0.81$, $n = 10$, $p = 0.002$) and temperature ranges were significantly higher in catchments subject to recent agricultural, logging and surface mining activities ($U = 24$, $p = 0.01$). Temperature data collected within stream channels showed similar results (data not shown). Several published studies link changes in temperature regimes to disturbances within headwater stream catchments, demonstrating increased temperature ranges in areas exhibiting silvicultural, agricultural, urban, and surface mining impacts (Swift and Messer, 1971; Rischel et al., 1982). Simmons et al (2008) reported that average temperature in mined headwater catchments increased 3.5 ^{o}C during summer and a decreased 7 ^{o}C during winter compared to unaltered areas. Larger temperature ranges are observed in recently altered areas are attributed to a lack of canopy shading and insulation from vegetative cover. A number of biogeochemical processes relating to decomposition and nutrient cycling exhibit temperature dependence (Allan, 1995) with changes in temperature due to disturbance impacting metabolic process rates and microbial community assemblages (Dodds, 2002).

Conductivity values observed throughout the study ranged from 18 – 1670 μS/cm with an average of 199 μS/cm. Evaluating average conductivity values provides limited utility, especially in headwater streams that periodically lack flowing water. As a result, maximum values are often examined in order to determine environmental thresholds (i.e., the limit of biological tolerance; USEPA, 2011). Maximum conductivity values ranged from 67 – 1458 μS/cm [Figure 4.5(C)]. Lindberg et al. (2011) reported maximum conductivities of 253 μS/cm in unaltered Appalachian headwater streams, while catchments impacted by surface mining

ranged from 502 – 2540 μS/cm. Comparisons of maximum conductivity with predicted results of the biogeochemical cycling FCI score yielded significant negative correlations ($r = -0.69$, $n = 9$, $p = 0.019$) and maximum conductivity values were significantly higher in recently altered catchments ($U = 24$, $p = 0.024$).

4.4.4. Time since alteration

Time is not explicitly included as a factor within the rapid assessment approach. However, because rapid assessment variables recover following a landscape alteration, temporal factors are implicitly included into assessment scores (Detenbeck, 1996). For example, the diameter of trees increases with time and contributes to higher rapid assessment scores. As a result, a built-in time scale is included in the rapid assessment [Table 4.2; Figure 4.5(D)]. Results indicate that recovery time and alteration type both drive rapid assessment scores. For example, catchments subject to surface mining display a wide range of rapid assessment scores (FCI = 0.29 – 0.72) within a narrow 5 year range since last alteration occurred, demonstrating the variability within that alteration type. Conversely, second growth forested areas displayed rapid assessment scores between 0.9 and 0.97 across a much larger age range (27 yr). Results suggest that increases in functional improvement slow as a study area recovers from disturbance (i.e., diminishing returns are observed in older sites). Klimas et al. (2011) and reported similar results in the form of projected recovery trajectories for a rapid assessment method examining bottomland hardwood forests within the Mississippi Valley. The development of projected recovery trajectories based on the current dataset remains difficult due to the novel mountaintop mining/valley fill techniques and the extent of landscape alterations being applied within the region (Palmer et al., 2010).

4.4.5. Economic analysis

As discussed above, economic constraints dictate the need for rapid and practicable methodologies (Turner, 1991; Stein et al., 2009) and measures of ecosystem function should incorporate economic considerations (Turner et al., 2000). The data presented above demonstrates that biogeochemical cycling FCI scores provide a useful measure of nutrient inputs, processing, stream loading, and water quality; FCI scores remained significantly lower in altered areas ($U = 24$, $p = 0.019$) compared to forested study reaches. Further, Berkowitz et al. (2011) reported that the rapid assessment approach requires only one site visit, providing a practicable application for resource managers, regulatory personnel, and the academic community. The cost of the rapid assessment is easily estimated at $184 USD (2013 dollars), based on a six hrs of work (three hrs onsite evaluation and three hrs of offsite preparation and analysis), plus the cost of equipment (Table 4.3). The rapid assessment cost remains within the scope of most resource management and evaluation projects. Alternatively, estimating the cost of measuring the nutrient inputs, processing, stream loading, and water quality parameters pose challenges in terms of determining the actual time required for field and laboratory analysis as well as equipment installation and maintenance. As a result we provide a conservative estimate of $5,295 for the 1.5 year field study including labor (30 site visits, two hours per site visit), equipment, and laboratory analytical costs. This cost remains beyond the scope of typical, site specific resource management efforts.

Economic cost and time estimates provide perspective on the utility of the rapid assessment approach, and the contrast in time and economic cost requirements remains clear. In the current study, the rapid assessment approach required 10% of the time and < 3.5% of the economic costs compared to collecting hard data on nutrient inputs, processing, and stream

loading. Additionally, an examination of error propagation conducted at three study sites demonstrates that the rapid assessment method displayed error between 4 and 9% (Figures 4.3, 4.4, and 4.5). While Turner et al. (1991) and others point out the potential shortcomings associated with rapid ecological assessment methods, results from this study demonstrate that the rapid assessment approach provided a useful proxy measure of biogeochemical processes. Although not all of the parameters examined displayed significant correlation with the biogeochemical rapid assessment score (e.g., DOC loading), the fact that multiple lines of evidence provide linkages between rapid assessment results and proxies of biogeochemical cycling (leaf nutrient inputs, litter bag processing, stream loading and water quality) assists in validating the rapid assessment methodology (Stein et al., 2009). Further, the contrast of economic and time requirements for both the field measurement and rapid assessment methodologies demonstrates the strength and applicability of rapid assessment approaches, especially when they can be validated, verified, and calibrated based on more intensive data collection efforts.

4.5. Conclusion

The landscape alterations examined significantly affected biogeochemical proxies including nutrient inputs, processing, stream loading, and water quality. Headwater streams impacted by surface mining, agriculture, and contemporary logging activities received fewer nutrients and less nutrient processing occurred compared to older second growth forests. Also, altered headwater catchments displayed higher levels of nutrient, sediment, and conductivity loading than forested areas. Overall, recently altered areas received and processed less nutrients while transporting higher nutrient loads; this indicates that biogeochemical cycling was decreased in altered locations.

Table 4.3. Cost estimates to complete the biogeochemical rapid assessment approach and biogeochemical proxy measurements of nutrient inputs, processing, loading (2013 US dollars).

Rapid assessment approach						
Labor cost		Hourly rate	On site hrs	Offsite hrs	Site visits required	Total
		$14	3	3	1	6 hrs
						$84
Equipment costs			Measuring tape, flagging, tools			
			$100			$100
Total rapid assessment cost per site						$184
Total rapid assessment cost – all sample sites						$1,840

Biogeochemical proxy measurements						
Labor cost	Hourly rate		Onsite hrs	Offsite hrs	Site visits required	Total
			2	--		60 hrs
	$14				30	$840
Equipment costs	Discharge and temperature loggers		Trapezoidal flume		Leaf bags, tools	
	$900		$875		$100	$1,875
Laboratory costs	Analyte:	DOC	TIN		TP	
	Price per sample:	$15	$15		$20	
	Number of samples:	30	30		30	
	Subtotal:	$450	$450		$600	
	Analyte:	TSS	Conductivity		Leaf bag C and N	
	Price per sample:	$15	$1		$20	
	Number of samples:	30	30		30	
	Subtotal:	$450	$30		$600	$2,580
Biogeochemical proxy cost per site						$5,295
Biogeochemical proxy cost – all sample sites						$52,950

The rapid assessment provided a useful tool for estimating the level of biogeochemical cycling with assessment scores driven by a combination of alteration type and recovery period. However, challenges remain in determining the long term impact of recent landscape alterations on biogeochemical cycling, including novel impacts from surface mining utilizing mountaintop mining and valley fill. An economic analysis highlighted the utility and cost effectiveness of rapid assessment approaches, especially in cases where biogeochemical proxy data supports rapid assessment results.

CHAPTER 5: TECHNICAL STANDARD FOR THE DEVELOPMENT AND APPLICATION OF RAPID ECOLOGICAL ASSESSMENTS

5.1. Abstract

Over 60 different rapid ecological assessment approaches are currently in use throughout the United States. The lack of coherent standards for wetland and stream assessment method development and application has resulted in recent debate, with newly implemented approaches potentially increasing project cost and mitigation requirements. This document describes a technical standard for rapid ecosystem assessment methods used to evaluate the condition or function of ecosystems including wetlands and streams for the purposes of determining mitigation requirements, estimating impacts resulting from project implementation, quantifying the success of restoration efforts, and other applications. The standard outlines the principle components required to promote technically valid, science-based, defensible, and transparent approaches to rapid ecosystem assessment. The technical standard consists of nine components including: 1) Interagency development team, 2) Clearly defined assessment goals, 3) Ecosystem classification, 4) Geographic extent, 5) Rapid application, 6) Collection of quantitative, numerical data, 7) Calibration based on data, 8) Peer review, and 9) Applicable assessment outputs. Approaches for improving existing assessment methods that currently fail to meet the technical standard are also included. Each component represents a testable standard capable of evaluating current approaches and guiding the development of new rapid assessment protocols.

5.2. Introduction

Wetland and stream ecosystems support a number of well established biological, chemical, and hydrologic conditions and functions linked to ecosystem services that prove beneficial to society (Novitski et al., 1996; Smith et al., 1995). Continued human development

pressures resulted in regulation and permitting procedures for activities impacting streams and wetlands (Cole, 2006; Ainslie, 1994). The need to quantify ecological function or condition remains rooted in section 404 of the U.S. Federal Water Pollution Control Act of 1972 and the 'no net loss' policies implemented during the 1980s (USC., 2011; Carletti et al., 2004). The 2008 Mitigation Rule further outlined the requirement that aquatic resource functional losses be offset by mitigation (USACE and USEPA, 2008). Requirements of the Mitigation Rule include characterization of baseline information, consistent mitigation credit determination methodology, ecological performance standards, and monitoring; all of which necessitate ecosystem assessment.

A variety of ecosystem assessment strategies developed with the goal of improving natural resource management, and recent trends in evaluation focus on rapid assessment methods (RAMs) (Stein et al., 2009; 2009b; Wardrop et al., 2007; Figure 5.1). For example, Fennessy et al. (2007) and Berkowitz and Wilder (In Preparation) identify over 60 rapid assessment protocols and development of additional methodologies continues (Johnson et al., 2011; Wilder et al., 2012). The increased use of rapid assessments methodologies emphasizes the need for techniques that are sensitive to ecosystem impacts, are easily attained in a short period of time, and remain somewhat insensitive to seasonality (van Dam et al., 1998; Fennessy et al., 1998; 2004; Mack et al., 2000; Berkowitz et al., 2011). In most cases, the term 'rapid' refers to assessment methods that can be completed in one day or less (Smith, 1995; USEPA, 2004). While Sutula et al. (2006) and others make a distinction between methods addressing ecosystem condition and function, the technical standard outlined in the current work considers all assessment methodologies that can be accomplished within one day or less to be RAMs (Figure 5.1). Further, the technical standard is designed to be robust and does not favor any particular assessment methodology (e.g.,

Hydrogeomorphic Approach, California Rapid Assessment Method), or any specific ecosystem type (e.g., wetlands, streams).

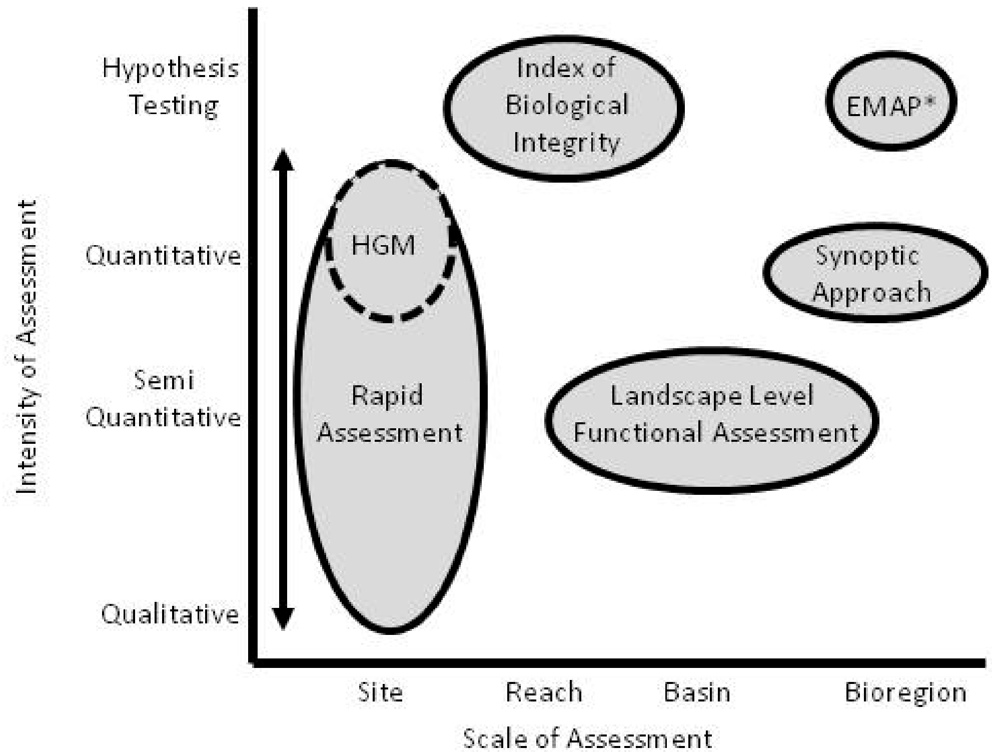

Figure 5.1. Example of ecological assessments methods with respect to the geographical scale and sampling intensity required. Modified from Sutula et al. (2006). *Environmental Monitoring and Assessment Protocol (EMAP).

The wide variety of assessment methods in use as well as modifications made to some existing methods has resulted in recent debate regarding the assessment of ecological condition, function, and associated mitigation requirements and costs. As a result, the technical standard for RAMs provides a template for developing and conducting assessment approaches and outlines strategies for improving existing methodologies.

5.3. Results and Discussion

Rapid ecosystem assessment methods that meet the technical standard incorporate and document the development and implementation of the following elements: 1) Interagency development team, 2) Clearly defined assessment goals, 3) Ecosystem classification, 4) Geographic extent, 5) Rapid application, 6) Calibration based on data, 7) Collection of quantitative, numerical data based upon written protocols, 8) Peer review, and 9) Applicable assessment outputs.

The following section discusses the major features of RAM approaches, each of which represent an essential component of the technical standard. The various RAM approaches currently in use exhibit a wide variety of characteristics. However, each assessment method can be described and categorized based on a few key components (Carletti et al., 2004; Sutula et al., 2006). The technical standard identifies nine key elements. Each component receives a description, a testable standard, and approaches for improving aspects of rapid assessment methods that fail to meet one or more aspects of the technical standard. Additionally we discuss issues related to the modification of existing methods and the need to maintain flexibility when applying rapid ecological assessments.

5.3.1. Interagency development team

RAM developers range from individual resource managers to large teams of ecological experts working together. Clairain (2002) recommends the formation of an interagency, interdisciplinary assessment team of ecological experts in the development of a RAM approach. The size and composition of the development team impacts assessment method outcomes, with more experienced and interdisciplinary teams capable of producing methods applicable to larger geographical scales across multiple ecosystem types (Sutula et al., 2006). Development team

members often include federal, state, and local resource agency staff, academics, and other subject matter experts.

RAM approaches should incorporate input from an interagency team of experts familiar with the ecosystem functions or conditions found within the geographical area of interest. Team members should include individuals with experience concerning rapid ecological assessment techniques, sampling methods, and exhibiting knowledge regarding ecosystem impacts occurring within the region. The names, occupation, and professional resume of all participants should be recorded and maintained for documentation purposes.

An interagency review of the rapid assessment method should be conducted for RAMs currently in use that were not developed utilizing input from an interagency development team as described above. The interagency review should include available information documenting the decision making procedure utilized in method development and focus on 1) the RAM protocol, 2) calibration and scaling of assessment parameters, and 3) the application of assessment method results. Interagency review comments should receive a written response. Changes to the assessment method should be incorporated where appropriate, especially when available data supports recommendations and the proposed changes enhance the validity, efficiency, and goals of the rapid assessment method.

5.3.2. Clearly defined assessment goals

RAM developers establish the goals of the assessment method. The goals of a RAM approach include 1) the purpose and 2) potential applications for the assessment methodology. 1) Rapid ecological assessment methods evaluate either ecosystem condition or function (Fennessy et al., 2004). Ecosystem condition is defined in the Mitigation Rule as the relative ability of an aquatic resource to support and maintain a community of organisms having a species

composition, diversity, and functional organization comparable to undisturbed aquatic resources in the region (Federal Register 2008). Ecosystem functions include the physical, chemical, and biological processes that occur in ecosystems (Smith et al., 1995). RAM approaches examining condition include the California Rapid Assessment Method (CRAM) which evaluates aquatic resource characteristics (e.g., plant or animal communities; CWMW, 2012). The Hydrogeomorphic (HGM) approach has proven to be rapid and focuses on determining individual ecosystem functions (e.g., removal of elements and compounds) or general classes of function (e.g., habitat) (Berkowitz et al., 2011; Brinson, 1996; Noble et al., 2010). Many assessment approaches examine a combination of conditional and functional measures (Berkowitz and Wilder, In Preparation).

2) An additional goal in RAMs includes identifying the intended uses and applications of the assessment approach. Commonly cited applications include quantifying expected adverse impacts of project alternatives, prioritizing wetlands for special protection, calculation of mitigation requirements, or estimation of anticipated conditional/functional lift associated with a restoration or enhancement actions (Mack, 2001; USACE New Orleans District, 2012). The combination of the RAM purpose and intended applications dictate the precision required to accomplish the goals of a given assessment method and the level of data collection, expertise, time, and expense necessary to achieve those goals (Sutula et al., 2006). Documentation supporting RAM development and implementation should clearly identify the purpose of an assessment approach and outline intended and potential applications.

Documentation should be developed outlining the intended purpose and potential applications addressed within the assessment method for rapid assessment methods currently in use that fail to clearly identify assessment method goals as described above. Documentation of

goals should include available information regarding the decision making procedure utilized in method development and focus on 1) whether the assessment method is intended to evaluate ecosystem condition or function and 2) suitable applications for the methodology (e.g., determination of project impacts, calculation of mitigation requirements). Additionally, if the purpose or intended applications of an existing assessment method are expanded or modified, documentation outlining the rationale and any data, end user feedback, or other information supporting the decision making process should be incorporated into RAM materials.

5.3.3. Ecosystem classification

The classification of ecosystems remains critical in any assessment approach (Reppert and Sigleo, 1979; Karr and Chu, 1997; 2006; Stoddard et al., 2006). Ecosystems display variability within the same natural regions (e.g. a forested floodplain vs. a tidal marsh in the Atlantic Coastal Plain) and classification reduces the effect of variability on assessment method output, thus increasing the ability to discern differences among individual sites (Stein et al., 2009). Ecosystem classification also increases the accuracy and repeatability of assessment methods by defining target ecosystems (Wharton, 1978; Smith et al., 1995), which enables the in-kind comparisons required by the Mitigation Rule (USACE and USEPA, 2008). In streams, common classification schemes include stream order, flow regime (e.g., perennial), and the approaches outlined by Dave Rosgen (Wildland Hydrology, Inc.) and others (Noble et al., 2010; Rosgen, 1996). Examples of classification systems applied to wetland and other aquatic resource include vegetative, hydrologic, and geomorphic characteristics (Cowardin et al., 1979; Brinson, 1993).

RAMs should incorporate ecosystem classification. The rationale for selecting a particular classification system should be documented along with a description of the target ecosystem characteristics (including potential adverse impacts) found within the region.

Documentation should be developed integrating a classification scheme and defined geographic extent into the assessment method for RAMs currently in use that fail to classify ecosystem types as described above. Documentation of classification should include available information documenting the decision making procedure utilized in method development and focus on 1) the ecosystem types examined, 2) the location of target ecosystems within the landscape, and 3) what physical drivers (e.g., hydrologic regime, salinity) and 4) structural and biological community factors (e.g., channel morphology, vegetative growth form) distinguish target ecosystems. In some cases, modifications of existing assessment approaches have expanded the ecosystem classification types initially addressed with an assessment method. The expansion of existing ecosystem classes should be accompanied by data collection supporting the expansion process or resulting in modifications to existing methodologies prior to implementation.

5.3.4. Geographic extent

Assessment approaches differ in geographic extent. The geographic extent of RAM applications are defined using ecological or geophysical regions (e.g., watershed, ecoregion, Land Resource Region) and approaches restricted to a geopolitical area (e.g., state, county) (Omernik, 1987; USDA, 2006; Smith et al., 2013). Limiting the geographic region addressed by a RAM improves accuracy and efficiency by accounting for regional differences in climate, geology, soils, hydrology, plant and animal communities, and other factors effecting ecological conditions and functions (National Research Council, 1995; Wakeley, 2002).

RAMs should define the intended geographical extent of the methodology. The rationale for selecting a particular geographical area should be documented along with a description of the target ecosystem(s) occurring within the region of interest (including potential adverse impacts).

Documentation should be developed defining the intended geographic area for assessment method application for RAMs currently in use that fail to define a geographic region as described above. Documentation of geographical extent should define the applicable assessment region using political, ecological, or geophysical boundaries and include a rationale supporting the selected boundaries. In some cases, modifications of existing assessment approaches have expanded the assessment method beyond the geographical region or ecosystem type originally intended. The expansion of existing geographical ranges should be accompanied by data collection capable of supporting the expansion process or resulting in modifications to existing methodologies prior to implementation in the expanded region.

5.3.5. Rapid application

RAM protocols impact efficiency through time requirements and the need for onsite visits. Approaches necessitating a large number of field measurements or time consuming analysis are beyond the scope of a RAM. In order to provide usability and remain practicable, assessment methods are designed to be rapid (Stein et al., 2009; 2009b; Wardrop et al., 2007). Need for rapid assessments methodologies is well documented and has been defined as one-half day or less of field data collection (van Dam et al., 1998; Mack et al., 2000) and a half day or less of office preparation and analysis (Smith et al., 1995; Berkowitz et al., 2011). Fennessy et al. (2007) and Berkowitz and Wilder (In Preparation) evaluate over 60 methodologies, reporting that rapid functional and conditional approaches require 1 day or less to complete. Most RAMs

provide a written protocol outlining data collection and analysis procedures which promote consistent, rapid application.

RAMs should remain rapid requiring an average of one day or less to complete based upon the application of a clear, written protocol. Documentation demonstrating time requirements should be compiled during method development, field testing, and implementation. Under some circumstances (e.g., difficult terrain, highly disturbed areas) more time may be required to complete the assessment method.

Documentation should demonstrate the time required to apply the assessment method as written, including the time required to complete each component of the assessment protocol for RAMs currently in use that do not meet the definition of rapid defined above. Method developers should examine the components of the assessment protocol and determine if redundancy exists within the assessment procedure or if sampling techniques can be altered to increase efficiency. Smith et al. (2013) provides guidance on identifying and removing redundant and unnecessary parameters from assessment methods. The removal or alteration of assessment protocols should be documented including rationale for any changes and, when possible, data demonstrating the effect of changes on assessment outcomes. In some cases, a RAM approach may not be possible or appropriate and alternative, more intensive methods should be applied.

5.3.6. Calibration based on data

RAM calibration (i.e., standard of comparison) scales observations and measurements to the level of ecosystem condition or function. Common calibration approaches include the comparison of assessment results with scores generated in areas representing undisturbed, natural conditions (i.e., reference or reference standard) and along a gradient of ecosystem disturbances (Hughes et al., 1986; Brinson, 1993; Smith et al., 1995). Wigham (1999)

supports the use of calibration based upon collected quantitative data for promoting efficiency, consistency, and establishing a framework for comparison between impacted, unimpacted, and potential mitigation areas. Other approaches employ calibrations derived from available literature or from the opinions of resource professionals (i.e., best professional judgment [BPJ]; USACE, 1999). Many assessment methods employ a calibration scheme combining BPJ, available literature, and collected data (Berkowitz and Wilder, In Preparation).

Rapid ecological assessment calibration can occur at two levels. First, assessment approaches can be calibrated to individual metrics, parameters or variables. For example, the level of ecological function or condition can be calibrated to the size or abundance of canopy trees [Figure 5.2(A)]. Second, the level of ecological function or condition can be compared to the final assessment method outputs [i.e., scores; Figure 5.2(B)]. Smith et al. (2013) provides comprehensive guidance on approaches to RAM calibration.

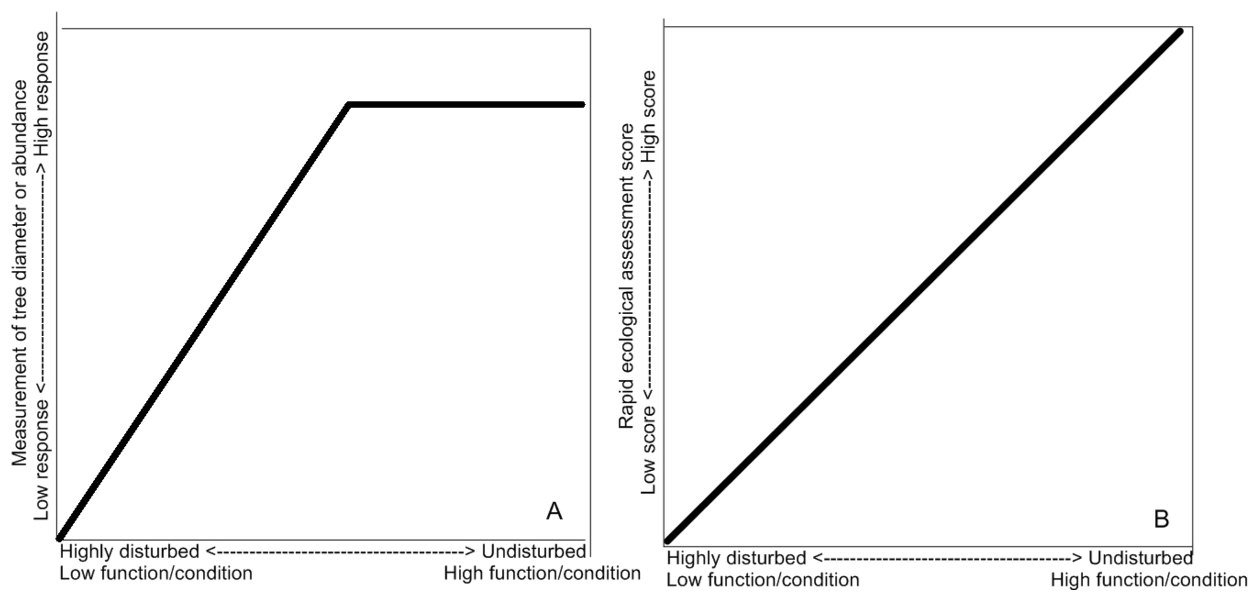

Figure 5.2. A) Example of RAM calibration examining the relationship between the level of ecosystem function/condition and a measure tree diameter or abundance, B) Example of RAM calibration examining the relationship between the level of ecosystem function/condition and RAM score.

RAMs should be calibrated based upon collected and/or published data. Documentation should support the selection and implementation of the calibration process.

Calibration should be conducted for RAMs currently in use that fail to calibrate assessment method scoring based on collected or published data. The decision making procedure utilized in assessment method calibration should be documented along with all data supporting the use of the existing assessment method and any modifications implemented based on the calibration procedure. Calibration data can be used to verify assessment results in instances where best professional judgment forms the basis of the assessment method scoring. In many cases, a small amount of calibration data can verify existing assessment methods and provide a defensible basis for modifying methods currently in use. Figure 5.3 displays the relationship between a hypothetical variable and a gradient of site condition/function. Results in Figure 5.3(A) support the use of the hypothetical variable within the RAM, while the scatter observed in Figure 5.3(B) suggests that modification to the RAM protocol, calibration, or variable selection is required.

5.3.7. Collection of quantitative, numerical data

Both quantitative and qualitative data remain common within RAMs. Quantitative data includes repeatable measurements and estimates, generating numerical values (Berkowitz et al., 2011). For example, wetland RAM protocols often require quantification of the amount of ground cover, the shrub density, or the average tree diameter within a defined area (Smith and Klimas, 2002). Noble et al. (2010) describes methods for determining stream ecosystem components including stream particle embeddedness based on measurements of the percentage of the particle surface covered by fine sediments.

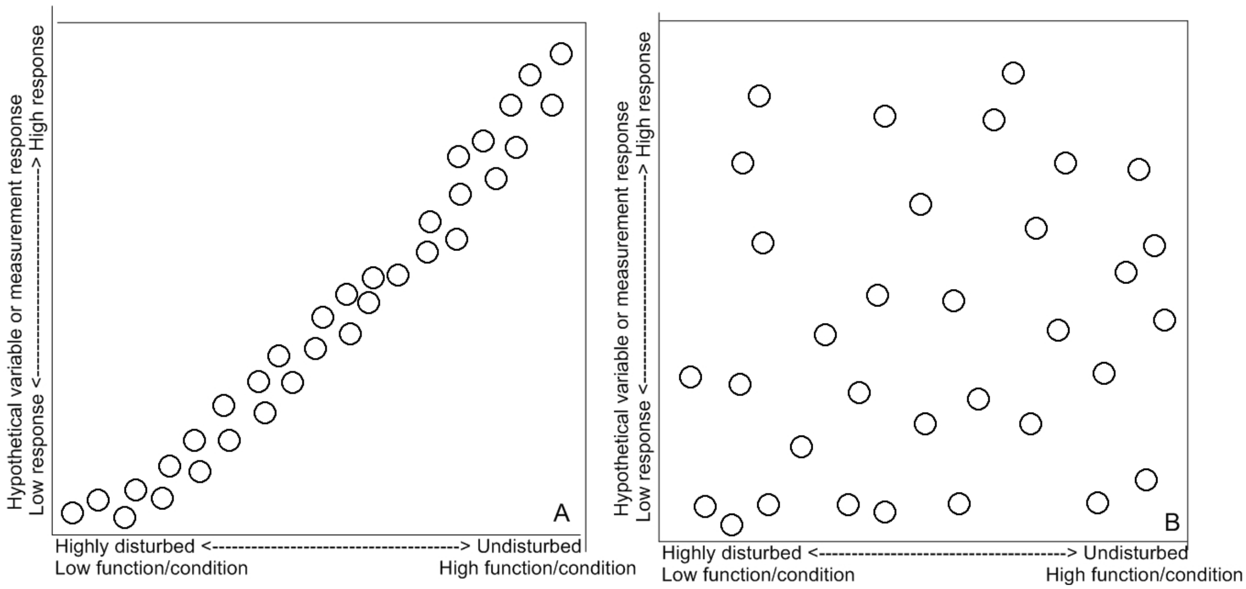

Figure 5.3 A) Example of RAM variable that responding appropriately to changes in site function/condition. Note that the figure displays a clear relationship between the hypothetical variables and site condition; a positive linear example is shown here, but asymptotic, negative linear, or other configurations are possible. B) Example of RAM variable that does not respond predictably to changes in site function/condition. Modified from Smith et al. (2013).

The embeddedness measurement examines 30 individual particles, then places observed values into categories or groups (Table 5.1). This approach results in the generation of quantitative, numerical data, and maintains both repeatability and efficiency by grouping elements into categories. The technique of grouping or categorizing quantitative data remains widely applied (Daubenmire, 1959; Floyd and Anderson, 1987) and Fennessy et al. (2004) suggests that categorizing assessment components dampens variability resulting in a more robust method. Other common quantitative measures applied in stream assessments include measurements of width:depth ratios and measurements of substrate particle size (Rosgen, 1996).

Qualitative variables commonly employ narrative statements describing an ecosystem impact or condition based on a visual estimate of ecosystem structure or the presence/absence of ecological stressors. For example, Table 5.1 provides the descriptive categories utilized in the Universal Mitigation Assessment Methodology (UMAM, 2012). Although qualitative approaches

potentially increase rapidity, they lack sensitivity and result in increased subjectivity (Klimas, 2008). Many assessment methods contain a combination of quantitative and qualitative elements, including the presence/absence approach to environmental stressors (Table 5.3; Berkowitz and Wilder, In Preparation).

Table 5.1. Example of a quantitative variable for embeddedness (Noble et al., 2010).

Rating	Rating Description
5	<5 percent of surface covered, surrounded, or buried by fine sediment
4	5 to 25 percent of surface covered, surrounded, or buried by fine sediment
3	26 to 50 percent of surface covered, surrounded, or buried by fine sediment
2	51 to 75 percent of surface covered, surrounded, or buried by fine sediment
1	>75 percent of surface covered, surrounded, or buried by fine sediment

Table 5.2. Example of qualitative narrative statements applied in determining Vegetation and Structural Habitat scoring (UMAM, 2012).

•A score of (10) means that the vegetation community and physical structure provide conditions which support an optimal level of function to benefit fish and wildlife utilizing the assessment area as listed in Part I.
•A score of (7) means that the level of function provided by plant community and physical structure is limited to 70% of the optimal level.
•A score of (4) means that the level of function provided by the plant community and physical structure is limited to 40% of the optimal level.
•A score of (0) means that the vegetation communities and structural habitat do not provide functions to benefit fish and wildlife.

The type of data collected dictates the output generated by each RAM. Numerical outputs include scoring mechanisms in which outputs appear across a continuous or categorical range. For example, many assessment approaches vary scoring between 0.0 and 1.0, while others score between 0.0 and 100 (Smith and Klimas, 2002; Sutula et al., 2006). Narrative, qualitative outputs often describe an ecosystem functional or conditional status as “Good”, “Fair”, or “Poor” or as “High”, “Medium”, or “Low.” Klimas (2008) suggests that utilizing broad narrative and qualitative categories increases user bias, potentially decreasing accuracy.

The objectivity of an assessment impacts RAM outcomes, with more objective measures associated with decreased variability. For example, measuring the diameter of each tree within a defined area yields accurate and quantitative, repeatable results as shown in Berkowitz et al. (2011). Qualitative data utilized in assessment methods generally relies on narrative statements (e.g., Table 5.2) or the perceived presence of absence of ecological stressors (Table 5.3; Wardrop et al., 2007). A direct measurement of tree diameter is objective, while the categories presented in Tables 5.2 and 5.3 remain subjective with responses depending on experience, training, and perception. For example, the description of Nutrient Enrichment or Eutrophication (Table 5.3) includes the statement “Heavy or moderately heavy cover of algal mats.” This qualitative description is problematic because what one user considers “heavy or moderately heavy” may be interpreted as normal and appropriate to another user. As a result, Klimas (2008) notes that although qualitative approaches are designed to be rapid and repeatable, they lack sensitivity and potentially decrease accuracy and repeatability.

Quantitative and qualitative data can be collected generated onsite or generated using office based resources (e.g., GIS, aerial photos). However, most RAMs require an onsite field visit (Fennessy et al., 2007). The need for a site visit adds to the amount of time required to complete the assessment, but allows for evaluation of both quantitative, numerical measurements (e.g., tree diameter) and increased precision of the assessment method (Sutula et al., 2006).

RAMs should utilize quantitative, numerical data and incorporate a site visit when possible. Both onsite and offsite measurements can be organized into groups or categories to promote repeatability and efficiency. Field testing conducted during RAM development ensures that method outputs (i.e., scores) respond as intended to various levels of alteration or stress.

Documentation should be developed integrating additional measures into the assessment method for rapid assessment methods currently in use that do not collect quantitative, numerical data. Data collection should include measurements of ecosystem structure (e.g., channel morphology), vegetation community (e.g., composition, abundance, invasive species), site hydrology (e.g., flood frequency) and/or other rapid measures with established relationships to ecological condition or function. The decision making procedure utilized to select quantitative measures should be documented along with data supporting the use of the existing assessment method and any modifications made based on the data collected. Similarly, assessment methods that currently do not require a site visit can be improved by incorporating onsite measures into the existing methodology. However, remote locations with difficult access may prevent the collection of onsite data. In instances where narrative statements or the presence/absence of ecosystem stressors form the basis of the assessment method, quantitative data can be used to verify assessment results. In many cases, a small amount of data gathered across study areas exhibiting a gradient of alteration can verify existing assessment methods or provide a defensible basis for the modification of existing methods (Smith et al., 2013).

5.3.8. Peer review

Many RAMs incorporate independent peer review into the development process, while other methods do not undergo review (Berkowitz and Wilder, In Preparation). Clairain (2002) describes a detailed procedure for distributing assessment methods to independent peer reviewers and incorporating reviewer recommendations into the assessment approach. The peer review process promotes technical validity, defensibility, and encourages stakeholder buy-in through the incorporation of comments and recommendations from experts within Federal, State, Tribal, and local agencies, academia, and the private sector (Federal Register, 1997).

Table 5.3. Categories of stressors and stressor indicators used in a rapid assessment. Adapted from Wardrop et al. (2007).

HYDROLOGIC MODIFICATION
Ditching
Tile drains
Weirs or dams
STORM WATER INPUTS OR CULVERTS
Non-storm water point source
Filling, grading, dredging
Roadbed, railroad
Dead or dying trees

SEDIMENTATION
Active or recent adjacent construction, plowing, heavy grazing or forest harvesting
Dominance (> 50% cover) of sediment tolerant vegetation
Sediment deposits or plumes
Eroding banks or slopes
Silt on ground or vegetation
URBAN OR ROAD STORM WATER INPUT OR CULVERTING

HIGH BIOLOGICAL OXYGEN DEMAND
Excessive density of aquatic plants or algal mats in water column
Excessive deposition or dumping of organic waste (e.g., leaves, grass clippings)
DIRECT DISCHARGES OF ORGANIC WASTEWATER OR MATERIAL (E.G., DAIRY OR FOOD PROCESSING WASTE)

TOXICITY DUE TO CONTAMINANTS
Severe vegetation stress
Obvious spills, discharges, plumes, odors
Wildlife impacts (e.g., tumors, abnormalities)
ADJACENT INDUSTRIAL SITES, PROXIMITY OF RAILROAD

SALINITY
Obvious increase in concentration of dissolved salts

VEGETATION ALTERATION
Dominance (> 50% cover) of exotic or invasive plant species
Mowing
Grazing
Tree cutting (> 50% canopy removed)
Brush cutting (mechanized removal of shrubs and saplings)
Removal of woody debris
Aquatic weed control (mechanical or herbicide)
Excessive herbivory
Evidence of chemical defoliation

NUTRIENT ENRICHMENT OR EUTROPHICATION
Dominance (> 50% cover) of nutrient tolerant vegetation
DIRECT DISCHARGES FROM AGRICULTURAL FEEDLOT, MANURE PITS, ETC.
DIRECT DISCHARGES FROM SEPTIC OR SEWAGE TREATMENT SYSTEMS
Heavy or moderately heavy cover of algal mats
Methane odor
Dead fish

ACIDIFICATION
ACID MINE DRAINAGE DISCHARGES
ADJACENT MINED LANDS OR SPOIL PILES
Excessively clear water
Absence of expected biota

TURBIDITY
High concentration of suspended solids in water column
Moderate concentration of suspended solids in water column

THERMAL ALTERATION
Significant increase in water temperature
Moderate increase in water temp

RAM approaches should incorporate an independent peer review. A minimum of two peer reviewers should exhibit experience with rapid ecological assessment techniques, sampling methods, and exhibiting knowledge regarding the ecosystem types addressed by the RAM. Assessment method developers should provide written responses to peer review comments and document any associated modifications to the assessment approach.

Independent peer review should be conducted For RAMs currently in use that did not undergo peer review as described above. The review should include available information documenting the decision making procedure utilized in method development and focus on 1) the

assessment protocol, 2) calibration and scaling of assessment parameters, and 3) the application of assessment method results. Peer review comments should receive written a response. Changes to the assessment method should be incorporated where appropriate, especially when available data supports recommendations and the proposed changes enhance the validity, efficiency, and goals of the rapid assessment method.

5.3.9. Applicable assessment outcomes

The final outcome or product generated as part of the RAM should remain clear, usable, and coincide with the stated goals of the assessment application. Wakeley et al. (2001) demonstrates that in many instances, the calculation of existing or baseline functional or conditional values may be all the information that is needed. In this instance a simple, numerical functional or conditional score meets the assessment objectives. However, resource managers often require determination of gain or loss of function/condition within an ecosystem, including alternatives analysis designed to avoid, minimize, and mitigate for negative ecosystem impacts (USACE and USEPA, 1990). For example, an assessment approach designed for determining mitigation requirements should generate numerical assessment scores that facilitate calculation of required mitigation credits including the aerial extent of land and the type of ecosystem involved.

Additionally, in cases where multiple ecosystem functions or conditions are evaluated, some methods combine individual functional or conditional scores into a single value; often using simple averages. This approach promotes efficiency and simplifies the comparison between ecosystems. However, combining scores limits the ability to target individual conditional or functional elements, increases uncertainty, and makes method validation (i.e.,

determining method accuracy by applying independent measures) more difficult (Wakeley and Smith, 2001; Fennessy et al., 2004).

The outcomes generated by the RAM should provide a clear and useful measure of the condition or function of the target ecosystem that achieves the stated assessment goals. Assessment method documents should demonstrate how all results are determined, including the mathematical formulas and/or multimetric relationships utilized in scoring conditions and functions (including examples of spreadsheet calculations).

The reporting of results should be modified in order to address the goals established by the assessment method for RAMs currently in use that lack practicable assessment outcomes as described above. Increasing the usability of assessment method results could include developing a mechanism for incorporating a spatial dimension. For example, Smith and Klimas (2002) provide a rapid assessment method that scores wetland functions on a scale of 0.0 to 1.0. The score is then adapted in order to incorporate a dimensional component by multiplying the function score by the size of the project area.

5.3.10. Rapid assessment method expansion and modification

Recent debate has resulted from the modification of RAMs and the application of modified approaches to expanded geographic regions or ecosystem types. The debate stems from the application of new requirements for permit applicants, potential changes in mitigation requirements, or a lack of familiarity with rapid assessment protocols and methodologies. In some cases, a decreased level of documentation is utilized when assessment modification occurs compared to initial RAM development. For example, initial RAM development may include an interagency, interdisciplinary team of ecological experts, field data collection, method protocol testing, peer review, and other steps documenting the decision making process. Similar

approaches are not always clearly applied to modified RAMs or the decision making process is not well documented. However, opportunities exist for modification and geographical expansion of current RAMs. For example, the assessment approach developed by Noble et al. (2010) for headwater streams in western WV and Eastern KY is undergoing expansion into surrounding areas. The expansion process is based on collection of data, testing of field data collection protocols, and well documented adjustment of the assessment method based on data where appropriate.

The expansion of RAMs into larger geographic regions or additional ecosystem types and modifications to existing methodologies based on new information or user feedback should be documented. Important aspects include: 1) the decision making process regarding modification or expansion, 2) the collection of data, 3) field testing of the modified or expanded method, 4) the potential need for peer review of expanded or modified methods, and 5) the expected outcomes resulting from expansion of modification of the RAM. Assessment method developers should consider how modifications will affect the baseline conditions, calibration, potential adverse impacts, and mitigation requirements.

5.3.11. Flexibility

Natural environments can exhibit a high degree of variability within a given geographic area and ecosystem class. Often higher levels of heterogeneity are observed in altered and disturbed ecosystems. As a result, approaches to rapid ecosystem assessment must allow for flexibility in order to remain robust and practical. The technical standard and accompanying recommendations outlined in this document provide a template for the development of RAMs and approaches for improving existing methodologies. However, the standard may not be applicable or appropriate in all cases. In instances where RAM approaches deviate from the

standards outlined above, documentation should be developed outlining the decision-making process involved and providing data supporting selected methodologies and the application of those methodologies.

5.4. Conclusion

A technical standard for the development and application of rapid ecosystem assessment methods is discussed. Nine components of the technical standard receive a description, a testable standard, and approaches for improving existing RAMs. Technical standard components include: 1) Interagency development team, 2) Clearly defined assessment goals, 3) Ecosystem classification, 4) Geographic extent, 5) Rapid application, 6) Calibration based on data, 7) Collection of quantitative, numerical data, 8) Peer review, and 9) Applicable assessment outputs. The standard promotes technically valid, science-based, defensible, and transparent approaches to rapid ecosystem assessment. In addition to the technical standard components outlined above, special considerations are required when modifying existing approaches to new geographical areas or ecosystem classes. Further, assessment method developers and end users must maintain a level of flexibility in order to account for the variability and diversity associated with both natural and altered ecosystems.

CHAPTER 6: CONCLUSIONS

Widespread human alterations have decreased the wetland acreage within the contiguous United States by 50%, resulting in deceases in wetland functions and associated benefits. Additionally, stream ecological function has been reduced, with over half of the nations' waterways exhibiting degraded or poor quality. As a result, federal policy requires the management of aquatic and wetland ecosystems including policies emphasizing avoidance, minimization, restoration, and mitigation of negative impacts. These policies necessitate the assessment of ecological condition and function. Presently, the US Army Corps of Engineers conducts over 70,000 wetland and stream regulatory evaluations per year. An estimated 92% of regulatory determinations are completed within 60 days, demonstrating the need for efficient, science-based rapid ecological assessment methods for wetlands and streams.

The current work examined the wide variety of rapid ecological assessment methods in use by resource professionals (>60), and investigated the applicability of several assessment methods addressing biogeochemical functions in both bottomland hardwood wetlands and headwater streams. Further work established restoration trajectories and milestones within project relevant timescales by identifying metrics that mirror ecological performance. Finally, we drafted a technical standard for the development and application of rapid ecological assessments, incorporating nine testable components designed to ensure technical validity and repeatability within ecological assessment approaches.

This research is one of the first to apply direct measures of hydrology and biogeochemical functions to rapid assessment results. The dissertation examines the current status of rapid ecological assessments, develops novel tools based on rapid assessment protocols,

and forwards the application of ecological assessments through the development of a comprehensive technical standard.

Research demonstrated the validity of a rapid assessment approach designed for bottomland hardwood forested wetlands in the Lower Mississippi Valley; an area that has undergone extensive restoration activities within the past three decades. The rapid assessment approach proved valid in wetlands exhibiting a range of reforestation ages across three biogeochemical functions (nutrient cycling, export of organic C, water quality improvements). Measures and proxies of biogeochemical cycling significantly correlated with rapid assessment outcomes, providing evidence that the approach supplied valid, reliable outcomes while satisfying the requirements of a rapid methodology.

The rapid assessment approach applied in the study utilized elements of 1) classification, 2) quantitative data collection, and 3) scaling based on reference data. Results indicate that assessment approaches incorporating these key elements provide a valuable tool for resource managers, the scientific community, and the public. However, the wide variety of rapid assessments in use increases the necessity for evaluation and validation of methods with measures of ecosystem function or services. Further, efforts to validate rapid assessment procedures requires coupling nutrient and microbial concentration measures with transfer/processing mechanisms in order to adequately address wetland functionality, especially with respect to biogeochemical functions. The study demonstrated that measurements of restored wetland functionality increased with forest succession across the restoration chronosequence examined, but did not achieve the level of functionality observed at control sites. As the growth and evolution of rapid assessment approaches continues, the need for an accepted technical criteria addressing rapid assessment development and validation procedures continues to expand.

The rapid assessment proved an effective method for determining the extent of biogeochemical functions occurring across a variety of study site ages and conditions. As a result, further research sought the construction of a management tool for developing restoration trajectories and milestones. Determining the performance of restoration projects remains problematic due to the time necessary for forested wetlands to reach maturity, limited monitoring requirements, and a lack of coherent performance standards. Identifying measurable rapid assessment variables enables resource managers to establish early restoration milestones that examine the likely trajectory of a reforested area within project relevant timescales. Four rapid assessment variables showed strong correlations within recently reforested agricultural areas. Soil O and A horizon increased throughout the restoration chronosequence, providing direct relationships with forest age. Shrub-sapling density and ground vegetation cover increased in young restoration sites, followed by decreasing variable output with the onset of canopy closure, thus providing performance standards in both early and intermediate age forests.

Assessment variables showing a rapid response following reforestation define early restoration trajectories and performance, allowing for corrective action within project relevant monitoring periods. For example, landscape management could be conducted to promote growth of desirable plant species, or invasive plant species could be removed from reforested areas. Assessment variables classify into three categories: 1) rapid response variables with a high potential to change in the first years following reforestation, 2) response variables requiring additional time (e.g., >15 years) to display a measureable effect, and 3) stable variables that remain fixed over time. Establishing three variable categories helps determine ecosystem structural and functional responses to reforestation. In situations where project goals include determining ecosystem conditional/functional change over time, resource managers should focus

on variables that respond within the timescale of interest, and additional emphasis must be placed on the subset of assessment variables that display rapid response and address both practical and ecological concerns.

Since rapid assessment approaches are applied to a number of diverse ecosystems across wide geographic regions, additional research examined biogeochemical proxy measures in headwater streams within Appalachia, USA. Rapid ecological assessments have previously not been applied to headwater streams where ephemeral hydrologic conditions limit the use of traditional stream metrics including benthic macroinvertebrate sampling and water quality analysis. Study results demonstrate that headwater streams impacted by surface mining, agriculture, and contemporary logging activities receive and process fewer nutrients while transporting higher nutrient loads; indicating that biogeochemical cycling decreases in altered locations. The rapid assessment method applied provided a useful tool for estimating the level of biogeochemical cycling. Assessment scores were driven by a combination of alteration type and recovery period. However, challenges remain in determining the long term impact of recent landscape alterations on biogeochemical cycling, including novel impacts from surface mining utilizing mountaintop removal and valley fill. An economic analysis highlighted the utility and cost effectiveness of rapid assessment approaches, especially in cases where biogeochemical proxy data supports rapid assessment results.

The dissertation examined over 60 different rapid ecological assessment approaches currently in use throughout the United States and compared the core elements of assessment development, application, geographical extent, and modification. The research described above demonstrates that rapid assessment utilizing classification, quantitative data collection, and calibration (i.e., scaling) based on reference data produced validated results, provided valuable

tools for determining ecological function and performance. However, the lack of a coherent standard for wetland and stream assessment method development and application has resulted in a potential lack of technical validity, transparency, and defensibility. As a result, we designed a technical standard for rapid ecosystem assessment methods used to evaluate ecosystems including wetlands and streams for the purposes of determining mitigation requirements, estimating impacts resulting from project implementation, quantifying the success of restoration efforts, and other applications. The standard outlines the principle components required to promote technically valid, science-based, defensible, and transparent approaches to rapid ecosystem assessment. The technical standard consists of nine components including: 1) Interagency development team, 2) Clearly defined assessment goals, 3) Ecosystem classification, 4) Geographic extent, 5) Rapid application, 6) Calibration based on data, 7) Collection of quantitative, numerical data, 8) Peer review, and 9) Applicable assessment outputs. Each component represents a testable standard capable of guiding the development of new rapid assessment protocols and evaluating existing approaches. Guidance for improving existing assessment methods that currently fail to meet the technical standard are also included.

Research suggests that rapid assessment approaches exhibit a wide range of development strategies, application, and other method components. As a result, the dissertation supports assessment approaches that integrate the components outlined in the technical standard. Further research is needed to determine the efficiency and validity of many other methods in use. Also, while the current work establishes a framework for determining early restoration performance trajectories, more research is needed to determine mid- and long-term restoration milestones, especially concerning novel activities including mountain top removal mining and valley fill. The incorporation of science-based approaches, including biogeochemical data, into the

development and application of rapid ecosystem assessment increases confidence in method validity; improving natural resource management in wetland and stream environments.

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VITA

Jacob Berkowitz was born in Columbia, South Carolina in 1979. He earned a Bachelor of Science degree in Environmental Studies from the University of North Carolina at Asheville in 2003, followed by a Masters of Science in Soil and Water Sciences from the University of California at Riverside in 2005. Jacob then accepted a full time research position at the Pacific Northwest Forestry Sciences Laboratory located in Juneau, Alaska. Research focused on the impact of marine derived nutrients on landscape evolution, carbon export from temperate rainforest wetlands, and general studies in forest soils, nutrient cycling and wetland biogeochemistry. In 2008, he began work as a Research Soil Scientist with the US Army Corps of Engineers Engineer Research and Development Center in Vicksburg, Mississippi. Research interests shifted toward regional and national scale wetland initiatives including the development and implementation of novel approaches to wetland delineation and assessment, technical training for natural resource managers, wetland restoration, and participation in regulatory policy and enforcement activities. In 2011, the Army Corps of Engineers selected Jacob for participation in the Long Term Training Program at Louisiana State University in pursuit of a doctorate in Oceanography and Coastal Sciences specializing in wetland biogeochemistry and ecosystem assessment.