Long-term Trends in Mitigation and Wetland Restoration: Ecological Condition and Soil Organic Carbon

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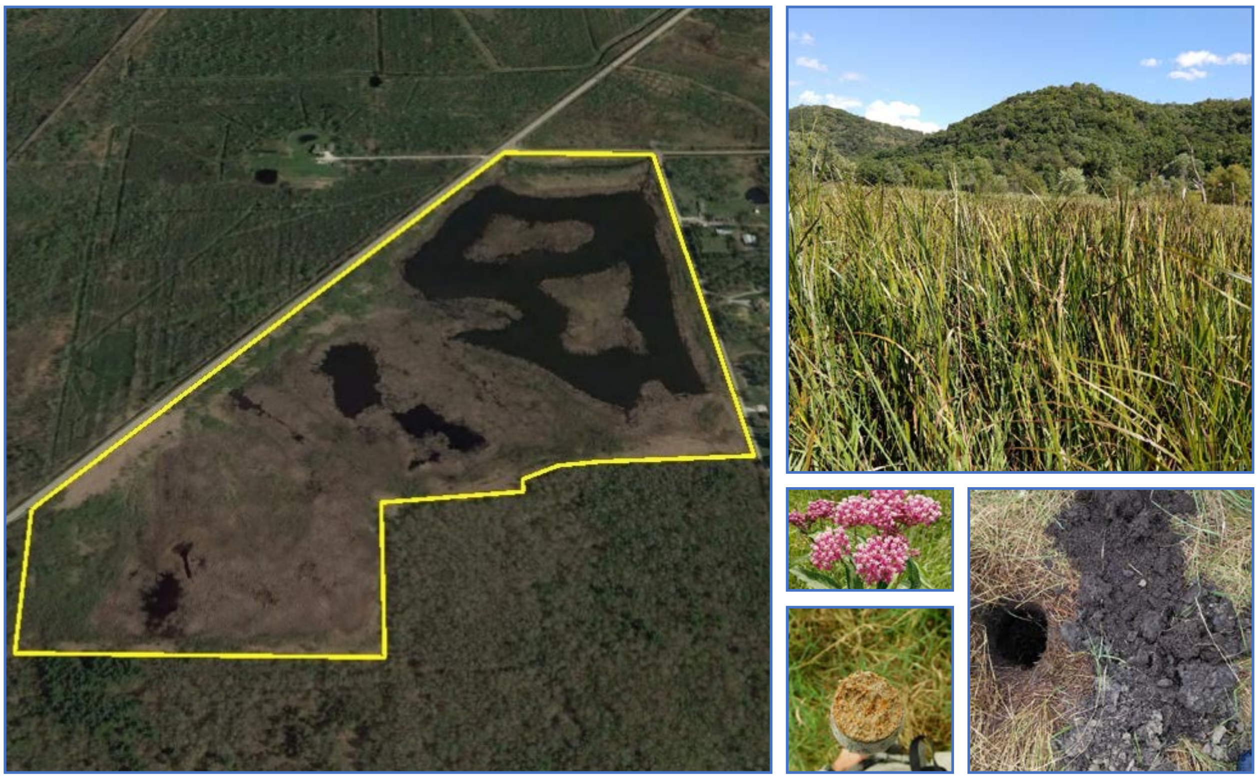

Clockwise from left, a 26 -year-old mitigation site in Oconto Co.; a rehabilitated shallow marsh dominated by *Sparganium eurycarpum* in the Driftless Area; a soil profile from a wet-mesic prairie restoration; hydric soil with low organic carbon from an excavation in Barron Co.; Swamp milkweed (*Asclepias incarnata*) from a 17-year old restoration.

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2 Table of Contents

3 Executive Summary

Compensatory mitigation for permitted wetland losses in Wisconsin has been in practice for almost 30 years now allowing a long-term assessment of ecological outcomes to improve the state's mitigation program and other entities restoring wetlands. In Part I of this study we examine ecological outcomes of wetland mitigation sites a minimum of 5 years post-monitoring period. In Part II we combine data from mitigation sites with previously collected data from 35 additional wetland restoration projects to measure the influence of soil attributes, ecoregion, hydrology, initial wetland drainage status, and hydrologic restoration technique on floristic quality (wC) and condition outcomes.

A total of 15 mitigation sites were randomly drawn from a list of 121 Corps-approved mitigation projects in Wisconsin. Selected mitigation sites ranged in age from 13 to 26 years old and varied in size from 4 to 465 acres. A floristic quality assessment survey and soil samples were collected from each unique restored wetland community encountered during site visits in 2018-19. Restored wetland plant condition 13 to 26 years post-restoration was variable with a small proportion of wetland assessment areas (3%) achieving EXCELLENT condition, 42% achieving GOOD or FAIR condition, and 56% resulting in POOR or VERY POOR condition. Restorations in the Southeastern WI Till Plains had significantly less success than those in the Northern Lakes and Forests ecoregion. Hybrid cat-tail (*Typha X glauca*) and reed canary grass (*Phalaris arundinacea*) were the most frequent species associated with POOR and VERY POOR condition results and together covered approximately 47.3% (449 acres) of compensatory wetland across all sites. Mitigation sites were, with few exceptions, not receiving long-term maintenance of plant communities.

Soil carbon levels were significantly lower in restored wetlands (range = 0.4 – 17.8% dry wt.; median = 3.8%) compared to three natural wetlands datasets (range = 0.2 -58.7%; medians = 19.0%, 22.9%, 42.8%). We found no evidence of a trend of increasing floristic quality or soil carbon levels with age; instead, older sites had slightly lower condition and soil carbon than younger sites. Overall, wetland types that were rarest on compensation sites include wetlands in GOOD or EXCELLENT condition, forested wetland communities of all conditions, and wetlands with organic soils. These results suggest a recovery debt may still exist for regulated impacts to wetlands of these types.

Across 163 wetlands restored by mitigation programs, non-profit groups, and wildlife habitat programs, the strongest factors associated with successful (FAIR or better) plant communities were: 1) summer saturation level at the surface (but not ponded); 2) soils with $\geq$ 10% organic carbon 3) location in the Northern Lakes and Forests ecoregion; 4) restored wet to wet-mesic prairies (not marshes); and 5) soil texture was mucky-mineral.

To avoid future functional loss in mitigation practice, and to continue to improve wetland restoration outcomes, we recommend: 1) Improving compliance with no-net-loss policy by employing wetland functional assessments for impacted and restored wetlands, 2) Developing strategies to avoid the establishment and spread of *Typha X glauca* and *Phalaris arundinacea* at all project stages; 3) Taking advantage of valuable remnant wetland functioning in soils and in watersheds when planning restorations, and 4) Employing long-term adaptive management strategies for restoration sites to improve low-functioning areas, and maintain or improve the underlying hydrology that sustains the system.

4 Introduction

4.1 Wetland Mitigation in Wisconsin

Wetland compensatory mitigation is the restoration, enhancement, creation, or preservation of wetlands to compensate for adverse impacts to other wetlands (NRC 2001). Mitigation was first mentioned in federal law in the 1980 version of the Clean Water Act Section 404(b)(1), but written guidance was not put in place until a 1990 Memorandum of Agreement between EPA and the US Army Corps of Engineers (Corps). This memorandum (USACE/EPA 1990) established that the overriding policy of wetland compensatory mitigation is to "strive to achieve a goal of no overall net loss of values and functions".

Wisconsin's compensatory mitigation program may be thought of as beginning in 1990 with the first 36 acres of credit listed jointly for Wisconsin and Minnesota in 1990 (RIBITS 2019). Since then, Wisconsin Department of Transportation (WisDOT) alone has developed 7,175 acres of wetland compensation (WisDOT 2018). Other sources of mitigation projects in Wisconsin include private mitigation banks and permittee-responsible mitigation administered jointly by the U.S. Army Corps of Engineers and the Wisconsin Department of Natural Resource (WDNR's) mitigation programs and mitigation for utility projects regulated by WDNR's Office of Energy. Estimates of total credits developed from these agencies were not obtainable, however, total acreage is likely to be considerably less than WisDOT's contribution.

4.2 Evaluating the Effectiveness of Mitigation for Authorized Impacts

Successful compensation should result in "one for one functional replacement (i.e. no net loss of values)" as stated in the 1990 Memorandum of Agreement. Evaluating success would therefore require measuring lost wetland functions and values and comparing them with those developed for compensation. Wetland function is defined by the Corps as "the physical, chemical, and biological processes that occur in ecosystems"; and values as the "utility or satisfaction that humans derive from aquatic resource services". Both are difficult to measure, let alone to measure in the same way for both the impacted and the replacement wetlands.

Because of the difficulty in measuring wetland functions and values, guidance documents for compensatory mitigation use wetland acreage and wetland type as proxies. Wetland "type" or "kind" is defined by a wetland's "structure and/or function" (RIBITS 2020) and there is a stated preference for compensation wetlands to be "in-kind" or "structurally and/or functionally similar to the impacted wetland in type" (USACE/EPA 1990). Credit ledgers from the Corps indicate wetland type is measured in terms of structure (i.e. forest, shrub, or herb-dominated), with forested wetlands considered to have greater function than an herb-dominated wetland. When functions and values are not known or measured, guidance states that a 1:1 credit to acreage replacement can be used as a surrogate and this ratio can be adjusted higher or lower depending on the relative functional values of the impacted wetlands compared to the replacement wetlands. Therefore, acreage ratios are part of the equation when estimating the effectiveness of mitigation in compensating for lost wetland function and the practice of trading higher-functioning wetlands for a greater acreage of lower-functioning wetlands is common.

Many studies have concluded that mitigation has not been successful in replacing lost wetland functions and values due to factors ranging from sites not being built in the first place, to not meeting agreed-upon performance standards, to not replacing the ecological functioning of lost wetlands (Mitch & Gosselink 2015; Moreno-Mateos 2012; Morgan & Hough 2015; NRC 2001).

There is also increased awareness that it takes time for wetland functions to develop once restoration is complete. The minimum time frame to expect the return of wetland functions may be on the order of 20 to 100 years (Zedler & Callahan 1997) but most wetland mitigation sites are only monitored for 5 to 10 years. One of the goals of this project is to use WDNRs monitoring and assessment methodology to evaluate the effectiveness of mitigation in a time frame longer than the required 5-10 year monitoring period. Now that compensatory mitigation has been established for almost 30 years in Wisconsin, it is possible to evaluate the outcome of some of these projects within a more appropriate time frame. Are restored wetlands replacing lost functional values 10 or 25 years after restoration took place?

4.3 The Development of Wetland Condition Assessment Methodology in Wisconsin

The premise of bioassessments like floristic quality assessment (FQA) is that taxa can be used as indicators of ecological health due to their differing capacities to withstand human-altered conditions (EPA 2002). As an ecosystem recovers from alteration, species that are sensitive to disturbed conditions, or "conservative", are expected to increase, and disturbance-tolerant species decrease. More natural hydrologic conditions and soils are expected to be reflected in the presence of more ecologically conservative plant species occurring in the wetland plant community. Differences in the number and dominance of conservative species over time or across treatments can thus be measured and compared quantitatively using FQA methodology.

The Wisconsin Department of Natural Resources (WDNR) began developing FQA methodology in 2002 with the assignment of Coefficients of Conservatism to Wisconsin's vascular plant flora (Bernthal 2003). Coefficients of Conservatism values (C-values) rate the degree of conservatism or sensitivity of each plant species to alteration on a scale of 0 to 10, with non-native or very tolerant species at the low end and species that are restricted to intact natural areas given a 10. These ratings are the foundation of FQA allowing plant inventories to serve as estimates of site integrity.

The assignment of C-values to Wisconsin's flora was followed by the development of mean coefficient of conservatism ($\overline{C}$) and Floristic Quality Index (FQI) benchmarks for plant communities in Southeast Wisconsin in 2006 (Bernthal et al. 2007). These benchmarks were then applied to evaluate restored wetlands in 2008 in the project, "Improving Wisconsin's Wetland Compensatory Mitigation Program: Factors Influencing Floristic Quality and Methods for Monitoring Wildlife" (Wilcox 2009).

Since that time the development of FQI benchmarks has been expanded across all four major ecoregions of the state (see Figure 1 for example) and quantitative estimates of plant cover were added, making cover-weighted metrics (denoted by adding a "w" to the beginning of the metric) such as weighted mean Coefficients of Conservatism (wC) and weighted FQI (wFQI) possible (Hlina et al. 2015, Marti & Bernthal 2019).

Through the process of developing floristic quality benchmarks WDNR has developed a database of vascular plant data and floristic quality metrics from over 1,100 natural wetlands across the state. Recently, accompanying soil chemistry data has been added to approximately one third of these wetlands, allowing comparisons of both floristic quality and soil parameters of natural wetlands to restored wetlands.

Natural Community		Condition Category				
		Least Disturbed			Most Disturbed	
		Excellent	Good	Fair	Poor	Very Poor
Emergent	Emergent Marsh	> 5.7	4.1 - 5.7	2.1 - 4.0	1.0 - 2.0	< 1.0
	Southern Sedge Meadow	> 6.3	5.6 - 6.3	3.8 - 5.5	1.0 - 3.7	< 1.0
	Wet-Mesic Prairie	> 5.5	4.6 - 5.5	3.1 - 4.5	1.9 - 3.0	< 1.9
	Calcareous Fen	> 7.0	6.2 - 7.0	3.6 - 6.1	2.2 - 3.5	< 2.2
Shrub	Shrub-Carr	> 5.1	4.7 - 5.1	3.2 - 4.6	2.3 - 3.1	< 2.3
Forested	Northern Hardwood Swamp	> 6.2	5.4 - 6.2	3.6 - 5.3	3.4 - 3.5	< 3.4
	Southern Hardwood Swamp	> 4.7	4.0 - 4.7	2.9 - 3.9	2.0 - 2.8	< 2.0
	Cedar Swamp	> 6.5	6.5	5.8 - 6.4	5.3 - 5.7	< 5.3
	Floodplain Forest	> 4.0	3.4 - 4.0	2.3 - 3.3	2.2	< 2.2

Figure 1. Condition benchmarks for wetland floristic quality based on Weighted Mean C (wC) for the Southeastern WI Till Plains and Central Corn Belt Plains Ecoregions of Wisconsin (Marti & Bernthal 2019).

4.4 Assessing Wetland Soil Organic Carbon

As a complement to assessing wetland functions related to floristic quality and condition, we also measured soil organic carbon (SOC) content in restored wetland soils. SOC is an important indicator of soil health and biogeochemical functioning, the transport, storage, and transformation of nutrients such as nitrogen, carbon, and phosphorus. SOC is highly associated with functions such as denitrification, carbon sequestration, and water quality improvement (Mitch & Gosselink 2015) considered among the most valuable functions of wetlands. SOC has several properties that contribute to these functions such as high pore-space size, high cation exchange capacity, and serves as the primary support structure and food source for microorganisms that perform the work of chemical transformation.

4.5 Improving Outcomes

In our previous report, "*Condition Outcomes in Wetlands Restored Using a Variety of Hydrologic Restoration Techniques: An application of Wisconsin DNR's FQA Methodology*" (Gibson et al 2019), we examined floristic quality and condition outcomes, focusing on the effects of hydrologic restoration technique. We found that in addition to hydrologic restoration technique, pre-restoration wetland drainage, and post-restoration maintenance were factors influencing restoration outcomes, however, small sample sizes combined with the large number of variables involved in wetland restoration outcomes, made it difficult to draw conclusions. With this additional data collected from mitigation wetland restorations, we hope that relationships will be clearer, allowing us to formulate better recommendations to guide wetland restoration practice.

4.6 Project Objectives

This study consists of two parts, one focusing on long-term vegetation and soil outcomes from randomly selected mitigation sites and the other on the factors influencing outcomes in wetland restoration more broadly. In Part I we asked, what level of wetland vegetation condition and biogeochemical functioning are mitigation wetlands achieving a minimum of 5 years after the required monitoring period has been completed? In Part II we asked, what restoration techniques and environmental variables are associated with successful vegetation outcomes in wetland restorations across a broad range of ages and restoration practices?

This study addressed these questions using FQA to measure biological attributes and soil organic matter as an estimate of biogeochemical functioning. Our hope is that the results will help to refine WDNR's FQA methodology for future use in evaluating restorations, aid in developing performance standards for wetland mitigation sites, evaluating wetland functions, and understanding the effectiveness and limitations of wetland compensatory mitigation in Wisconsin.

5 Methods

5.1 Site Selection

The goal of our sampling design was to obtain an unbiased sample of completed mitigation sites from the full range of wetland compensatory mitigation projects in Wisconsin. We used USACOE's RIBITS (U.S. Army Corps of Engineers' Regulatory In-lieu Fee and Bank Information Tracking System) which contains the most comprehensive list we could find. A limitation of RIBITS in this regard is that it does not include permittee-responsible or project specific sites, which did not generate extra credits beyond what was necessary for compensation of specified impacts, therefore these are not included in this study. We estimate that this may leave out more than 200 sites. However, most are extremely small projects compensating for individual impacts rather than the more modern practice of consolidating mitigation projects for many small impacts together on a larger site. Also, these project-specific mitigation sites were not well-documented, and, because the majority were associated with road construction, many were likely to be adjacent to highways and difficult or dangerous to access (WisDOT, pers. comm.).

The population of mitigation sites from which our sites were selected included 121 approved sites of all ages across Wisconsin (Figure 2). To evaluate mitigation sites in the long-term, sites were only accepted if it was determined to be a minimum of 5 years past its required monitoring period (typically 5 years, sometimes 10). Therefore, sites included in the study were a minimum of 10 years post-construction. In addition, because we were interested in evaluating regional differences in mitigation performance, site selection was stratified by the four major Omernik Level III Ecoregions: Northern Lakes and Forests, Northern Hardwood Forests, Southeastern Till Plains, and Driftless Area. The smaller ecoregions, Western Corn Belt Plains and Central Corn Belt Plains were joined with the Driftless Area and Southeastern WI Till Plains, respectively.

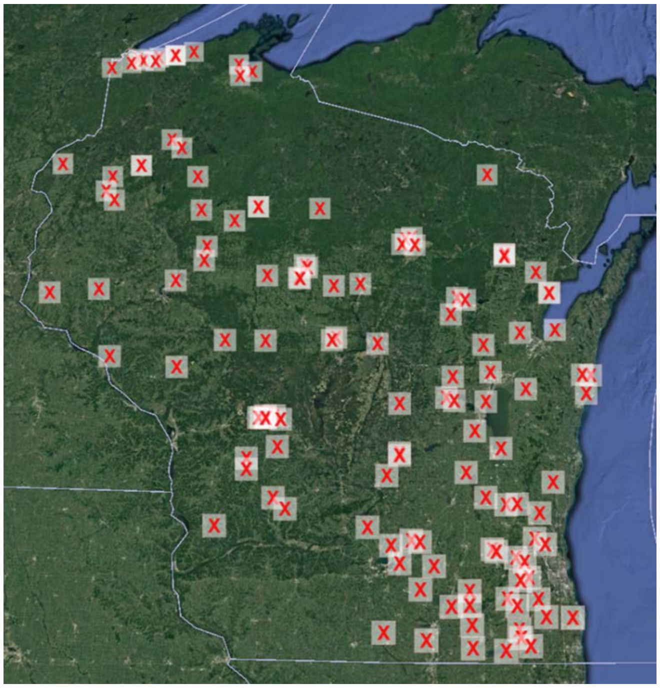

Figure 2. Population of 121 approved wetland mitigation sites listed on USACOE's RIBITS website in 2018. Approximately 76% of sites were created to mitigate wetland losses associated with WisDOT projects.

Source: https://ribits.ops.usace.army.mil/ accessed 3/2018.

5.2 Field Sampling Methods

Site visits to collect vegetation and soil data took place between mid-June and mid-September of the years 2018 and 2019. In comparison with the 1981-2010 period, Wisconsin experienced higher than normal precipitation during the two sampling years and the prior year, 2017. In 2018 some areas of northwest Wisconsin received up to 4" less precipitation than normal but the remainder of the state ranged from 0" to 24" more than the 1981-2010 period. In 2019 all areas of Wisconsin received excessive precipitation ranging from 4" to 20" greater than normal (data from Wisconsin State Climatology Office).

Vegetation sampling

All unique wetland communities found within the mitigation site were assessed. Assessment areas (AA's) were defined as areas of homogeneous vegetation a minimum of a quarter acre (0.1 ha) in size. AAs were surveyed using Wisconsin DNR's *Timed-Meander Sampling Protocol for Wetland Floristic Quality* (Trochlell 2016). All vascular plant species were recorded as they were encountered while moving through the plant community. Surveys were ended once a five-minute period had elapsed with no new species (or less than 5% of the total) encountered and no additional areas of diversity were visible. We then estimated the areal cover (from 1 – 100%) for each plant species on the list.

Soil sampling

Soils were collected and described for a minimum of one AA per community type per mitigation site. A soil description and soil sample were collected from the AA in an area with vegetation typical of the AA. Soil horizons were described by depth, color, and texture to a minimum of 18" and the depth to groundwater and saturation was noted, typically after 10 to 20 minutes had elapsed. The process of digging and describing the soil followed the *1987 Corps of Engineers Wetland Delineation Manual* and its respective regional supplements in Wisconsin. Soil samples consisted of approximately a quarter-gallon of soil collected from the top 10 inches of the soil horizon, placed in a plastic bag, and kept on ice until it could be transported to the University of Wisconsin – Madison's Soil and Forage Analysis Laboratory.

5.3 Analysis

Vegetation

Two metrics were used for analysis: wC (cover-weighted mean C-value) and condition category based on wC (EXCELLENT, GOOD, FAIR, POOR, and VERY POOR) using WDNRs FQA benchmarks specific to each wetland's ecoregion and community type. To simplify, the five condition tiers were converted to a binary success rate: "FAIR+" for FAIR, GOOD, or EXCELLENT condition, or "POOR-" for POOR and VERY POOR results.

The wC value for each AA was calculated from timed meander survey data using WDNR's Floristic Quality Calculator (Gibson 2017). AA's were classified by wetland type according to two classification systems, WDNR's Natural Community Classification system using WDNR's "Key to Wetland Communities of Wisconsin" (O'Connor 2020), and Eggers & Reed (2015).

Using the AA's natural community classification (Table 1) and ecoregion, a condition category (EXCELLENT, GOOD, FAIR, POOR or VERY POOR) was assigned using FQA benchmarks for wC (Hlina et al 2015; Marti & Bernthal 2019). In cases where no condition benchmarks have been developed for a community/ecoregion combination, the closest available benchmarks geographically and ecologically were substituted. See Appendix A for benchmarks used to assign condition to wetlands in this study.

Table 1. Cross-walk between Eggers & Reed (2015) community types found on mitigation sites and WDNR natural community types with calculated condition benchmarks (Marti & Bernthal 2019). Also shown are Omernik Ecoregions where condition benchmarks for the wetland community are available for use.

Eggers & Reed Community Type	WI Natural Community	Ecoregions Available*
Shallow Marsh & Deep Marsh	Emergent Marsh	All
Sedge Meadow & Fresh (Wet) Meadow	Southern Sedge Meadow	SETP, DRFT, NCHF
	Northern Sedge Meadow	NCHF, NLF
Wet to Wet-mesic Prairie	Wet-Mesic Prairie	SETP
Shrub-carr	Shrub-carr	All
Alder Thicket	Alder Thicket	DRFT, NCHF, NLF
Floodplain Forest	Floodplain Forest	DRFT, SETP
Hardwood Forest	Southern Hardwood Swamp	SETP
	Northern Hardwood Swamp	SETP, NCHF, NLF
Coniferous Bog	Black Spruce/Northern Tamarack Swamp	NCHF, NLF
Open Bog	Open Bog	NLF
Shallow, Open Water	Submergent Marsh	None

*SETP = Southeastern Wisconsin Till Plains; DRFT = Driftless Area; NCHF = North Central Hardwood Forests; NLF = Northern Lakes and Forests.

For the purposes of this study we define a successful restoration as one achieving a plant community condition outcome of FAIR or better, as determined using wC benchmarks. Results from our previous study (Gibson et al 2019) showed that FAIR condition wetlands are dominated by native species (although non-native invasive species may be present) and appear to be a reasonably achievable objective of restoration, with approximately half of all restored wetland AAs surveyed for our two studies meeting this benchmark.

Acreage estimates of community types

To map the boundaries of wetland communities, GPS tracks from timed meander surveys were superimposed onto high-resolution aerial imagery from Google Earth or natural color, leaf-off imagery from the 2015 WI Regional Orthophoto Consortium (WROC). In addition, when available, LiDAR (Light Detection and Ranging) imagery was used to detect fine-scale changes in elevation to improve mapping resolution. While our objective was to map the entire mitigation site, some areas did not have enough information to confidently estimate the community type and were left unmapped.

Soil analysis

Soil samples were sent to University of Wisconsin – Madison's Soil and Forage Analysis Laboratory in Marshfield, WI for analyses. Samples were analyzed for pH using a 1:2 soil to water extraction; percent total phosphorus (TP) using a nitric/peroxide method; and percent organic matter (% OM) using the weight loss-on-ignition (LOI 360 degrees) technique. Total nitrogen (TN), total carbon (STC) and total organic carbon (SOC) percent dry weight were determined using dry combustion.

Soils were classified as Mineral, Mucky Mineral, or Organic using notes from in-field soil texturing and laboratory measurement of total organic carbon (TOC) from the top 10 cm of soil (Marti 2016; USDA 2018). Soils containing less than 5% SOC were considered to be mineral, soils between 5% and 12% SOC were designated as either mineral or mucky mineral depending on soil texture description; soils between 12 and 18% SOC were either mucky mineral or organic based on soil texture, and soils with TOC greater than 18% were all considered to be organic. Mineral soils were further divided into two groups based on their in-field soil texture as either predominantly silts and clays (silt loams, clay loams, silty clay loams, silty clay) or predominantly sand (sandy loams, sandy clay loams, loamy sands).

Comparison with other soil datasets

Soil carbon measurements from three natural wetlands datasets from Wisconsin were accessed for comparison:

1) The 2011 National Wetland Condition Assessment (USEPA 2016) which contains data from wetlands selected probabilistically from across the nation, with 20 sites falling within Wisconsin. We selected only data collected from the topmost layer of soil samples from each probability site.

2) The 2012 Wisconsin Intensification Study (Marti & Bernthal 2016), which includes data from 46 wetlands selected probabilistically from Wisconsin's Lake Michigan basin.

3) The WDNR wetland FQA benchmark project dataset (Hlina et al 2015; Marti & Bernthal, 2019) which has floristic data from 1100 wetlands and soil data from the top 15 cm of 337 of those wetlands. Wetlands from this dataset were targeted to include equal numbers of most-disturbed and least-disturbed sites from a range of community types from all level III Omernik ecoregions with a slight bias toward the southern half of the states.

In addition, we used data collected from 35 restored wetlands from our previous study, the "Restoration Techniques Study" (Gibson et al 2019) that were restored by non-profit groups or for wildlife habitat improvement in addition to some mitigation projects.

Hydrology

Hydrology was documented as "depth to groundwater" and "depth to saturation" during the soil profile description to a minimum of 18" depth. Observations were made between June 16 to October 3, during the years 2018– 2019 for the Mitigation Study, and 2016 – 2017 for the Restoration Study AAs with most observations concentrated within the months of July and August (Figure 3a). Observations of depth to saturation and depth to water table were grouped into 4 categories (Figure 3b).

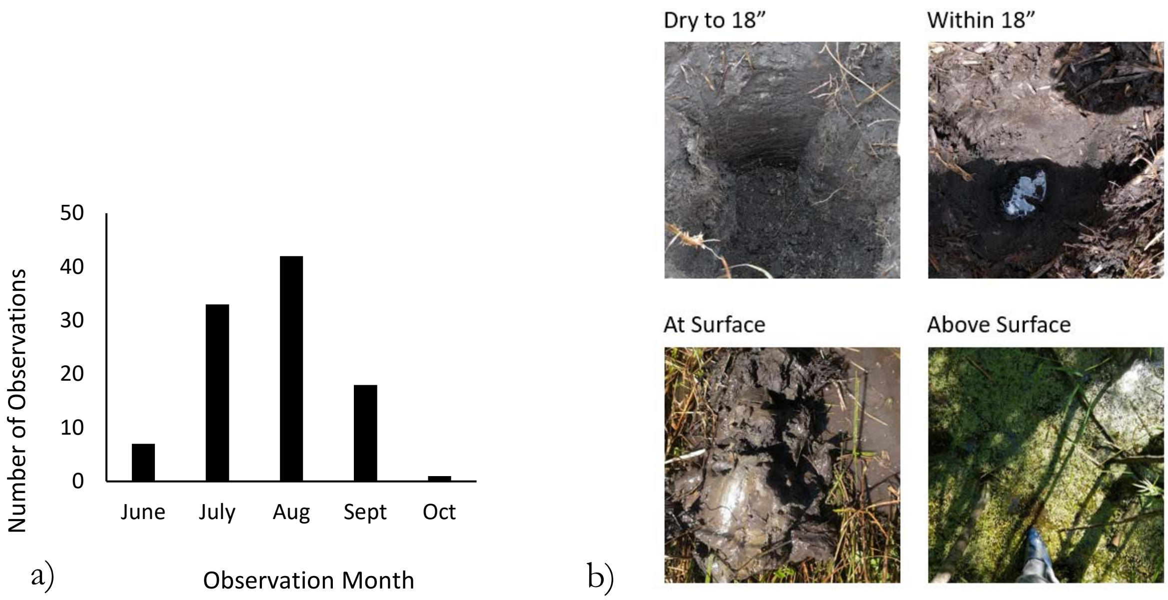

Figure 3. Soil saturation observations: a) distribution by Month 2016 -2019; and b) a visual depiction of assigned categories as viewed during soil profile descriptions.

Active site maintenance

Information about site maintenance since completion of the restoration was provided by site managers. Managers were asked if there has been any ongoing maintenance to the plant community (invasive species control, burning, or mowing) and their answers recorded as "Yes" or "No." Maintenance activities that only occurred once or a few times and were not ongoing or recent (within the past 5 years) were recorded as "No."

Pre-restoration state: fully – drained, partially- drained, or non-wetland

Pre-restoration state was categorized for each AA as either partially-drained wetland, fully-drained wetland, or non-wetland. This is ecologically significant as well as the primary factor used by regulatory agencies to assign restoration approach (i.e. re-establishment, rehabilitation, or creation) and assign restoration credit ratios. The mapped Natural Resources Conservation Service (NRCS) soil type was used to indicate if the restored area had historically hydric soils. Wetland delineation reports of the site prior to restoration activities indicated fully-drained areas (i.e. upland) and wetland areas. Wetland areas existing in proximity to drainage structures were considered partially-drained. In the absence of a wetland delineation**,** historical imagery, including 1937-41 aerial imagery and Google Earth historical imagery, was used to assess conditions based on land use. Cranberry farms were an exception in that cultivation was not assumed to indicate drained conditions. Using aerial imagery alone may have missed farmed wetlands and resulted in some underestimation of partially-drained wetland conditions.

Restoration Technique

Individual techniques (e.g. ditch plugging, scrapes) used to develop wetland hydrology were determined mainly from site plans or monitoring reports. These were later grouped into broader categories. Those that added impounding structures or excavations were placed in a "surface modification" group and those that removed alterations to restore historical wetland hydrology placed in the "alteration removal" group. The alteration removal group was further divided to separate those that attempted to completely remove the alteration versus those that disabled them partially.

Analysis of factors influencing restoration outcome

For the analysis of factors influencing restoration outcome we combined the data collected for this study with data from wetland restorations previously collected for the "Restoration Techniques" study (Gibson et al 2019). Sampling methods were similar, however, restoration sites for the previous study were selected by their use of specific restoration techniques including less commonly-used techniques such as complete tile system removal, complete ditch filling, and legacy sediment removal. Sites also came from a wider assortment of restoration practitioners (including non-profit groups and wildlife habitat restoration projects) and ranged in age from 4 to 22 years post construction, in contrast to the 10-year minimum age of sites selected for the mitigation study.

Two response variables and 10 explanatory variables (Table 2) were used in the following analyses:

1. To look for differences in outcomes between groups: Proportion tests for binomial data (Fair+/Poor- or success rate); ANOVAs and t-tests for categorical variables.
2. To look for relationships between explanatory variables and response variables we used linear regression for the two continuously variable explanatory variables, TOC and age, when paired with wC or logistic regression when paired with success rate.
3. To estimate the effects of different factors on success rate and wC we calculated the difference between success rate or mean wC of the factor and the mean success rate or wC of all samples.

To sum up our results we divided the 9 factors of interest (excluding age) into two groups based on practical significance: "Site Selection Factors" includes environmental variables associated with prospective restoration sites that a restoration practitioner cannot normally control once a site is selected: ecoregion, soil type, soil pH, soil organic carbon content and drainage condition. "Restoration and Management Factors" includes factors that a practitioner may have some ability to influence. These include hydrologic restoration technique, post-restoration maintenance, summer saturation level, and community type.

Table 2. List of collected data and each variable's category or range used in analyses.

Explanatory Variables	**Categories/Range**
Hydrologic Restoration Technique	Excavation/Impoundment, Partial Alteration Removal, Complete Alteration Removal
Active Maintenance	Yes, No
Pre-restoration Drainage Status	Fully-drained, Partly-Drained
Ecoregion	SETP, DA, NCHF, NLF
Community	Forest, Shrub, Marsh, Meadow, Prairie
Soil Type	Sandy, Silty/Clayey, Mucky Mineral, Organic
Soil pH	Alkaline, Neutral, Moderately Acid, Strongly Acid; (4.6 – 8.0)
Soil Organic Carbon (% dry wt.)	$<10\%$, $\geq 10\%$ / (0.7 to 28.2%)
Summer Hydrology	Above Surface, At Surface, Within 18" Below, >18" Below Surface
Age (yrs.)	<10, 10 to 19, ≥ 20 / (4 to 26yrs).
Response Variables	
wC	0.2 - 8.1
Success Rate	1 (Fair+) or 0 (Poor-)

6 Results Part I: Long-term Ecological Outcomes on Mitigation Sites

6.1 Attributes of Selected Sites

A total of 15 mitigation sites (Figure 4; Table 3) were randomly drawn from the list of 121 Corps-approved mitigation projects and met the criteria for inclusion. The most common reason for rejection was not meeting the age requirement, but at least one was rejected because it was never built, another was built but not used for compensation, and a few did not have enough information available to accept or reject. Selected mitigation sites ranged in age from 13 to 26 years since construction, with construction years spanning 1992 to 2005. One selection included two separate sites 5 miles apart from each other that were developed simultaneously, making the total number of surveyed sites 16. The smallest mitigation site selected was 4 acres in size and the largest was 465 acres. The 15 selected sites together contained approximately 1011 acres of wetland to compensate for the loss of 558 acres of wetlands thus far (half of sites are still open to credit sales). Most sites were built to compensate for losses due to road construction (11), but other losses were due to school construction, landfill expansion, or were part of a commercial bank compensating for many small impacts from a variety of sources. From these 15 sites we completed 89 vegetation surveys and collected 72 soil samples.

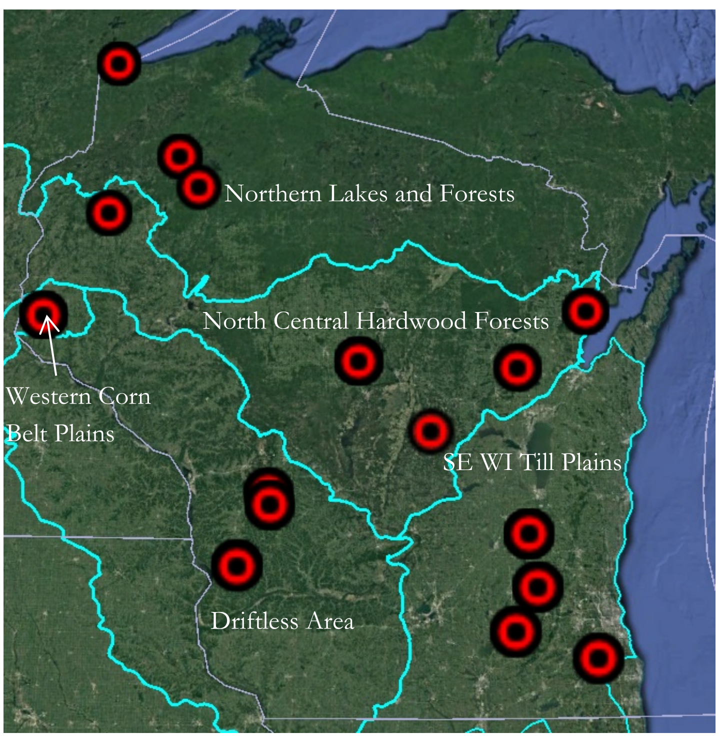

Figure 4. Location of mitigation sites selected for this study. Blue lines outline Omernik Level III Ecoregions.

Table 3. Selected mitigation sites by name, developer, county, level III ecoregion, construction year, credit sales status in RIBITS with available credits for open sites shown, and property size. Two sites, Jug Creek and Wildcat, are considered as one site although they are 5 miles apart.

Site Name	Developer	County	Level III Ecoregion	Construction Year	Status	Size (ac)
Jug Creek	WisDOT	Vernon	Driftless Area	2002	Closed	10
Wildcat	WisDOT	Vernon	Driftless Area	2002	Open (21.5)	40
Veolia Glacier Ridge SE	Veolia Landfill	Dodge	Southeastern WI Till Plains	2005	Open (22.5)	57
Sikma	WisDOT	Oconto	Northern Lakes & Forests	1992	Closed	95
Shiocton	WisDOT	Outagamie	North Central Hardwood Forests	1994	Closed	465
Weirgor Core 48	WisDOT	Sawyer	Northern Lakes & Forests	1993	Closed	5
Princes Point	WisDOT	Jefferson	Southeastern WI Till Plains	1996	Closed	71
Branca	WisDOT	Barron	North Central Hardwood Forests	1995	Open (3.4)	236.5
Pechacek-Gilbertson	WisDOT	Pierce	Western Corn Belt Plains	1996/1997	Open (4.3)	108.6
Pickle Row	WisDOT	Waushara	North Central Hardwood Forests	1997	Closed	4
State Hwy 20	WisDOT	Racine	Southeastern WI Till Plains	1998	Open (1.5)	10
Bell Center	WisDOT	Crawford	Driftless Area	1998	Open (72.0)	146.8
Northland Cranberries	Private Commercial Bank	Wood	North Central Hardwood Forests	1998	Open (2.7)	156
Lang Road	WisDOT	Waukesha	Southeastern WI Till Plains	2002	Closed	80
Crawford Creek	City of Superior School District	Douglas	Northern Lakes & Forests	2003	Open (0.0)	39
Upper Chippewa	Private Commercial Bank	Sawyer	Northern Lakes & Forests	2004	Closed	47.4

Restoration techniques

Two thirds of sites used surface modifications, including impoundments, berms, scrapes, excavations, and water control structures to create wetland hydrology. Seven sites (47%) used partial alteration removal techniques, ditch plugs, tile breaks, dike cuts and 6 sites (40%) used complete alteration removal techniques such as ditch filling, dike removal, tile removal or roadbed removal.

The most common individual techniques used on the 15 sites were impoundments (5 sites), scrapes (4), and ditch plugs (4). The technique with the highest number of separately surveyed wetland communities was ditch plugs, which was associated with 25% of AAs, followed by impoundments, excavations, scrapes, and tile breaks. Although 40% of sites used a complete alteration removal technique, these impacted only 17% of the AAs we surveyed.

Table 4. Techniques used to develop wetland hydrology on 15 mitigation sites listed as individual techniques and in larger technique groupings. Most sites used multiple techniques.

Technique	# Sites	# AAs
Surface Modifications	**10**	**32**
Impoundment (>3ft)	5	14
Berm (< 3ft)	2	7
Scrape (< 3 ft)	4	12
Excavation (> 3ft)	3	13
Terracing	1	3
Water Control	2	3
Partial Alteration Removal	**7**	**28**
Ditch plug	4	18
Transverse disking/plowing	2	6
Tile Breaks	2	12
Dike Cuts	1	4
Complete Alteration Removal	**6**	**12**
Ditch Fill	3	8
Natural Levee Repair	1	5
Dike Removal	1	4
Roadbed Removal	1	2
Tile Removal	1	1

Credit generation approach/ initial drainage conditions

Re-establishment, restoring hydrology to fully-drained areas, was the most common approach comprising 62% of all surveyed AAs (Table 5). Rehabilitation, restoring hydrology to partially-drained wetlands occurred on 22 AAs but only one site, a former cranberry farm, was exclusively rehabilitation. Creation was employed on only two sites and only on a small portion of the total area of these sites. Areas of preservation and buffer were surveyed on some sites, but we did not attempt to survey all such areas on all sites, so the results shown are not comprehensive for these approach types.

Table 5. Credit generation approach assigned to individual AAs based on pre-restoration conditions and the presence of hydrologic restoration techniques. Most sites used more than one credit generation approach. '*' areas of preservation and buffer were not surveyed on all sites.

Credit generation approach	# Sites	# AAs
Creation	2	4
Re-establishment	13	55
Rehabilitation	9	22
Preservation*	3	5
Buffer*	2	3

Hydrology

Most wetland AAs from mitigation sites were either saturated to the surface or no water was observed in the 18" soil profile after approximately 20 minutes. Ponded conditions were found in 9 out of 44 observations and only 4 AAs were saturated within 18" below the surface.

Table 6. Results of saturation level observations from 44 AAs from the mitigation sites. Also shown are observations from 57 AAs from wetland restorations from the restoration techniques study (2016-17).

Category	Observed Hydrology	Mitigation Samples	Restoration Techniques Study Samples	Total
Above surface	Standing surface water greater than 1".	9	0	9
At Surface	Soil saturated to within 1" of surface.	15	23	38
< 18" Below	Soil saturation or water table within the top 18" of soil profile.	4	8	12
Dry to 18" Below	No water observed in top 18" of soil profile.	16	26	42
	TOTALS:	44	57	101

6.2 Plant Community Condition Results

Community types and acreage estimates

Using Eggers and Reed's (2015) classification system, eleven different community types were encountered during site visits. The most common community type was shallow open water (23) however, these were not surveyed due to the absence of an approved protocol and floristic quality benchmarks for condition assignment. After shallow open water, the most common community type encountered was shallow marsh (17), followed by fresh (wet) meadow (disturbed subtype) (12) and sedge meadow (12). Other types include wet to wet-mesic prairie (9), shrub-carr (9), and with lesser frequency, fresh (wet) meadow (native subtype), floodplain forest, hardwood swamp, open bog, and alder thicket.

A total of 1011 acres of developed wetland were mapped out of a total of 1571 acres of land and 1095 acres of released developed wetland acres listed on RIBITS for these 15 sites. Mapped and surveyed wetland acreage was less than property acreage and released wetland acres likely because 1) small areas (<30 acres total) were not mapped because the wetland community type was unclear from aerial imagery, and 2) areas within mitigation sites that were not used to generate mitigation credits because they were preservation or part of an upland buffer were also not mapped. For instance, 189 acres of the Branca site and 76 acres of the Lang site were upland areas not used for wetland credits and 70 acres of the Bell Center site was pre-existing wetland not used to generate wetland credits.

While shallow open water was the most frequently encountered type, shallow marsh had the greatest areal extent (Figure 5), covering approximately 387 acres, or 41% of the restored area from the 15 mitigation sites. Fresh (wet) meadow (disturbed subtype) was the second most extensive type, comprising a minimum of 162 acres, or 17% of the total wetland development area.

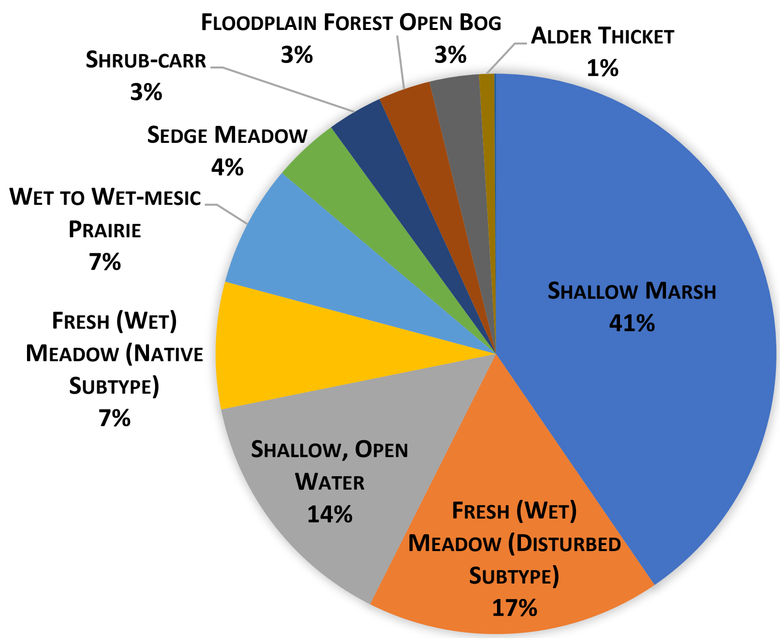

Figure 5. Summed acreage of all 11 wetland community types encountered on 15 mitigation sites using Eggers and Reed (2015) wetland community classification. Hardwood Swamp comprised less than 1% and is not shown.

Ecological condition of restored wetland communities

Weighted mean C-values (wC) ranged from 0.2 in a *Typha X glauca* marsh in the Driftless Area to 8.1 in an open bog restored from former cranberry production. Across all wetland communities assessed, mean wC was 2.9. Condition results determined using benchmarks for wC spanned the full range from VERY POOR to EXCELLENT (Figure 6). VERY POOR was the most common outcome (35%), and EXCELLENT the least (3%). Overall, 45% of wetland AAs were in FAIR or better condition and 55% were in POOR or worse condition.

Although some restored wetlands fell within each of the five condition categories, restorations in POOR condition had the greatest acreage extent, with 347 acres or 46% of the total acres measured (Figure 7). VERY POOR condition wetlands had the 2nd highest areal extent with 187.3 acres (25%). Overall, 218.2 acres (28%) were in FAIR or better condition and 534 acres (72%) were in POOR or VERY POOR condition.

Figure 6. Condition results by frequency. Condition categories are derived from benchmarks for wC (cover-weighted coefficient of conservatism) (Marti & Bernthal 2019). Shown as a percentage of 80 restored wetland communities surveyed from 15 mitigation sites. Does not include 23 shallow, open water areas.

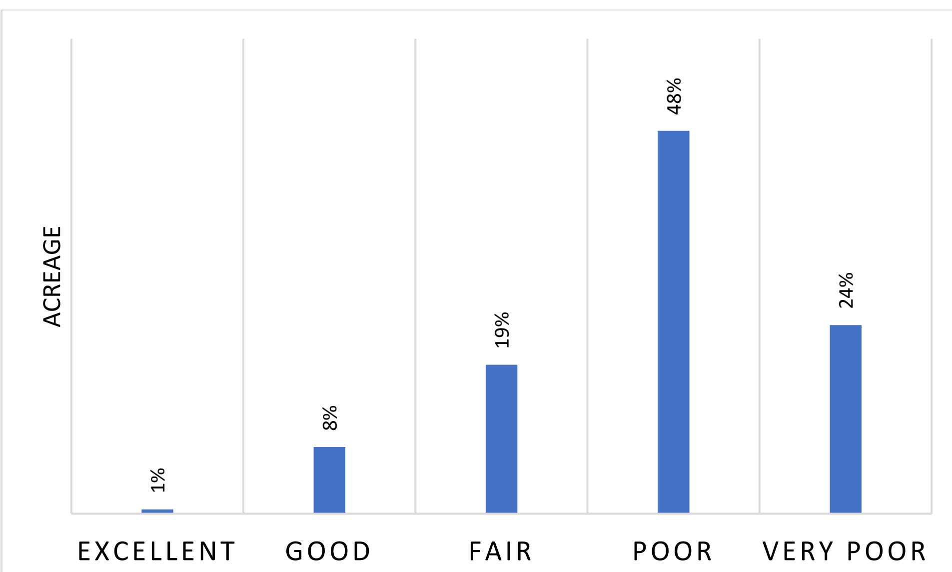

Figure 7. Condition results by acreage extent. Condition categories are derived from benchmarks for wC (cover-weighted coefficient of conservatism) (Marti & Bernthal 2019). Total acreage = 742. Does not include 138 acres of shallow, open water.

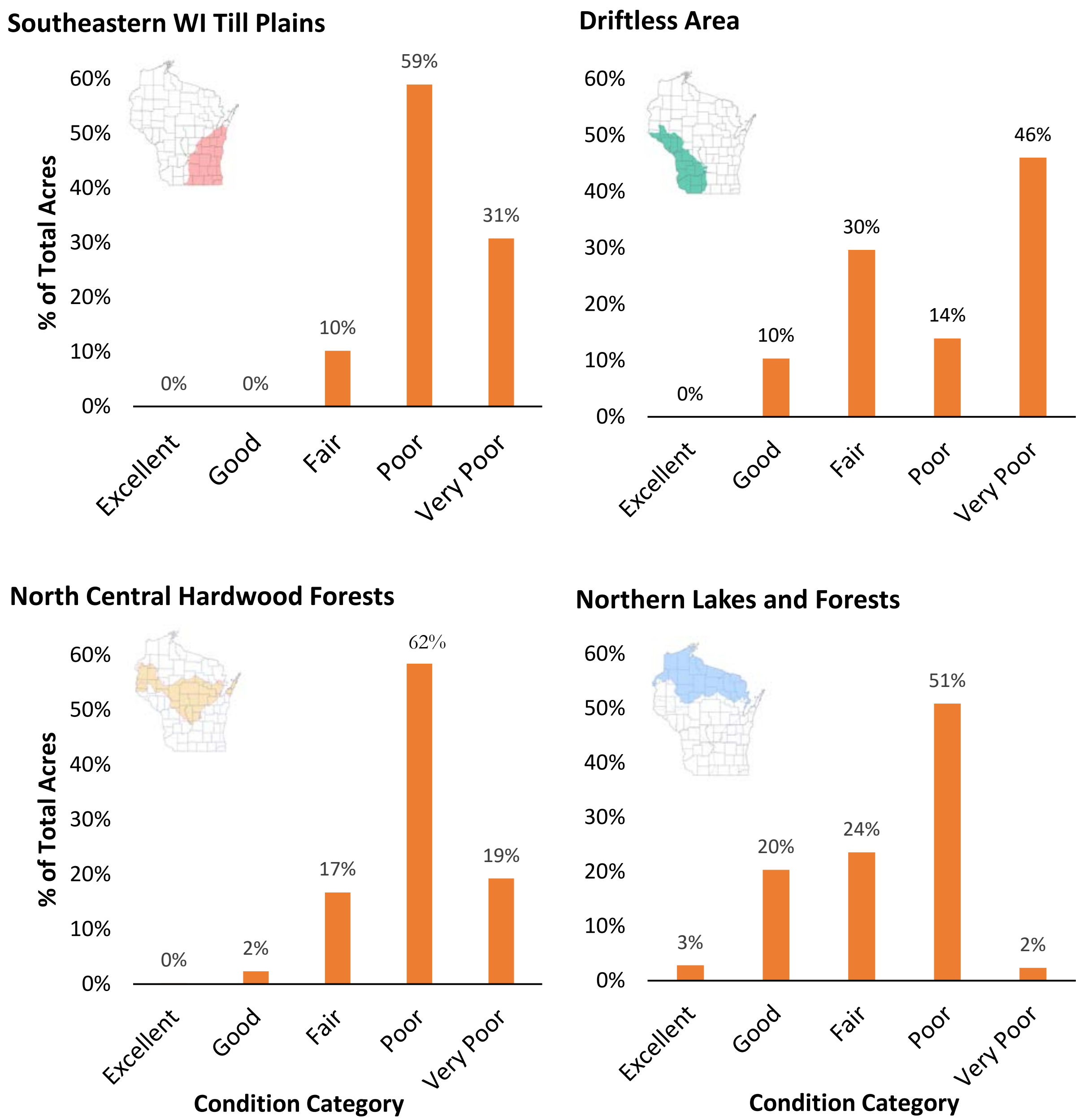

Figure 8. Condition results determined using benchmarks for wC by acreage from mitigation sites drawn from each of Wisconsin's four major Omernik Level III Ecoregions. Survey results from the Western Corn Belt Plains (1 site) are combined with Driftless Area results.

Wetland condition was lowest on sites from the Southeastern WI Till Plains with only 10% of wetland acreage falling in the FAIR or better category (Figure 8). In contrast, the Northern Lakes and Forests mitigation sites were found to have 63% of restored wetland acreage in FAIR or better condition. Forty (40%) percent of Driftless Area mitigation sites were in FAIR or better condition but this ecoregion also had the highest proportion of wetland area in VERY POOR condition (46%). Only 20% of North Central Hardwood Forests ecoregion wetlands were in FAIR or better condition. This ecoregion had the highest proportion of restored wetland in POOR condition (62%); however, this can be attributed to a single large site.

Maintenance

Only 3 out of the 15 sites reported maintenance activities taking place on some portion of their mitigation site. Within these 3 sites only 5 AAs were affected (Table 7). Only two sites were under regular management, one by the federally-owned Kickapoo Valley Reserve, and one a DNR-owned property managed for waterfowl. The remaining site that reported maintenance had been visited once or twice to spray areas of giant reed grass (*Phragmites australis)* using federal GLRI (Great Lakes Restoration Initiative) funds. The 5 AA's with management or invasive species treatments did not necessarily have better condition outcomes, with only one exceeding the FAIR or better benchmark.

Conversely, of the 519.2 acres (75 wetland AAs) that were lacking maintenance, 34 wetlands covering 49% of the total acreage were in FAIR or better condition.

Table 7. Reported presence or absence of maintenance activities on mitigation sites, showing the number of wetland AAs, acreage, and condition results as FAIR or better (FAIR+) or POOR/VERY POOR (POOR-).

	Maintenance	
	Absent	**Present**
# AAs	75	5
Total Acres	519.2	238.2
# FAIR+	34	1
# POOR-	41	4
Acres Fair+	254.8 (49%)	30.5 (13%)
Acres Poor-	263.4 (51%)	207.7 (87%)

Composition of dominant plant species in restored wetlands

A total of 43 different plant species dominated (had a minimum 20% estimated areal cover) in at least one restored wetland from the 15 mitigation sites, and only 5 were non-native (Table 8). Most shallow marshes were dominated by *Typha X glauca* (hybrid cattail), but a few had *Sparganium emersum* (narrow-leaved bur-reed), *Sparganium eurycarpum* (common bur-reed), *Phragmites australis subsp. australis* (common reed), or *Schoenoplectus tabernaemontani* (soft-stem bulrush) as a dominant. Fresh (wet) meadows - disturbed subtype were dominated by *Phalaris arundinacea* (reed canary grass), but native subtypes were dominated by *Scirpus cyperinus* (wool-grass), *Scirpus atrovirens* (dark-green bulrush), *or Symphyotrichum puniceum* (swamp aster). Dominants of restored sedge meadows were *Carex lacustris* (lake sedge), *Carex stricta* (tussock sedge), *Carex utriculata* (yellow lake sedge), *Carex trichocarpa* (hairy-fruit sedge), *Carex pellita* (broad-leaved woolly sedge), *Carex atherodes* (hairy-leaved lake sedge), or *Carex lasiocarpa* (narrow-leaved wooly sedge). Wet to wet-mesic prairies were dominated by *Andropogon gerardii* (big bluestem), *Solidago canadensis* (Canada goldenrod), *Silphium perfoliatum* (cup-plant), *Calamagrostis canadensis* (blue-joint grass), *Phalaris arundinacea*, or *Poa pratensis* (Kentucky blue-grass).

The two restored wetlands in excellent condition consisted of one emergent marsh dominated by *Sparganium emersum* and *Glyceria borealis* (northern manna grass), and a sedge meadow dominated by *Carex atherodes* and *Carex lasiocarpa*. At the other extreme, wetland restorations in very poor condition were dominated by *Phalaris arundinacea*, *Typha x glauca*, *Solidago canadensis or S. gigantea* (giant goldenrod) in the herb layer and for shrub or forested communities, *Acer negundo* (box elder), *Alnus incana* (tag alder), *Salix interior* (sandbar willow), or *Cornus foemina* (gray dogwood).

Successful restored plant communities

Overall, 45% of restored wetland communities surveyed achieved FAIR, GOOD, or EXCELLENT condition 13-26 years post-construction, however, these communities tended to be small, covering only a total of 28% of the areal extent of the 15 mitigation sites.

Two assessed wetlands had achieved EXCELLENT condition, the highest condition tier. These wetlands were both in northern WI, one 23- year old emergent marsh and one a 16 year old sedge. Both wetlands were receiving surface water inputs from high-quality natural wetlands adjacent to or upstream of the mitigation site.

In addition to the two EXCELLENT wetlands, 9 restored wetlands achieved GOOD condition. One of these, a site in the Western Corn Belt Plains ecoregion (Pierce County) in an area with high groundwater gradients, resulted in a good-quality sedge meadow 22 years after plowing stopped and ditches were plugged. The remainder of GOOD outcomes occurred on areas less severely impacted by historical drainage and plowing than is typical. In the northernmost ecoregion an alder thicket in GOOD condition developed 25 years after a roadbed was removed. Partially removing dikes from a cranberry farm resulted in GOOD quality open bog and sedge meadow after 15 years or less. In the North Central Hardwood Forests Ecoregion, a GOOD quality shrub-carr developed from a former wet area on a field 20 years after shallow berms were constructed and the area was plowed across the slope to retain water. Further south in the Driftless Area an unsuccessfully drained area on a plowed field resulted in a GOOD quality sedge meadow 16 years after tile was removed.

Some commonalities of these high-achieving restorations, besides a tendency to be in the northern part of the state, were surface water inputs from high-quality natural wetlands and fully-saturated soils in mid-summer.

Restorations resulted in GOOD or EXCELLENT outcomes in 14% of all assessed restored areas. However, these wetland areas tended to be small, with the combined area of GOOD and EXCELLENT outcomes covering only 45.4 acres, or 7% of the total restored wetland area from the 15 mitigation sites.

FAIR condition outcomes were more common, with 22 (29%) of wetland AAs meeting this benchmark and a combined area of 148 acres, or 19% of the total restored wetland area. FAIR condition wetlands were found in all ecoregions though were least common in the Southeast WI Till Plains. At the high end of this group was an open bog restored from a cranberry farm, a shallow marsh with *Schoenoplectus tabernaemontani*, and sedge meadows dominated by *Carex lacustris* or *Carex stricta (*See Table 7 for dominant species by condition). At the low end was a shrub-carr dominated by *Salix discolor* (pussy willow) and *Solidago canadensis* as well as a sedge meadow with dominance shared between *Carex haydenii* (long-scaled tussock sedge) and two non-native grasses.

Less successful restoration outcomes: Poor and Very Poor results

Over half of surveyed communities (55%) fell in POOR and VERY POOR categories. These communities tended to be large, comprising 70% of the total restored wetland area, 534 acres (Figures 6 and 7). Wetland restorations with POOR results at the high end of wC scores were dominated by a mix of natives with low

Table 8. Plant community dominants 13 to 26 years later by condition. Species listed below dominated (by areal cover) a stratum of at least one surveyed plant community in the indicated condition category. The number of restored wetland AAs falling under each condition category is indicated. '*' indicates a non-native species. Species in bold were listed as part of a seed mix or plantings for at least one site in which the species dominated.

Excellent Condition (n = 2)		
Carex atherodes *Carex lasiocarpa* *Sparganium emersum* *Glyceria borealis*		
Good Condition (n = 9)		
Herb layer: *Carex lacustris* *Carex trichocarpa* *Carex stricta* *Carex utriculata* *Carex haydenii* *Chamaedaphne calyculata* *Vaccinium macrocarpon*** *Scirpus cyperinus*		Tree/Shrub layer: ***Larix laricina*** *Alnus incana* *Salix petiolaris*
Fair Condition (n = 23)		
Herb layer: ***Andropogon gerardii*** *Boehmeria cylindrica* ***Calamagrostis canadensis*** *Carex lacustris* *Carex stricta* *Impatiens capensis* *Phalaris arundinacea** *Schoenoplectus tabernaemontani* ***Scirpus atrovirens***	Herb layer continued: ***Scirpus cyperinus*** *Scirpus microcarpus* ***Silphium perfoliatum*** *Solidago canadensis* *Sparganium eurycarpum* *Spirea alba* *Symphyotrichum puniceum* *Vaccinium macrocarpon***	Tree/Shrub layer *Populus deltoides* *Populus tremuloides* ***Quercus bicolor*** *Salix discolor* *Salix eriocephala*
Poor Condition (n = 17)		
Herb layer: *Phalaris arundinacea** *Poa pratensis** *Typha X glauca** *Solidago canadensis* *Phragmites australis subsp. australis**		Tree/Shrub layer *Acer saccharinum* *Fraxinus pennsylvanica* *Salix interior*
Very Poor Condition (n = 24)		
Herb layer: *Phalaris arundinacea** *Typha X glauca** *Solidago canadensis*		Tree/Shrub layer: *Acer negundo* *Alnus incana* *Salix interior* *Cornus foemina*

** *Vaccinium macrocarpon* was likely a horticultural variety remaining from cranberry cultivation.

conservatism values and non-invasive non-natives, for instance a fresh (wet) meadow dominated by *Solidago canadensis* and *Agrostis gigantea*. At the low end they were dominated exclusively by *Typha X glauca*. VERY POOR wetlands included floodplain forests dominated by *Acer negundo* and *Phalaris arundinacea*, shrub-carrs dominated by *Salix interior* and *Phalaris*. However, for the most part VERY POOR wetlands were meadows and shallow marshes dominated by *Phalaris arundinacea* and *Typha X glauca,* respectively.

Problem species: Hybrid cat-tail, reed canary grass, and sandbar willow

Hybrid cat-tail *(Typha X glauca)*, the hybrid of native *Typha latifolia* and introduced *Typha angustifolia*, dominated 88% of the total area of shallow marsh developed. We estimated 300 acres of wetland were dominated by *Typha X glauca* on these 15 mitigation sites. This species was found state-wide in all ecoregions, even in Douglas county in the far northwest. Only one site, currently managed by DNR as a wildlife area, was managing cat-tail, removing it in small patches to create areas of open water for waterfowl. The native cat-tail, *Typha latifolia,* was found on only one site in Pierce County, but not in the wetland restoration area.

Reed canary grass (*Phalaris arundinacea*) dominated all communities classified as fresh (wet) meadow – disturbed subtype, a minimum of 162 acres. Not counted in this total are communities in which it dominated the ground layer of shrub-carrs, alder thickets, and forested communities. Only one site was managing the species by burning the area every few years, but the community still only met the VERY POOR condition benchmark for wet mesic prairie.

Combined, these two species, hybrid cat-tail and reed canary grass, are estimated to cover approximately 45% of the 1008 acres of mapped wetland from these sites.

Sandbar willow (*Salix interior*) was a dominant in all POOR and VERY POOR condition shrub-carrs. It was frequently associated with the edges of scrapes and other excavated areas. This species is only rarely associated with natural shrub-carrs, even disturbed shrub-carrs, according to data from WDNRs FQA survey database. It likely colonizes freshly exposed mineral soil associated with earthmoving activities during restoration. In natural areas it is found almost exclusively in the disturbance zone of rivers, on sandbars or other newly exposed mineral soils (Ryan O'Connor, WDNR pers. comm).

Fate of seed mixes 13-26 years later

We found that in most cases plants which dominated the site 13-26 years post-restoration were not part of the original plantings nor were they species typically found in seed mixes (Table 7). The exceptions are two prairie species: *Andropogon gerardii* and *Silphium perfoliatum*; three wet meadow species: *Calamagrostis canadensis, Scirpus atrovirens* and *Scirpus cyperinus*; and the trees *Larix laricina* (in a former cranberry bed) and *Quercus bicolor* (on a floodplain). Other seed mix species that were found to be persistent on several sites at low cover (1-5%) were *Eutrochium maculatum, Eupatorium perfoliatum, Asclepias incarnata, Verbena hastata, Zizia aurea, Rudbeckia subtomentosa, Vernonia fasciculata, and Hypericum ascyron.*

There were seven examples in which seeded species spread and came to dominate FAIR-condition restorations, however, GOOD and EXCELLENT-condition restorations were not dominated by plants that were in seed mixes or plantings. Instead they were likely dispersed from connected or adjacent wetlands in the area. The same influence of adjacent vegetation is true for restorations with a POOR and VERY POOR outcome, which were commonly colonized by hybrid cat-tail or reed canary grass.

Overall, seeded species were only rarely important after the first 10 years on these sites, which lacked long-term plant management. However, the existence of even a few instances where seeded species spread and created a stable matrix or increased site diversity in the long term makes seed mixes a worthwhile investment.

6.3 Soil Texture and Carbon Results

Soil texture, organic matter, and soil carbon

Soil classification of 68 soil samples from the 15 mitigation sites found 47 (69%) of collected soils to be mineral with silt or clay predominant, 13 (19%) mucky mineral, 7 (10%) were sandy and one sample (1%) was organic. Average percent soil organic matter (SOM) for each soil texture type is shown in Table 9. Mean soil organic carbon across all 68 samples was 4.9%. Soil total carbon (STC), which includes inorganic forms of carbon as well as organic, from 68 soil samples taken from surface soils (<10cm) ranged from 0.4 to 17.7% dry weight, with an average of 4.9%. Other soil chemistry results not shown include soil organic matter (SOM), with average amounts of 7.0%. Total nitrogen (not shown) had a close relationship with SOC in our dataset (R^2 = 91%). Phosphorus (not shown) did not have as tight a relationship (R^2 = 30%) but generally increased with SOC.

Table 9. Results of in-field texturing and soil sample analysis from 68 soil samples shown with mean soil organic matter (SOM), organic carbon (SOC), and soil total carbon (STC) which includes inorganic and organic forms of carbon.

Soil Type	n	SOC %	SOC Range	STC %	STC Range
Silty/Clayey	47	3.7	0.4 - 8.6	3.8	0.4 – 9.5
Sandy	7	2.6	0.7 – 5.5	2.7	0.7 – 5.1
Mucky Mineral	13	9.5	5.0 – 17.7	9.6	4.6 – 17.7
Organic	1	16.1	-	16.3	-

Soil carbon from mitigation sites compared to other wetland datasets

Natural wetlands datasets included only STC in common, so for comparison purposes STC was used to discuss soil organic carbon. STC amounts were nearly identical to soil organic carbon (SOC) amounts alone, (Table 9) with a mean percent dry weight of 5.0%. Natural wetland soils (NWCA 2011-12, WDNR FQA surveys) had mean STC ranging from 23.3% to 38.5% (Figure 9). Soil carbon amounts from mitigation sites were significantly lower than natural wetlands regardless of community type (Figure 10). STC amounts from restored wetlands that were not part of a mitigation project but implemented by other government or non-profit organizations to improve wildlife habitat or general ecological functioning had slightly higher amounts (mean = 8.2; p-value = 0.04).

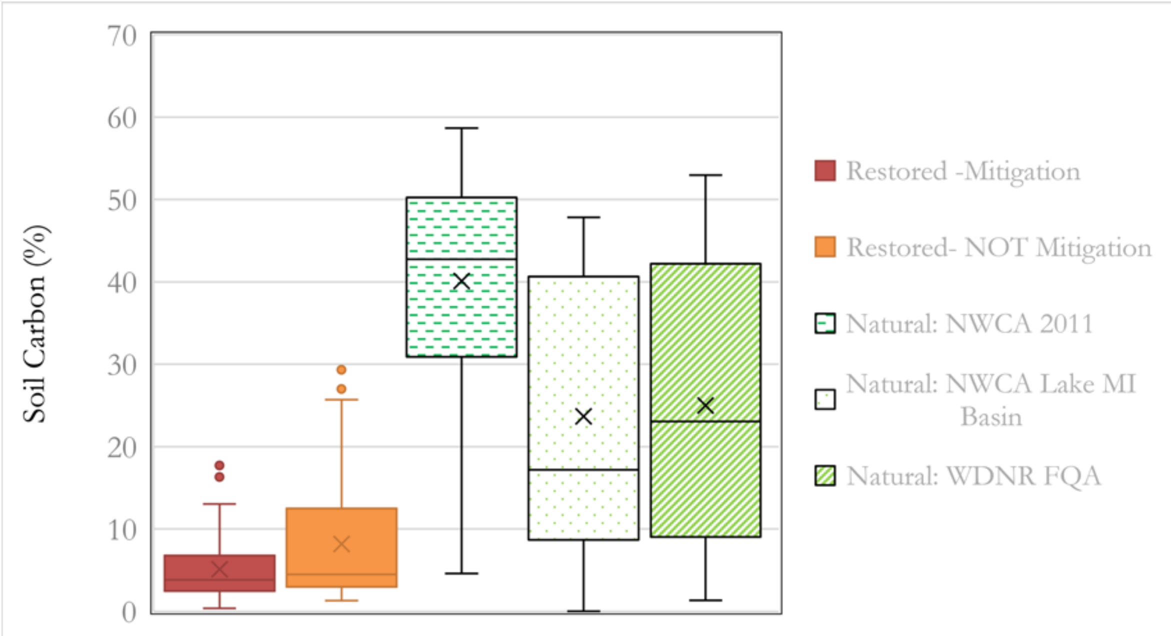

Figure 9. Soil total carbon from restored wetlands: mitigation sites (n = 69), other restoration projects (n = 29); and natural wetlands: NWCA 2011 (n = 20), NWCA Lake Michigan Basin (n = 46), WDNR FQA (n = 337).

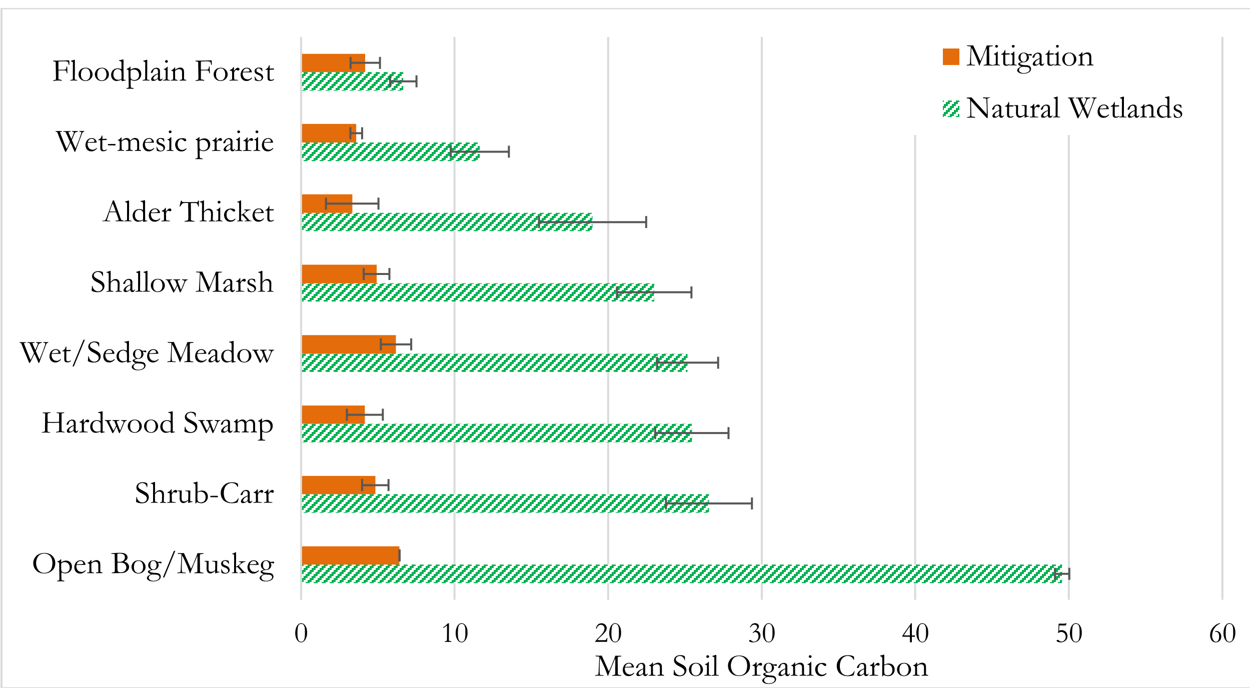

Figure 10. Mean (+/- standard error) soil organic carbon by community type from 77 mitigation sites ≥ 10 years old (solid orange) and 277 natural wetlands from WDNRs wetland FQA data (striped green). Restored open bog/muskeg bar represents only one sample, a restored open bog from a former cranberry operation.

Soil total carbon (STC) amounts from the 15 mitigation sites were low, ranging from 0.4 to 17.8 percent dry weight. In contrast, soil total carbon in natural wetlands ranged from 0.5 to 58.7% (Figure 9). The lowest amounts of STC (< 2% dry weight) were found in excavations or scrapes 20 to 26 years old from central and northern Wisconsin

6.4 Trends with Age

Using regression analysis for continuous variables and ANOVA or t-tests for categories found no significant trends in condition outcomes (p-value = 0.76, wC *(*R^2 = 0.0004) or SOC *(*R^2 =0.01) with age among mitigation sites, which ranged in age from 13 to 26 years post-construction.

Table 10. Number of samples, % Fair+, mean wC and mean soil organic carbon (% dry weight) for vegetation and soil samples from 15 mitigation sites divided into two age groups.

Age Group	n	% Fair+	wC	SOC
<20	36	47%	3.1	5.5
>20	48	44%	2.8	4.7

While our results found no evidence of a general trend toward increasing recovery of vegetation and soils with time, there was a slight tendency for older sites to have lower levels of vegetation and soil functioning than younger sites (Table 10), though this was statistically insignificant.

6.5 Records of Impacted and Released Wetlands

Impacts

Impact information from RIBITS associated with the 15 selected sites was restricted to quantity (acres) and type and was available for all but 3 sites (Lang Rd. and Princes Point sites had no information; Veolia has had no debits thus far). Type information was provided for only one third of the remaining reported impact acreages and varied in specificity with most indicated only as structural type (emergent, shrub, forested) and a few given as a community type, (e.g. fresh wet meadow, hardwood swamp, sedge meadow). Of the 12 sites with impact acreage listed, 394 acres had no information about wetland type.

Impact acreage compared to compensation acreage

The total acreage of wetland impact associated with the 12 sites with records was 582.7. The total amount of developed wetland we mapped from the same 12 sites, not including upland, preservation, and areas outside the wetland development area, was 932 acres. The overall ratio of impact acreage to developed wetland acreage, based on this sample of 12 sites is 1.6 acres of compensation for every one acre of loss. However, 7 sites are still open to credit sales (Table 3) with a total of 105.4 credits still available to mitigate new impacts, comprising 10.5% of the total released credits from the 12 sites. Therefore, the final ratio, only available once all sites are closed will be lower than our estimate of 1.6 acres of compensation for every acre of impact, perhaps closer to 1.4: 1, assuming the status quo.

This may be an underestimate of the compensation ratio considering we may have underestimated compensation acreage, and given than we did not include upland buffer, or preservation, which are typically given some compensation credit. However, this estimate does fall within the range of stated compensation ratios from the projects in RIBITS (where available) of 1.2 to 1.7 acres of compensation for every acre of fill for permanent impacts, 0.5 for conversions of wetland type, and 0.25 to 0.35 for temporary impacts.

Impact type compared to compensation type

From the limited information available we found that forested wetlands held the greatest share of the impacts (Figure 11). Reported impacts to forested wetlands totaled 112 acres, or 60% of the total of losses with a reported type, while aquatic or shallow, open water impacts totaled 0.3 acres.

Comparing impact type with the results of surveys in 2018 -2019 (Figure 12) suggests a loss of forested wetlands in exchange for gains in open water and emergent wetlands for the selected sites.

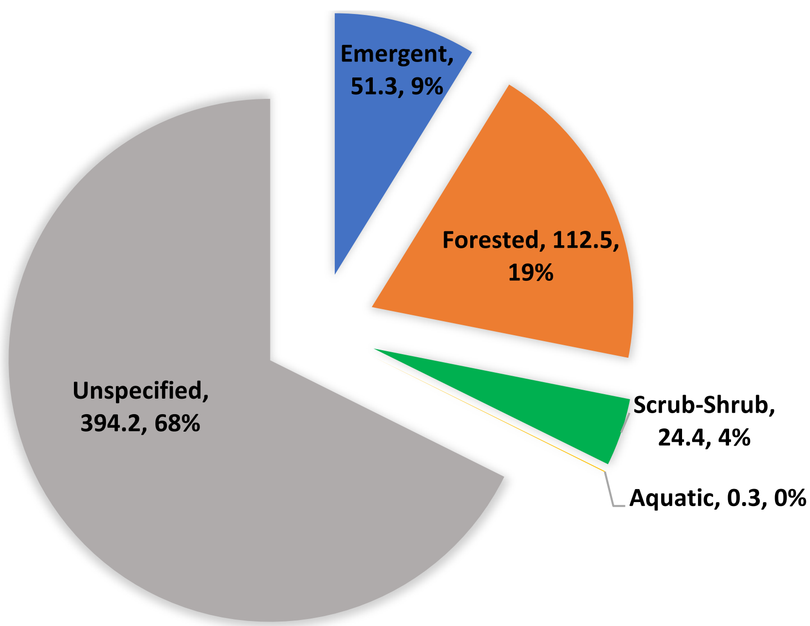

Figure 11. Wetland impact type, acres, and % of a total of 582.7 acres debited from 12 mitigation sites from RIBITS (2020). Three sites selected for the study had no impact information. Many sites are still open and will add to this total in the future.

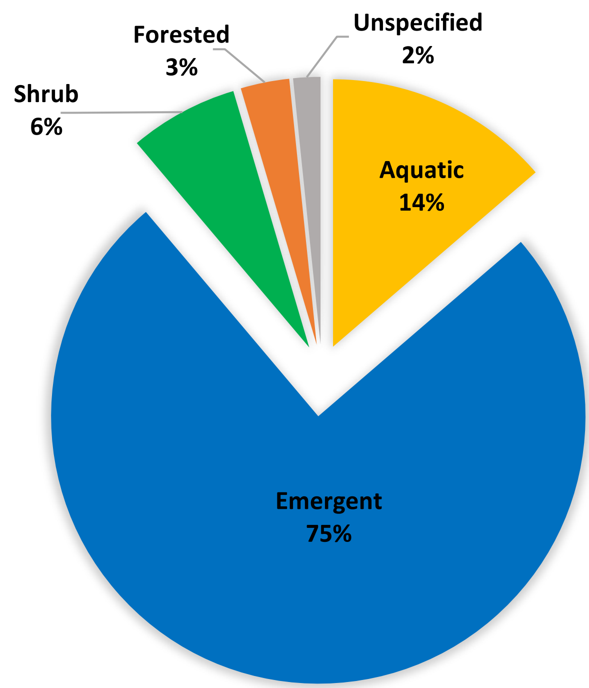

Figure 12. Wetland compensation type and % of a total 1008 acres surveyed and mapped from 15 mitigation sites in 2018-2019.

Releases

Records from RIBITS were incomplete when it came to recording wetland types developed and released, with 664 acres (60%) assigned no specific wetland type (Figure 5; Figure 13). Nine of the 15 sites reported released wetland by Eggers & Reed community types (Figure 13). Overall acreage surveyed and mapped from these sites was 17.4 acres less than released acreage probably due to an inability to assign a wetland type to 100% of wetland area during mapping and a tendency to underestimate rather than overestimate community boundaries when mapping. Despite this, many community types were found to have greater acreage when surveyed than was reported as released. For instance, wet to wet-mesic prairie, shrub wetlands, and shallow marsh appear to have developed or expanded in the years after release. On the other hand, it appears that sedge meadow and fresh wet meadow were lost in the years after release. Forested wetland and open water areas, although they expanded somewhat, were within 14 acres of released amounts.

Sedge meadow discrepancies were particularly high. Four sites reported sedge meadow releases totaling 126.3 acres: Wildcat Mountain (18 ac), Branca (23.1 ac), Pechacek-Gilbertson (56.8 ac.) and Crawford Creek (28.4 ac.). During our surveys we found 0.3, 1.8, 11.8, and 3.6 acres of sedge meadow on these sites respectively, a total of 36.5 acres.

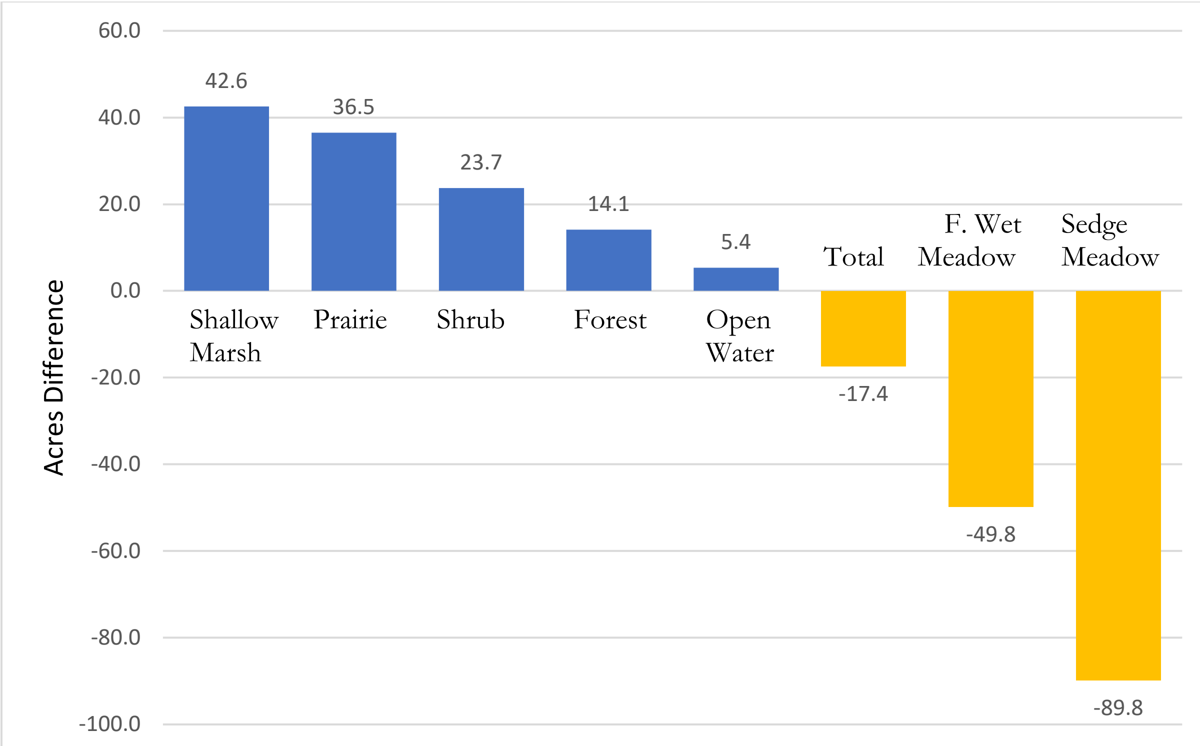

Figure 13. Differences between wetland type acreage released (RIBITS 2019) vs wetland type surveyed in 2018-19 using data from 9 of the selected mitigation sites.

6.6 Discussion Part 1: Long-term Ecological Outcomes on Mitigation Sites

Recovery of wetland plant communities

Conclusions drawn about mitigation in Wisconsin from this data should be made with care. Due to the method by which sites were selected results are biased toward the outcomes of the largest sites. For instance, one 465-acre site contributed 184 acres of poor-quality *Typha* marsh to the North Central Hardwoods ecoregion. Also, the total number of sites selected was small compared to the pool of mitigation sites with a minimum age of 10 years, a number which is unknown but is likely at least 70, and may be over 200 if other types of mitigation, (e.g. permittee-responsible, project-specific) are included.

A surprising result from this study is the attainment of EXCELLENT condition, the highest condition tier, by two of the assessed wetlands from the selected mitigation sites. We know of no previous study of restoration quality that has reported such high results. This demonstrates that it is possible, at least in the northern region of Wisconsin, for restored wetland communities to recover from plowing to resemble an EXCELLENT quality emergent marsh 23 years after excavation or an EXCELLENT quality sedge meadow 16 years after impoundment construction. Both wetlands were receiving surface water inputs from high-quality natural wetlands adjacent to or upstream of the mitigation site.

Our finding of low-quality in 56% of restored wetland plant communities (72% by area) on mitigation sites 13 -26 years post-construction is typical of findings from other studies. A similar study of randomly-selected mitigation sites in Ohio (Micacchion et al. 2010) found wetland vegetation quality to be 0% “Excellent”, 19.2% “Good”, 42.3% “Fair”, and 30.8% “Poor”. However, in this 4-category system, success was defined as reaching the upper tier of “Fair” scores. They concluded that the ecological condition of mitigation wetlands is “typically inferior to that of natural wetlands in Ohio” and considered 61% of wetlands to be failures using vegetation and amphibian indices.

A meta-analysis of 621 wetland sites by Moreno-Mateos et al. (2012) found that measurements of biological structure remained 26% lower in restored wetlands in comparison to reference levels even 100 years after restoration. Vertebrates and macroinvertebrates in this study were found to recover faster than plants which took on average 30 years to plateau at below-reference levels. Our results suggest that many of the sites in our study reached a similar plateau, though earlier and far below reference levels, given that once they are invaded by reed canary grass or hybrid cat-tail they are unlikely to improve with time. While results from this study show that good and excellent quality plant communities are possible under the right circumstances as early as 15 years post-restoration, on average, plant community condition 13 – 26 years post-restoration was found to be low. For instance the average wC value across all 81 restored wetlands was 2.9 which corresponds with a POOR or VERY POOR natural wetland for most community types for which benchmarks have been calculated in WI, though for a few communities in select ecoregions it meets the criteria for the low end of FAIR.

Moreno-Mateos et al. (2012) suggests that the factors limiting recovery may be dispersal limitation, establishment limitations, or sensitivity to the altered conditions found on wetland restorations, such as low soil organic matter. He also found that recovery was slower in colder climates such as those found in all but the southernmost regions of Wisconsin.

Recovery of soil biogeochemical functioning associated with soil organic carbon

Many studies of restorations have found lower soil organic carbon levels in restored wetlands (e.g. Bishel-Machung et al 1996; Bruland & Richardson 2005; Yu et al 2017) with one meta-analysis of 41 restorations finding 29.2% lower SOC compared with natural wetlands (Xu et al 2019) and another finding that SOC reached a plateau 8 years after restoration and remained 50% lower after 20 years (Moreno-Mateos et al 2012).

Low soil organic carbon suggests that many restored wetlands may be functioning at reduced levels in comparison to the wetlands they are replacing in terms of their capacity to hold water, sequester carbon and nitrogen, remove nitrates, phosphorus and contaminants, and transform organic nutrients to plant-available forms (Mitch & Gosselink 2015). Other studies of wetland restorations have also found that the soil bacteria community composition differed from that found in reference wetlands and were associated with lower denitrification potential (Peralta et al. 2010).

Non-mitigation wetland restorations (from the restoration techniques study) had slightly higher SOC than mitigation sites, ranging from 1.3 to 29.4%. This may be driven by the way in which mitigation activities are measured and credited- in acreage gains, with the highest credit given for wetlands developed from fully-drained former wetlands (re-establishment). Other wetland restoration practitioners are not as concerned with acreage gains so are more likely to restore areas that are only partially, rather than fully-drained. For example, filling or plugging a ditch cut through a wetland or reconnecting wetlands with river or stream floodplains. These partially-drained wetlands are likely to have higher SOC due to both more anoxic conditions and fewer years of plowing than the fully-drained wetlands on the same field.

SOC stocks should begin to rebuild as soon as hydrology is restored and vegetation becomes established. One study of an emergent marsh created from upland soils in Ohio found that organic matter began to accumulate almost immediately and continued at an average rate of 1% every 3 years (Mitch & Gosselink 2015). At that rate restorations in our study that were over 20 years old should have at least 6.7% organic matter. However, organic matter was less than 6.7% in 73% of restorations despite only 3 of them beginning on upland soils. While we expect that soil organic carbon levels increased post-restoration at least on some sites in our study, we could not detect an increase in soil carbon with site age. Instead, there was a slight decrease in soil carbon in the older sites probably due to the more prevalent use of excavation. Mean soil carbon levels were lowest in restorations that used excavations, scrapes, or impoundments (mean SOC = 3.8%) compared to those that removed alterations either partially (mean = 6.8%) or completely (mean = 7.7%, ANOVA p-value = 0.01). The highest amounts of SOC (10 -18% dry weight) were associated with ditch plugs in a groundwater fed valley and a dike removal in a cranberry farm restoration. Another was along the edge of a scrape in Houghton muck soils. Two of these sites had organic or muck soils prior to restoration. Overall, it seems that the SOC levels restorations started with are still the main driver of the SOC levels 13 to 26 years later. Future studies need to compare SOC amounts from the same site over time to understand which sites are building organic carbon stocks and therefore recovering, and which sites may be impaired in their ability to recover.

Results from this study align with published research that suggest the recovery of biogeochemical functioning in restorations is either a very slow process which could take as many as 500 years (Hossler & Bouchard 2010; Yu et al. 2017), or that many restorations are moving instead toward a lower-functioning alternative state (Moreno-Mateos et al. 2012).

6.7 Conclusions: Are restored wetlands replacing lost functional values 13 -26 years later?

We conclude that mitigation has been successful in replacing natural wetlands in POOR or VERY POOR condition, with 56% of individual restored wetlands found to be in this condition. Community types successfully and reliably replaced include cat-tail marshes, reed canary grass-dominated fresh (wet) meadow and shallow open water areas. However, if any of these lost wetlands had soil organic matter greater than 6.6%, (75% of restored wetlands in our study fell below this amount) there is little indication that the carbon loss and associated biogeochemical functioning have been replaced. Since mineral soils can range from 0 to 12% SOC, this indicates that even some mineral soil wetlands have carbon amounts that are not likely to be replaced by mitigation in the short time. There is no indication from our data that any organic soil can be replaced in a reasonable time frame using current methods. Even the restored cranberry bogs in our study had significantly lower organic matter than natural bogs, muskegs, and black spruce swamps (Figure 10), and the site with the most highly impacted soils, the deepest excavation in our study, still had only 0.7% soil carbon after 26 years, despite fully-saturated soils.

Medium-quality wetlands (i.e. FAIR-condition) were less reliably replaced, with only 19% achieving this quality. These include wet to wet-mesic prairies, fresh (wet) meadow- native subtype, sedge meadows, and shrub-carrs. While this study shows that wetland plant communities in GOOD and EXCELLENT condition are possible outcomes after 15 years, they are nevertheless very rare: only 9% of total statewide wetland development acres were high-quality and most were in the northern half of the state or began on sites with remnant wetland functioning.

We also found that forested wetlands were not being replaced on the sites we surveyed. Across all 1008 acres of developed wetland we found only 15 acres had been planned as a forested wetland restoration, and only 15 acres of forested wetland developed naturally. While we do not know the total acreage of forested wetland losses, we know at least 112.5 acres were debited from the selected sites (Figure 11). The diversity, habitat, and carbon storage associated with forested wetlands impacts may be the most significant plant-based function not being replaced via mitigation.

While these losses of function associated with wetland quality and type have been compensated for to some extent by restoring more wetland acreage, with approximately 1.4 to 1.6 acres of compensation wetland developed for every acre of loss on average, the sufficiency of this trade has not been evaluated. Also, the record-keeping that would be required to evaluate this comparison does not exist yet. In a review of the effectiveness of mitigation in conserving wetlands, Burgin (2010) concludes that "unfortunately the 'no net loss' of wetland habitat and associated species generally remains a concept. Scientific and management issues associated with environmental loss are yet to be overcome…" In the meantime, while replacement of jurisdictional wetland losses in terms of acreage appears to be successful, other results from this study, and other studies from the literature suggest that mitigation may be resulting in an overall net loss of wetland functions and services (Hossler & Bouchard 2010; Moreno-Mateos 2012).

7 Results Part II: Factors Influencing Restoration Outcomes

7.1 Part II Results

The results of testing 9 factors for differences in condition (Fair+/Poor-) and floristic quality (wC) are shown in Table 11. The 10th factor we tested, age, is shown separately (Table 12). Data used for this analysis come from two datasets: the 15 selected mitigation sites discussed in Part 1, and data from the "Restoration Technique Study" (Gibson et al 2019) for a combined total of 163 AA's from 50 wetland restoration projects. The age range from the combined datasets is 4 to 26 years.

Significant factors affecting wetland condition outcome (Fair +/ Poor -)

Ecoregion, general community type, and summer saturation level showed significant ($p \leq 0.05$) variation in condition outcomes. Variation in soil types, hydrologic restoration techniques, soil organic carbon, initial conditions, and soil pH had marginally non-significant differences in outcome ($p = 0.06$ to 0.13). Differences between maintained and un-maintained groups showed no significant differences ($p > 0.4$) nor were there differences in condition outcomes between age groups, despite a slight decline in FAIR or better outcomes from 58% in the youngest group to 45% in the oldest group. Of the significant and marginally significant variables, the factors with the strongest positive effect on the proportion of fair or greater outcomes were SUMMER SATURATION LEVEL = AT SURFACE (+24%); TOTAL ORGANIC CARBON = >10% DRY WEIGHT (+23%); SOIL TYPE = MUCKY MINERAL (+20%); and COMMUNITY = PRAIRIE (+19%). The factors with the strongest negative effect on condition outcome were SUMMER SATURATION LEVEL = ABOVE SURFACE (-27%); COMMUNITY = MARSH (-20%) and ECOREGION = SOUTHEASTERN WI TILL PLAINS (-19%).

Significant factors affecting floristic quality (wC)

Using wC to explore the effects of the 10 factors resulted in similar results as condition outcome. However, differences between groups were more likely to be statistically significant. Like the results using success rate, ecoregion, summer saturation level, and community type were highly significant. But also significant were approach (initial conditions), soil type, and soil pH, soil organic carbon, and restoration technique. Factors with particularly high positive effects on wC were ECOREGION = NORTHERN LAKES AND FORESTS (+1.7), APPROACH = REHABILITATION (+0.9), SUMMER SATURATION LEVEL = AT SURFACE (+0.9); and SOIL PH = STRONGLY ACID (+0.8). Factors with the strongest negative effect on wC were COMMUNITY = MARSH (-0.9), SOIL PH = ALKALINE (-0.7), COMMUNITY = FOREST (-0.6); and SUMMER SATURATION LEVEL = <18" BELOW SURFACE (-0.6).

Table 11. Results of significance testing of 9 factors on the proportion of Fair or better condition outcomes (Fair+) and mean $w\overline{C}$, using proportion tests and ANOVA. Effect size is the group's difference from the mean value of the dataset (50% Fair+ and $w\overline{C}$ = 3.1). Factors are listed in decreasing order of significance. P-values ≤ 0.05 are denoted with a '*'.

Factor	Groups	n	%Fair+	Effect Size	*p-value*	$w\overline{C}$ mean	Effect Size	*p-value*
Ecoregion	Southeastern WI Till Plains	55	31%	-19%	*0.00**	2.6	-0.5	*0.00**
	Driftless Area	29	62%	12%		2.8	-0.3	
	North Central Hardwood Forests	45	60%	10%		3.2	0.1	
	Northern Lakes and Forests	20	65%	15%		4.8	1.7	
Community	Forest	12	33%	-16%	*0.01**	2.4	-0.6	*0.01**
	Shrub	21	33%	-16%		2.9	-0.2	
	Marsh	30	30%	-20%		2.2	-0.9	
	Meadow	62	61%	12%		3.3	0.3	
	Prairie	22	68%	19%		3.1	0.0	
Summer Saturation Level	Above Surface	9	22%	-27%	*0.01**	2.9	-0.1	*0.00**
	At Surface	38	74%	24%		3.9	0.9	
	Within 18" Below Surface	12	42%	-8%		2.4	-0.6	
	> 18" Below Surface (Dry)	42	45%	-4%		2.7	-0.4	
Soil Type	Mineral- Silts or Clays	75	39%	-10%	*0.06*	2.7	-0.4	*0.02**
	Mineral -Sands	13	54%	4%		3.4	0.3	
	Mucky Mineral	25	70%	20%		3.7	0.6	
	Organic	8	63%	13%		3.7	0.6	
Hydrologic Restoration Technique	Excavation/Impoundment	45	38%	-12%	*0.07*	2.9	-0.2	*0.03**
	Partial alteration removal	39	54%	4%		2.8	-0.3	
	Complete alteration removal	47	62%	12%		3.6	0.5	
Total Organic Carbon	<10% dry wt.	95	45%	-4%	*0.07*	3.0	-0.1	*0.05**
	>10% dry wt.	18	72%	23%		3.7	0.6	
Approach	Reestablishment	106	47%	-3%	*0.11*	2.9	-0.2	*0.00**
	Rehabilitation	34	65%	15%		4.0	0.9	
Soil pH	Alkaline	29	31%	-19%	*0.13*	2.4	-0.7	*0.01**
	Neutral	35	54%	5%		3.0	-0.1	
	Slightly Acid	30	53%	4%		3.2	0.2	
	Strongly Acid	23	61%	11%		3.9	0.8	
Active Maintenance	No	114	47%	-2%	*0.42*	2.9	-0.1	*0.05**
	Yes	37	57%	7%		3.5	0.4	

Effect of site age on condition, floristic quality, soil carbon, and maintenance rates

No significant differences were found between four age groups, (Table 12) in percent FAIR+ condition outcome, mean wC, or mean soil organic carbon (SOC) despite declines with age in each of these variables. However, the decline in maintenance rates with age was significant. Similarly, regressing site age against or SOC showed only very weak declines with age.

Table 12. Results by age group. Condition results as %FAIR+, mean wC, mean soil organic carbon (SOC) % dry weight and percent of AAs with active maintenance by age since construction. N.S. = not significant (p-value ≥ 0.10). Significant values denoted with '*'.

Age Group	n	% FAIR+	$\overline{wC}$	SOC % dry wt.	% With Maintenance
<10	24	58%	3.3	8.3	50%
10 to 14	46	50%	3.1	7.6	26%
15 to 19	32	44%	3.2	5.5	22%
20 to 26	55	42%	2.9	5.1	16%
p-value (ANOVA)		*0.6 (N.S.)*	*0.6 (N.S.)*	*0.1 (N.S.)*	*0.02**
Regression R^2			*1%*	*5%*	

7.2 Discussion of 10 Factors Influencing Restoration Outcomes

We did not measure all factors affecting restoration outcomes, and data for many of the factors we did measure were gathered opportunistically resulting in uneven sample sizes and incomplete data. Therefore, any results discussed must be considered exploratory in nature. Nevertheless, the patterns found within these 163 restored wetland communities from 50 projects show some significant trends that may warrant further consideration.

Level III ecoregion

One of the stronger patterns in this study was the low proportion of successful outcomes (31%) in the Southeastern WI Till Plains. By contrast, restorations in the Northern Lakes and Forests resulted in 65% FAIR or better outcomes. Southeastern WI Till Plains is the most populated and developed region of the state and ambient water quality is lower here than in other regions. This highlights many important factors that contribute to restoration outcomes and wetland health in general that were not measured, including stressors such as nutrient loading, sedimentation, intensity of historical and current agriculture, wetland losses in the watershed, and other impacts to water quality and wetlands in the region. The suitability of soils for agriculture in a region plays a large role in determining both the legacy of historical agricultural practices on the soils of a prospective restoration site as well as the agricultural intensity in the surrounding area that will bring ongoing stressors to restored wetlands. Related to this, the presence or absence of high quality wetlands and abundance of invasive species in an area may play a significant role in determining which species come to dominate a site, given the conditions are appropriate.

Soil organic carbon and soil texture

In our data there seemed to be a drop-off in successful plant community outcomes when SOC was less than 10%. Mucky mineral soils, with amounts of organic matter in-between that of mineral or organic soils also appeared to do better than the other soil textures. It is not surprising that very low SOC levels have an adverse effect on vegetation success, given the central role organic matter plays in soil health and nutrient cycling. It has also been suggested that low SOC on restored sites may affect recruitment of native species (Galatowitsch & Van der Valk, 1996), since, with the exception of floodplain forests and wet-mesic prairies which have among the lowest mean SOC in WDNRs database of natural wetlands, many wetland natives may be adapted to soils with higher organic matter. For instance, 68% of natural wetlands from DNR database had more than 10% SOC.

Mineral soils dominated by silts and clays were the most common soil type in our study and had a lower success rate than all other soil textures, including sands. It appears that silt and clay soils are associated with more tolerant plant species and were prone to hosting *Phalaris arundinacea* and *Typha* communities. A possible explanation is that other soil types, including sandy soils and organic-rich soils, may be less prone to plant-available nutrient accumulation, which may be driving the competitive advantage of these invasive species in fine-grained soils.

Soil pH

Alkaline pH soils were associated with low condition outcomes. Only 31% of alkaline soil AAs achieved a FAIR or better outcome and mean wC of plants in restorations tended to be lowest in the alkaline soils and rose as pH became more acidic. Soil pH in the "alkaline" group ranged from 7.4 to 8.0 and are more accurately described as slightly to moderately alkaline soils. There were no strongly alkaline soils in the study. This result may largely be due to a high proportion of AAs with alkaline soil occurring in the Southeastern WI Till Plains (63%), and the true cause may be the higher development pressure and lower water quality in that area, confounding the interpretation of this result.

Mid-Summer Hydrology

Hydrology observations made at the time of our site visits provided only a glimpse of the year-round hydroperiod and would be expected to be highly influenced by the month of our visit and whether or not rain had fallen recently. Despite this we found a high degree of correlation with plant community condition and floristic quality with mid-summer soil saturation levels among our study sites.

Summer saturation level showed some of the strongest effects in the study (+24% increase in the likelihood of scoring fair or better). Restorations with summer saturation level above the surface had only a 22% success rate compared to a 74% success rate for AAs with soils saturated to the surface. Other saturation levels, "within 18" below the surface" and "greater than 18" below the surface" were in-between with success rates of 42% and 45% respectively. Because July/August (when most observations were made) is typically the driest time of the year, we assume that in most cases wetlands had higher water levels earlier in the year, especially after snowmelt. Therefore, a summer saturation level at the surface indicates that soils are typically saturated year-round.

While high-quality wetland communities are possible at any of these soil saturation levels, it may be that hybrid cattail and reed canary grass are both disadvantaged by water levels that do not remain ponded year-round but nevertheless are fully-saturated. Or it may be that native species, especially native sedges, are especially suited to these circumstances, or both. For meadow communities, which includes fresh (wet) meadow and sedge meadows in our study, there is evidence from this study to support this, at least where reed canary grass is concerned: Meadows with the driest hydrology in mid-summer were more likely to support reed canary grass than meadows that were saturated to the surface, a hydrology that was associated with sedge dominance in our study.

Wetlands with soils saturated to the surface in mid-summer were associated with groundwater-fed valleys but also with a impoundments (at a distance far enough that surface water was absent during the driest part of the year), scrapes, ditch plugs, ditch fills, and sediment removal techniques.

Wetlands that with hydrology above the surface (ponding) in mid to late summer were most often associated with excavations, scrapes and impoundments or were created using ditch plugs. Resulting plant communities were dominated by hybrid cattail in most cases, although one was dominated by sandbar willow, and one giant reed grass (*Phragmites australis subsp. australis*). There were only two exceptions, one EXCELLENT and one FAIR shallow marsh communities. Both had inputs from natural wetland systems, one naturally colonized by the conservative dominants *Sparganium emersum*, and *Glyceria borealis* from wetlands upstream and the other with a hydrology influenced by an adjacent riverine system.

Community type

Shallow marshes had the lowest success rate (30% in FAIR or better condition) of all the restored communities. Eighty percent (80%) of shallow marsh AAs were dominated by hybrid cattail covering 88% of the total shallow marsh acreage. The exceptions to dominance by hybrid cattail include marshes with strongly acid water and/or surface water inputs from higher quality wetlands in northern WI, a site receiving intensive treatments to remove hybrid cattail, and another in a groundwater-fed valley in the Driftless Area with saturated rather than ponded soils in late summer.

Forested wetlands also had a low success rate (33%) among our two studies. However, of the 9 forest restorations, only 3 were planted. Forests that succeeded naturally to floodplain forest or hardwood swamp were dominated by box elder or silver maple and often were dominated by reed canary grass in the understory resulting in a VERY POOR condition. The planted forests did better. A silver maple and river birch planting in a groundwater fed valley in the Driftless Area was in GOOD condition 10 years post-planting; Swamp white oak plantings in a floodplain resulted in a FAIR condition floodplain forest 25 years later; And black spruce and tamarack planted in a former cranberry farm reached POOR condition for a black spruce/ tamarack swamp, but it was only 7 years old at the time of the survey.

No intentionally-planted shrub-carrs were encountered in the two studies, however, 19 developed naturally. Of these 14 were sandbar willow-dominated, with reed canary grass a frequent dominant of the understory and most were in POOR or VERY POOR condition. Only those dominated by meadow willow (*Salix petiolaris*), pussy willow (*Salix discolor*), or diamond willow (*Salix eriocephala)* were in FAIR or better condition.

Alder thickets also were not intentionally planted but developed on two sites. Although they were restored in nearly the same year and both occurred in the Northern Lakes and Forests ecoregion, their restoration techniques were quite different as were their outcomes 25-26 years later. One grew on the periphery of a site that used heavy machinery to excavate and grade the area and the result was a VERY POOR alder thicket with a reed canary grass understory. The other used the rare wetland restoration technique of roadbed removal to uncover a long narrow wetland. The former roadbed area was adjacent to high quality wetlands and developed into a GOOD-condition alder thicket. Ground-layer dominants were tussock sedge and lake sedge.

Prairie wetlands had the highest success rate of the community types (68%) but meadows followed close behind with 61% in FAIR or better condition. Successful prairies were often dominated by prairie cordgrass, or big bluestem while POOR-quality prairies were frequently dominated by Canada goldenrod and Kentucky bluegrass. Wet or wet-mesic prairies may the community type that benefits the most from seeding and plantings because remnants are now rare. Canada goldenrod can be relied upon to seed itself to a site, but other native species require an initial planting. Examples of species that required planting but were able to persist on the sites were surveyed include big blue stem, prairie cordgrass, Canada bluejoint, cup-plant, switch grass, golden alexander, glade mallow, and sawtooth sunflower. The success of prairie wetlands may be due to their adaptation to mineral soils with lower SOC levels and drier hydrology, both common on typical restoration sites that begin on drained and plowed agricultural fields.

Despite many meadows being dominated by reed canary grass which often took up large areas of restoration sites, meadows still on average fared better than shallow marshes because a larger percentage of them had the right conditions to host native sedges, even if these areas were small. Approximately one third of meadows we surveyed were sedge meadows dominated by *Carex* species. Areas that hosted sedges tended to be wetter than reed canary grass-dominated meadows and frequently saturated to the surface in mid-summer. The species that dominated were commonly *Carex lacustris*, *Carex stricta*, or *Carex trichocarpa*. Rarer sedges were *Carex atherodes* and *Carex lasiocarpa* in Douglas County in the far north, while *Carex haydenii*, and *Carex pellita* appeared on one site with sandy soils. None of these sedges appear to have been planted, rather they seem to have appeared where the hydrology was suitable, often some distance from an impoundment, in valleys with high groundwater gradients, or in or near ditches.

Pre-construction drainage conditions

Restorations which started from partially-drained wetlands were more likely to result in a successful outcome (65% in FAIR or better condition) than those starting from fully-drained wetlands (47%). Only 4 AAs were thought to have started from upland soils and therefore counted as creation. These were all POOR or VERY POOR, except for the open water submergent community, which was not assigned a condition class, but appeared to be doing well floristically, with a wC of 5.8 and no invasive species presence.

Starting restoration from a partially-drained wetland condition had a higher effect on floristic quality than on condition outcome, likely reflecting the ability to restore much more conservative plant communities with partially-drained conditions than would be possible from fully-drained conditions. From our study, examples were open bog and black spruce/tamarack swamp from a cranberry operation which only achieved POOR condition. Restoring communities such as these may not achieve high condition outcomes, because they are naturally more conservative, but they are important communities to restore. Rehabilitation may be the only way to restore these community types.

Hydrologic restoration technique

With this study we were able again to compare outcomes between different restoration techniques grouped into three categories: surface modifications (impoundments and excavations); partial alteration removal (ditch plugs, tile breaks, partial dike removal); and complete alteration removal (ditch fills, tile removal, dike removal, sediment removal). Significant differences were found between these three groups with complete alteration removal having a 62% success rate and surface modifications only a 38% success rate. These differences can be tied back to the community types resulting from these techniques, with more hybrid cattail and sandbar willow associated with impoundments and excavations. In contrast, more prairies and sedge meadows were associated with complete alteration removal as well as the only open bog and black spruce/tamarack swamp restorations.

Soil organic matter may be another factor reducing successful restoration outcomes in the impoundment/excavation group. Mean soil carbon levels were lowest in restorations that used excavations, scrapes, or impoundments (mean SOC = 3.8%) compared to those that removed alterations either partially (mean = 6.8%) or completely (mean = 7.7%, ANOVA p-value = 0.01).

Maintenance

Maintenance had a surprisingly low effect size in our dataset, only improving success rate by 7% in comparison to no maintenance which lowered success rate by only 2%. This low effect may be partly attributed to the imbalance in sample sizes- with only 25% of AAs across the two studies reporting maintenance activities and the presence of many wetlands that did not appear to require maintenance to succeed. In addition, the maintenance activities reported often did not affect the condition outcome because they did not target the problem species. For instance, spraying *Phragmites* but leaving hybrid cat-tail, or spraying reed canary grass but leaving sandbar willow. In other cases, invasive species treatments were reported but invasive species cover remained high on the site. Despite these results, for those communities that do need maintenance, finding effective methods to reduce invasive species has the potential to have a large effect.

The following are examples of effective maintenance from the restoration projects included in the restoration techniques study by non-profit groups or mitigation sites still within their monitoring period:

1. Regular prescribed burns for prairie wetlands plus spot herbicide application.
2. Mowing cat-tail and removing clippings from the area plus seeding.
3. Routine spraying of hybrid cattail and adding new plantings.
4. Removing shrub encroachment by sandbar willow.
5. Adjusting hydrology: correcting weak spots in a ditch fill to prevent pooling water and cat-tail invasion.
6. Capturing and diverting nutrient-rich surface water inputs to a settling pond before it enters the wetland.

Age

The overall lack of significant trends with site age in this study is not surprising given that individual wetlands spanned the full range of condition outcomes, often on a single site. Also, there are competing factors at play in determining whether condition or SOC increases with site age. While we might expect individual sites to increase in condition over time due to succession, the process is obscured by invasive species spread and a reduction in maintenance rates with age. In addition, increases in both condition and SOC with time are obscured by improvements to restoration practices over the years, with less removal of organic upper soil horizons, and more natural hydrology and wetland communities being targeted.

7.3 Summary Of Nine Factors Affecting Success

Site Factors: Ecoregion, Soils, Drainage

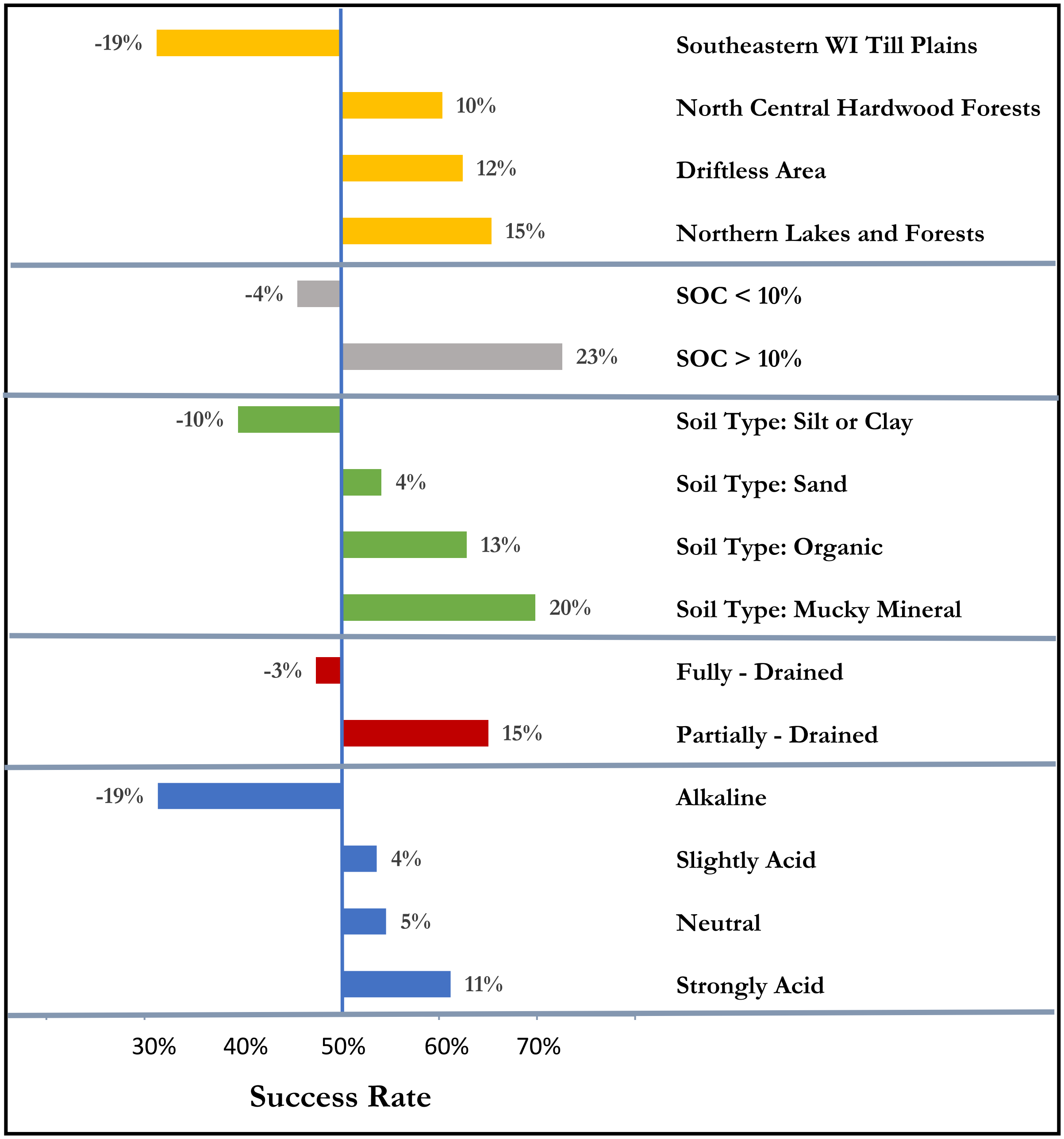

Figure 14. Effect size (difference from the mean of all samples) of 4 restoration site factors: Soil pH (4 levels, yellow); Soil Type (3 levels, brown); Initial Conditions (2 levels, blue); and Soil Organic Carbon (2 levels, red) on the proportion of FAIR+ condition outcomes.

The following site conditions were associated with a higher probability of resulting in a successful plant community (Figure 14).

1. Sites in the Northern Lakes and Forests ecoregion, or, more importantly, sites from less disturbed watersheds adjacent to high-functioning wetlands, or in groundwater-dominated areas.
2. Sites with areas that are partially-drained; i.e. some remnant wetland functioning exists.
3. Soils with higher organic carbon amounts (mineral soil with high organic material, mucky mineral, or organic soil.
4. Acidic rather than slightly or moderately alkaline soils.

Restoration Factors: Technique, Hydrology, Community, Maintenance.

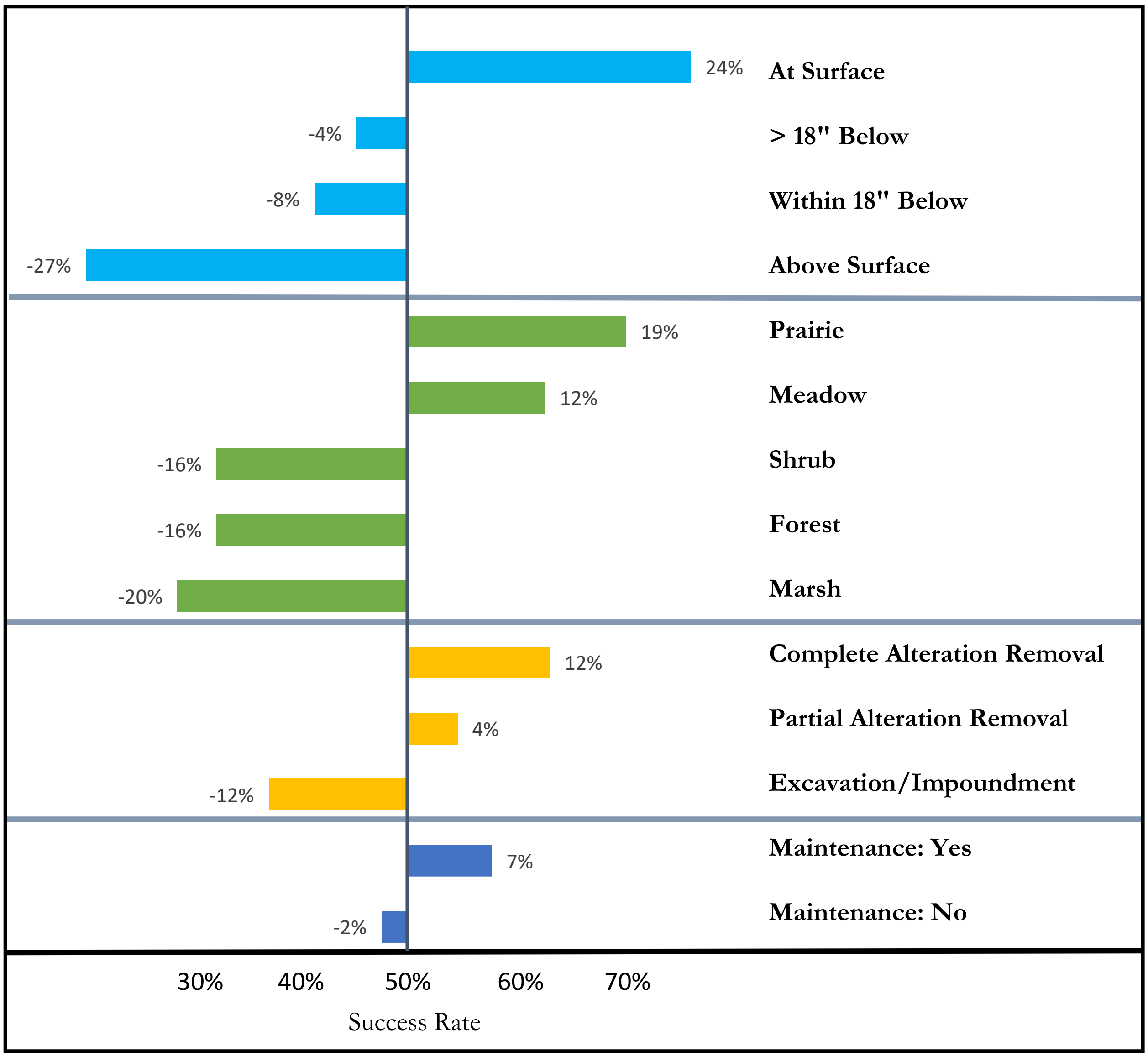

Figure 15. Comparison of restoration and management factors. Effect size (difference from the overall mean) of 4 restoration factors: Hydrologic restoration technique, Community type, July-Aug saturation level, and Maintenance on the proportion of FAIR or better condition outcomes.

The following restoration factors were associated with a higher probability of resulting in a successful plant community (Figure 15).

1. Fully-saturated soils without year-round ponding. (effective for shallow marsh, sedge meadows, wet prairies, shrub-carr, alder thicket, and open bog in this study).
2. Wet to wet-mesic prairie, sedge meadow, or fresh (wet) meadow native subtype communities, rather than marsh.
3. Planted rather than volunteer forested and shrub communities.
4. Complete alteration removal techniques, rather than introducing structures to impound.
5. Preventing the introduction of invasive species by addressing root causes of invasions or effectively managing invasive species once they are present.

8 Recommendations

8.1 Recommendations for Improving Wetland Restoration Outcomes

Based on the insights gained from examining wetland quality in restored and natural wetlands there are multiple tactics and modifications we suggest to improve long-term restoration outcomes including: what site attributes to consider when selecting a mitigation location, suggestions for design and construction, maintenance and management techniques, and post-construction considerations.

Site selection recommendations

Wetland mitigation success is often attributed to proper planning and thorough follow-up (e.g. maintenance). In our research, we have identified that there are some important factors to consider during the planning phase to improve successful mitigation outcomes and set a wetland up for long-term success. It is paramount to identify a wetland mitigation site that is not only conducive to restoration efforts but is appropriate for the specific wetland community or specific wetland functions planned for restoration.

Soil texture and organic matter content

Mitigation sites sited on mineral soils often resulted in less successful restoration outcomes. Therefore, we suggest identifying restoration sites with some residual organic matter. These soil types are more likely to be found in fields that have not been plowed frequently. Choosing a site with higher soil organic carbon is especially important if denitrification or carbon sequestration is one of the goals of the restoration. One recent study of carbon sequestration (Xu et al 2019) found that wetlands were more likely to sequester carbon when soils had been subject to less plowing prior to restoration. Often these are areas that may still delineate as wetlands despite drainage.

Sites with mineral soils may be best suited to wetland prairie communities or floodplain forests which typically have lower organic content than other wetland types (see Figure 12). Mucky mineral or organic soils are best suited to sites where a natural hydrology can be fully restored to keep soils saturated year-round. Under wet enough conditions these soils may be suitable for a wide assortment of more conservative wetland natives, including sedges.

Hydrology at the site selection stage

When multiple alterations to hydrology exist on a proposed restoration site and those alterations cannot be removed, it lowers the chances that a natural hydrological regime will be achieved. Most alterations to wetlands, whether ditches, tile, or dams have the effect of stabilizing water levels (Zedler 2000a) while it appears that many native wetland plants are adapted to water level variability. Proposed restoration sites that have ditches that cannot be disabled stand a poor chance of restoring the natural hydrologic regime that once existed and this should be taken into consideration before a site is selected.

Landscape context

It has been suggested that the replacement of wetland function is more likely and faster in landscapes that are more intact (Zedler & Callaway 1999; NRC 1992). While we did not collect data on landscape alteration, our finding that outcomes in the Northern Lakes and Forests Ecoregion, our most intact ecoregion, were much more likely to be successful (65%) than restorations in the Southeastern WI Till Plains (31%), our most human-impacted ecoregion, is strong support of this idea. Surface water inputs from higher-functioning wetlands bringing native seed sources and presumably clean water. These factors were both likely to have played a key role in the two **excellent**-quality restoration outcomes from this study. Sites from the floodplains of rivers in the Driftless Area tended to be diverse but host large reed canary grass populations, illustrating that the types and quality of wetlands upstream of the site need to be considered and planned when selecting a mitigation site.

Restoration and management recommendations

Hydrologic restoration techniques

Our data confirms that the practice of constructing surface structures to restore hydrology (excavation, impoundments) is not the most effective long-term solution for wetland restoration. This is likely because this practice does not effectively restore historic hydrologic conditions and does not utilize the site's historic soils due to soil removal.

Data from our study sites has also indicated that plant communities often are most successful over the long-term when water levels are roughly at the soil surface in mid-to-late summer. When water levels are much higher than the surface level, we saw reduced floristic quality due to cat-tail invasion. And when water levels were much lower than the surface water through the growing season, we observed more reed canary grass.

While water levels are hard to predict, selecting sites where drainage can be fully removed sets up a scenario where natural hydroperiods are more likely.

Plantings

A diverse seed mix or plantings are beneficial in the short term to get multiple species establishing in the diverse microtopography of any given site. A diverse seed mix is especially important for restorations which target prairie wetlands because naturally-occurring prairie wetlands and seed banks are not common, and these species will not seed in by themselves. Prairie wetlands also appear to be more diverse than other open wetland types by nature, which means that more seeded-in species are likely to persist and co-exist than wetter-end communities in which a few clonal perennials dominate with time.

Wetland plants can be put into three groups according to their potential lifespans: annuals, perennials, and vegetatively reproducing perennials (clonal perennials) (Van der Valk 1981). While sites are still younger than 5 years old, all three groups may exist in equal numbers on the site. This is supported by restoration studies from the literature that suggest that species richness in wetland restorations peak between 4 and 7 years post-construction (Stefanik & Mitsch 2012). Over time annuals decline and disappear in the absence of disturbance. Short-lived perennials too decline with time, providing diversity but not existing in high abundance. It is the clonal perennials that expand with time (generally after the 5-year monitoring period has expired) and create a matrix which determines the overall nature and structure of the community.

Examples of vegetatively reproducing, or clonal perennials to avoid are reed canary grass, cattail species, giant reed, and sandbar willow. Clonal perennials to encourage include blue joint grass, and many sedges including *Scirpus cyperinus*, *Carex stricta, Carex lacustris, Carex utriculata, Carex lasiocarpa, Carex pellita,* and *Carex trichocarpa.* Wet to wet-mesic prairies clonal perennials (which are not always hydrophytes) include big bluestem, prairie dropseed, prairie cordgrass and goldenrod. Overall, our research suggests that restoration sites would benefit more from identifying the 'right' dominants than focusing primarily on diversity or richness.

We recommend identifying the presence of clonal perennials early in the development of the vegetation, with the knowledge that the trajectory of the plant community will largely be determined by these few species. Given the importance of these vegetatively spreading perennials, it should also be noted that establishment of some species may be more successful by plugs rather than seed. At a minimum, success rates may be improved by including a mix of both seed and plugs for certain species. Appropriate species for many of Wisconsin's wetland community types have been identified using data from WDNRs wetland FQA surveys and should be made available on WDNRs website soon.

Maintenance

Maintenance and management activities reported from our study sites were in many cases ineffective in reducing invasive species. It is our belief that with community-specific and frequent maintenance, more favorable outcomes would be expected, especially if the underlying causes of invasions can be addressed.

Controlling invasive species is one of the most important activities that a mitigation site can commit ensure a sustainable site. Maintenance after the monitoring period is not required at this time but that may be changing with recent shifts in mitigation thinking in the St. Paul district. Already, Wisconsin's In-Lieu Fee mitigation program, The Wisconsin Wetland Conservation Trust, is requiring long-term endowments to fund post-monitoring maintenance activities. Often the compensation site plans will describe expected long-term monitoring and maintenance activities

The need for maintenance and the type of maintenance is site-dependent and may vary widely between different sites, soil conditions, hydrologic conditions, and target communities. For example, it is commonly understood that midwestern prairies require occasional burns; therefore, any mitigation site proposing prairie communities should plan for fire breaks and require a long-term maintenance plan that includes prescribed fire.

That said, restoration sites should be designed to be low-maintenance from the beginning. In our study of mitigation sites, 28% of the restored wetland area was not receiving any maintenance since the end of their monitoring period and did not require it to achieve a fair condition wetland community. Often the reason for this is a combination of several factors: the right hydrology, the existence of remnant soil functioning, and cleaner water inputs.

Recommendations To Promote A Low-Maintenance Wetland Restoration:

1. The largest impediments to the recovery of fair-condition communities were reed canary grass (*Phalaris arundinacea*) and hybrid cat-tail (*Typha X glauca*). These species, and their ecological root causes, should be considered at every phase of site development.

2. Maximize soil re-wetting, and natural fluctuations without year-round ponding. Monitor closely as the site develops and expect to practice adaptive management over many years to make corrections and improvements.

3. Sites with severe disturbance histories (e.g. intensive plowing and drainage) have the highest susceptibility to invasive species. Target sites that still have remnant function on which to build to encourage native species.

4. Avoid sites with ditches, drain tiles, impounding features, and legacy sediments unless they can be removed or disabled due to their negative effect on natural hydroperiods.

5. Address sources of nutrient and sediment inputs to the site before they reach the wetland. At least one mitigation site has had success diverting agricultural runoff to a settling pond near a road that can be accessed easily for maintenance.

6. Avoid or minimize soil disturbance and heavy machinery as much as possible, especially in mineral soils that contain most of their organic material in a thin upper horizon. Areas of soil compaction, spoil piles, over-filled ditches, and removal of topsoil will not recover for many decades, if not longer.

7. Restorations adjacent to higher-functioning wetlands will increase the opportunities for appropriate species to colonize, decrease nutrient inputs, and minimize long-term maintenance.

8. Plant trees and shrubs. Most sites are unlikely to succeed naturally to forested or shrub communities. Forested and shrub communities, once established, may have fewer problems with invasive species.

9. For other recommendations to promote a resilient site, including climate change adaptation, see the WICCI/NIACS Menu of Adaptation Strategies and Approaches for forested and non-forested wetlands at https://forestadaptation.org/learn/resource-finder/non-forested-wetland-adaptation-strategies-and-approaches

8.2 Recommendations for Improving Mitigation Site Monitoring and Assessment

Sampling Design

For future assessments of mitigation outcomes, we recommend using the Generalized Random Tessellation Stratified (GRTS) study design as outlined in the national study design (ELI 2013) and suggested by Fennessy et al (2013) and Morgan & Hough (2015). GRTS would provide a more balanced probability sample of the target population and a greater number of sites to be sampled in a more efficient manner. However, this method requires accurate geospatial data for all mitigation sites from the target population to be sampled. Currently the mitigation site boundaries are available for most sites on RIBITS but contains no geospatial information within the site boundaries. Given that most sites contained large areas that do not contribute to wetland compensation, and the boundaries of the compensation area are often found only on site plans which are not always available, having this information would greatly improve our ability to select and monitor areas of interest on wetland compensation sites. Having such a dataset would result in a more representative site selection and may require fewer wetland surveys by selecting based on individual wetland polygons rather than whole sites. Sites varied in size dramatically, causing large sites to be overrepresented in the pool of assessed wetlands. If GPS polygons could also be tagged by wetland type, then selection could be stratified to balance the selection of sites by wetland type.

Identifying monitoring units within mitigation sites

Identifying monitoring units early in the development of the site which could then be used in future monitoring would allow more targeted monitoring of community types making more efficient use of limited monitoring resources. We fully expect these community boundaries and types to shift over time, but that information is incredibly helpful. We see value in watching which communities thrive and which struggle in relation to its suite of soil and hydrology conditions. We also are interested in learning more about how restored sites may evolve over time from one community type or dominant species to another in response to climatic shifts, surrounding land use change, or just time (natural shifts from herbaceous to woody dominants, for example).

Among our study sites we found that many compensation areas designated as sedge meadow had either changed over time or never developed in the first place. A goal of continued site monitoring should be to evaluate such functional losses and use adaptive management to address problems. On the other hand, we also found that in several instances small areas of sedge meadow had developed in unexpected areas on mitigation sites, affording a learning opportunity that may increase our ability to reliably restore this community type.

Assessment

Assessing the effectiveness of mitigation begins with assessing the functions of the impacted wetlands to ensure appropriate compensation. It has been noted in the literature that, "what is not measured, is not compensated" when it comes to ecosystem loss (Moreno-Mateos et al. 2015). This rings true when it comes to soil and vegetation functions and is probably true of many other functions as well. However, evidence from these mitigation sites indicates that even well-established assessments of wetlands, like vegetation structure, are not resulting in replaced functions (e.g. forested wetlands).

More detailed vegetation assessments of impacted wetlands, or simply identifying the Eggers & Reed community type could help to improve our ability to replace lost functions. Also, given the high value of soil biogeochemical functions such as denitrification, nutrient cycling, water-holding capacity, and carbon sequestration, the identification of soil type on impacted wetlands could prevent further loss of these functions. Our ability to replace mineral soil functioning via compensatory mitigation appears to be high, but soils with more than 7% organic matter are not routinely being replaced in a reasonable time frame. We recommend that more thorough assessments are done not just before restoration but throughout the life of a restoration to see how conditions change over time.

Regulatory agencies currently collect hydrographs from hydrology wells on each compensatory mitigation site. This practice has become standard since some of the historic mitigation sites studied here. But hydrographs reside deep with an archived PDF and are not available as data to regulators or restorationists. We propose standardizing the data collected and that one agency develop a digital storage location for hydrologic regime data from restorations and available reference sites. Not only would this allow us to better understand how a site's hydrology responds to construction, transpiration rates changes, or adjacent land use, but we could also see on a regional scale how sites respond to shifts in groundwater levels or climate change.

Tracking a mitigation site's soils over time would be invaluable in better understanding how a soil changes in response to removal of farming practices, restored hydrology, enhanced hydrology, etc. And by tracking soil properties over time, we can better understand what functional values are being increased on a site and at what scale they are being increased. This information will be useful as Wisconsin continues to see the impacts of climate change and looks to develop ways to reduce atmospheric carbon emissions and increase carbon storage.

Data management and tracking

We found the Corp's tracking database, RIBITS, to be an invaluable resource, providing the only centralized, digital repository of mitigation records we could find. However, data from many of these older sites was missing, despite many being still open to credit sales. Few documents were digitally available and the Corps of Engineers' Regulatory In-Lieu Fee and Banking Information Tracking System (RIBITS) system was not designed to capture all compensatory mitigation sites or nuanced detail about site history, performance standard expectations, or credit ratios. As a result, a good deal of mitigation detail was lost about these valuable properties.

While record retention has improved since the time when most of the mitigation sites studied in this project were established, we have identified a few key areas where additional improvements are warranted. While RIBITS reliably tracks credit generation and credit purchases from a given mitigation site and does currently store PDF's of key documents for many mitigation sites in its cyber repository, some additional tracking information could prove useful.

We recommend that at least one of the regulatory agencies that reviews and approves mitigation sites develop a tracking database that can manage the following information:

1. A standardized method of documenting, at minimum, impacted wetland type and ideally, a suite of functions and values that were lost due to wetland impacts. Ultimately evaluating the effectiveness of mitigation requires a comparison of wetland values and functions lost to those replaced. Little information is currently available on impacted wetlands beyond acreage.

2. A system to track performance standards by restored community type or monitoring unit. Specifically list the standards and when each standard is achieved. Tracking this information will allow regulatory staff and land managers to easily check-in on closed-out mitigation sites. This tracking could also be

utilized by regulatory agencies when assessing the effectiveness of certain performance standards over time.

3. Spatial outlines and acreage of proposed wetland and upland communities. One of the issues identified with the historic data is that it was hard to track a site's natural communities' footprints and spatial change over time. Mitigation sites are not designed to be stagnant, in fact they are generally designed to be dynamic and sustainable, but knowing which community is successful in any given location (and its soil conditions, slope, aspect, etc.) would greatly inform wetland restoration design in the future. Ideally, this database could calculate acreage gains and losses by wetland type.

4. Monitoring data in a digital format that can be queried by site or region. For example, collecting raw hydrology monitoring data, species lists, or other functional values monitored. The RIBITS system has a wealth of ecological data that, were it complete and accessible, could be valuable to assess mitigation policy, restoration practices, and general ecological understanding.

8.3 Recommended Changes to Mitigation Policy and Guidance

Performance standard modifications

Invasive species

Results from this study show that two invasive species, *Phalaris arundinacea* and *Typha X glauca*, are key drivers of poor quality on mitigation sites. We observed from monitoring reports that in many cases invasive species were present from the beginning of the restoration but not addressed or were addressed ineffectively. We propose strengthening invasive species limit standards especially in early years. We also found no evidence that invaded areas succeed to other community types with time.

However, it is important to add that invasive species, in turn, are often a reflection of the disturbance history of the site and the surrounding landscape, from nutrient inputs, loss of organic matter, soil structure, compaction and sedimentation. In addition to strengthening invasive species limits, we recommend that the root causes of invasive species be addressed in the early planning stages to encourage a lower-maintenance site into the future.

Plant composition

Most plant communities in our study appear to have developed by natural processes, rather than a seed mix or plantings, and were dominated either by native wetland species or invasive species from the surrounding landscape (see Table 11 for full list of dominant plant species). The exception was wet to wet-mesic prairies which relied upon the original plantings to develop their characteristic species composition, and forested communities which failed to develop without plantings in our study. We recommend that performance standards be adapted to require the presence (whether self-seeded or introduced) of appropriate clonal perennial species that can offer an alternative to the default invasive clonal perennials, *Phalaris arundinacea* and *Typha x glauca.* The goal is to discourage the practice of using seed mixes that consist entirely of short-lived perennials that do not persist after the first 5 to 10 years. WDNR is in the process of developing suggested lists of desirable clonal perennials for many target community types using data from natural and successful restored wetlands.

Increase adaptive management opportunities

We recommend that the Interagency Review Team allow more opportunities for adaptive management on mitigation sites. It is nearly impossible to design the hydrology and vegetation perfectly and there will always be a need for modifications and adjustments in response to challenges. It is unrealistic to expect that a mitigation site be planned and restored in one attempt after over one hundred years of impacts such as drainage, plowing, sedimentation, nutrient accumulation, climate change, urbanization, invasive species, and other landscape changes. The previous landowners of restoration sites (e.g., farmers) likely spent lifetimes engaged in improving drainage on their fields by close attention, trial and error and many adjustments along the way. Getting hydrology right is frequently mentioned as the key to establishing native plant communities. A sustainable and high-quality wetland community that provides a range of functional services may take many years to get right. Our study shows that a one-time restoration and then 'walking away' does not always yield healthy wetlands.

Credit ratios

To avoid temporal if not permanent loss of function, credit ratios need to be revised to reflect the absence of evidence that mitigation sites are replacing lost function. In 2008 guidance on compensatory mitigation (40 CFR Part 230) it is recommended that "where the impact is to a high-value resource, more than one-to-one replacement on an acreage basis may be necessary just to achieve functional equivalence between the impact and mitigation sites. Note that replacement ratios may also be greater than one-to-one for other reasons, such as to address uncertainty of success or temporal losses."

While this study did not intensively review how credit ratios were applied as mitigation ratios, through the records that were available it appears that credit ratios on average were 1.4 to 1.6 acres of compensation for every acre of impact. While details about the wetlands impacted were frequently not included in RIBITS, our data suggest that wetlands developed for compensation are falling short when it comes to forested wetlands, wetlands in excellent and good condition, and all wetlands with soil organic matter more than 7%. The delay in the return of soil biogeochemical functioning appears to be longer than we can estimate at this time and certainly longer than two decades. Furthermore, it is not yet clear that many of the wetland restorations are on a trajectory towards replacing biogeochemical functioning. For these reasons and more, many have called for replacement ratios to be higher (Hossler & Bouchard 2010; Zedler & Callaway 1999).

Zedler & Callaway (1999) have suggested that mitigation policy recognize that "(1) compensation sites may never fully replace natural wetland functions, and (2), the time to functional equivalency may exceed the usual monitoring periods". They suggest that mitigation ratios should be adjusted according to what proportion of function compared to a reference they can recover in the short term. Applying this concept, a review of mitigation outcomes in Indiana suggested some communities should be replaced at a ratio of 7.6 acres to every acre impacted (Robb 2002).

Results from our study sites suggest that soil functioning is extremely slow to recover, likely in part due to Wisconsin's cold climate (see Moreno-Mateos 2012). To replace these functions in a timely manner, impacts to wetlands with high organic matter should be replaced with considerably more acreage, with one study suggesting a 5.1: 1 minimum to be conservative until soil organic matter recovery is better understood (Hossler & Bouchard 2010).

As a beginning we recommend a better system of tracking functional losses and what credit ratios are approved for various losses of wetland functions and values. We suggest that further discussion is needed to guide their establishment and usage.

Incentivize rehabilitation

We found that rehabilitation projects (restoring hydrology to pre-existing wetland) had a higher chance of hosting a FAIR or better plant community 13-26 years later and hosted higher-functioning soils as estimated from organic carbon levels. However, rehabilitation is commonly devalued because it does not result in acreage gains.

In order to incentivize rehabilitation, we recommend using hydrological gains to guide estimation of functional lift and allow for additional credits after the monitoring period if the site proves to be able to host a plant community of higher quality or different type than typical under re-establishment. Other incentives include allowing for a shorter monitoring period for such projects or reduction of required financial assurances to reflect the reduced risk of site failure.

Incentivize the restoration of forested wetlands

While we expect that restoration of forested wetlands has increased since the 1992-2005 period of our study, we found a potentially large shortfall in the replacement of functions associated with forested wetlands of all types and conditions among the impacts associated with our study sites. We also found that natural succession to forested wetlands was largely absent. Because longer monitoring periods are required for forested wetlands (10 years typically) bankers are less inclined to pursue them. They may also require a larger cost up front to construct. However, forested wetlands have a wide range of functional benefits from wildlife habitat, to carbon sequestration, to reducing cover of *Phalaris arundinacea*. They also appear to do well once seedlings are established judging from the few restorations in our two studies that planted trees (see also: Matthews et al 2020). Given the high function, historical losses, and rarity of these communities on mitigation sites we propose adjusting credits assigned to forest (and shrub wetlands) to overcome the added burden.

Require funds for long-term maintenance and adaptive management

We recommend that all regulatory agencies require long-term maintenance funds be provided by the banker and a detailed plan for long-term maintenance be required in compensation site plans. Further, we suggest that future iterations of the Guidelines for Compensatory Wetland Mitigation in Wisconsin provide guidance on how to estimate the cost of long-term maintenance and create a menu of the types of maintenance activities that should be considered for various target community types. More risky community restorations, such as shallow-open water communities, may require more long-term maintenance funds than less-risky communities. Now that we have more data on the outcomes of wetland restorations, we are in a better position to estimate risk.

We also recommend requiring long-term endowments or something comparable, to ensure management is funded for all mitigation sites. Further, we recommend that each site plan identify an entity that has agreed to be responsible for all long-term monitoring and management. This entity would be party to the endowment and responsible for completing the pre-determined monitoring and maintenance activities and generating adaptive management plans for review, as needed.

Adaptive management

Because wetland functions are dependent on hydrology, ongoing adaptive management of hydrology may be necessary to maintain sites into the future. Structures used to restore hydrology may need to be adjusted to maintain the desired plant community. In examples from study sites we found that adaptive management of ditch plugs and ditch fills should be required for a period well past the 5-year monitoring period. We found evidence that two sites that employed ditch plugs on large ditches were maintaining a continuous flow of water off site through failing ditch plugs. Ditch fills, while less likely to fail completely, require time to settle and may need spot refilling, or removal of too much fill many years later to avoid ponding or the creation of a berm, both of which can lead to invasive species colonization.

In addition, restoration practices have improved over time and further improvements are likely in the future. For instance, most restorations in our study used excavations and impoundments or partial removal of hydrological alterations to restore hydrology. The use of excavations and impoundments has fallen out of favor over the past several decades and even within the years we captured in this study (1992 -2005) they became less frequent over time. The use of excavations and impoundments and partial alteration removal techniques imply that hydrology was not fully restored on these sites (e.g., many sites that use scrapes and impoundments leave subsurface drainage systems in place) and improvements to functioning may still be possible on these valuable properties as knowledge improves. In the long-term we hope that the fate of many of these under-performing sites is not set in stone but can be adapted in the future based on new knowledge.

9 References

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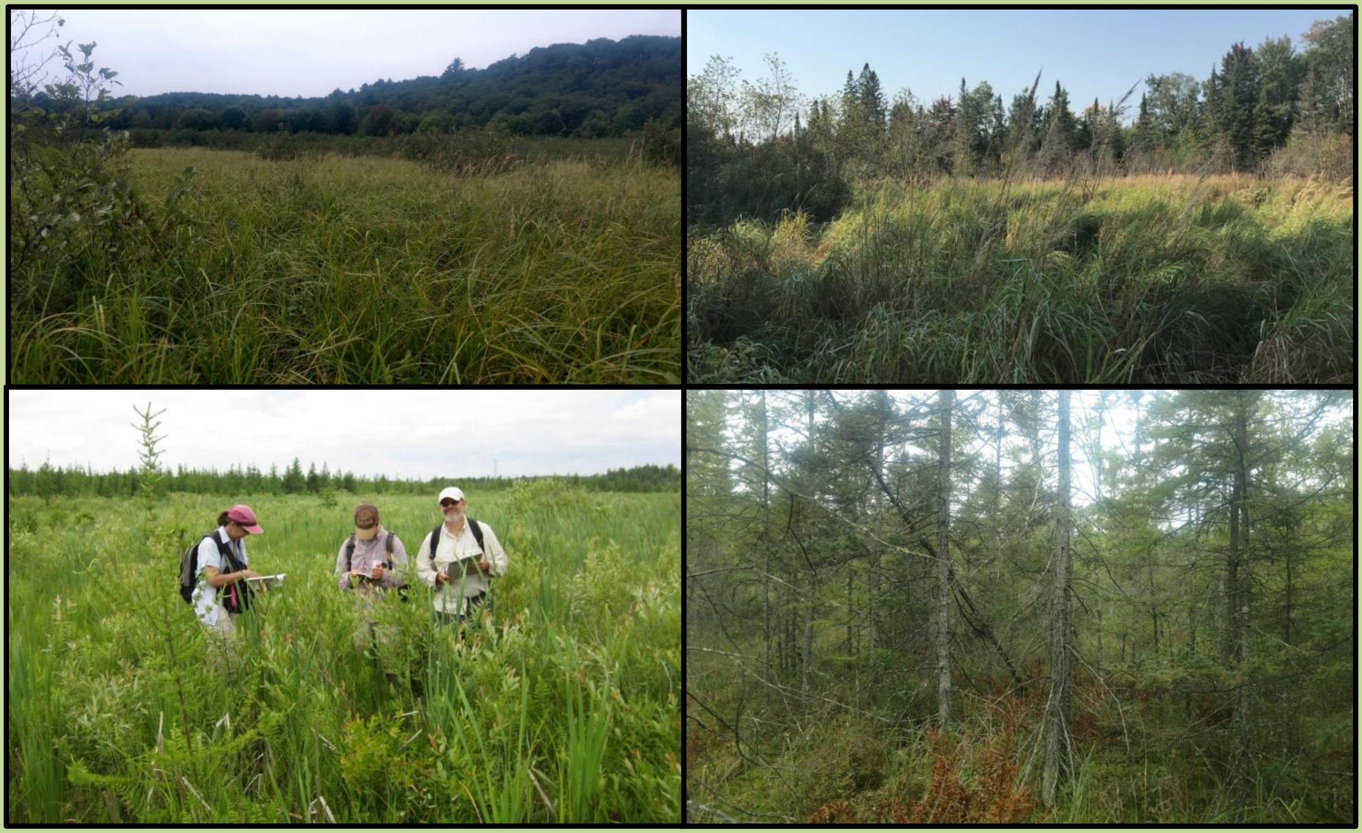
10 APPENDICES

Appendix A: Wisconsin's Provisional Wetland Floristic Quality Benchmarks

Abridged Handout: Wisconsin Wetlands Association 2020 Conference

Provisional Wetland Floristic Quality Benchmarks for Wetland Monitoring and Assessment in Wisconsin



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DISCLAIMER: **This abridged handout is a brief synopsis of the full report listed below. Please see the full report listed on the WDNR Wetland Assessment Reports website for more information and the latest updates:** https://dnr.wi.gov/topic/wetlands/reports.html

Citation

Marti, A.M. and T.W. Bernthal. 2019. Provisional wetland Floristic Quality Benchmarks for wetland monitoring and assessment in Wisconsin. Final Report to US EPA Region V, Grants # CD00E01576 and #CD00E02075. Wisconsin Department of Natural Resources. EGAD # 3200-2020-01.

Executive Summary

Over the past two decades, the Wisconsin Department of Natural Resources (WDNR) has developed and refined the Wisconsin Floristic Quality Assessment (WFQA or FQA) methodology to measure the biological condition or "health" of wetland plant communities. However, plant community metrics calculated as part of WFQA have lacked an overall framework for interpretation and comparison at both regional and statewide scales to utilize them effectively for meeting state and federal regulatory mandates to monitor, assess, and report on the condition of wetlands, to effectively enforce wetland-specific water quality standards that protect wetland health as well as functional values, and to inform wetland restoration, mitigation, and conservation efforts.

To adapt the WFQA method as a comprehensive, quantitative, and repeatable method for intensive, site-level monitoring and assessment of wetland condition, WDNR engaged with partners in 2011 to develop Floristic Quality Assessment Benchmarks for all common wetland community types across Wisconsin, known as the Wisconsin Floristic Quality Assessment Benchmarks Project. Since 2012, WDNR and partners have surveyed nearly 1,100 wetland assessment areas statewide towards this effort. FQA Benchmarks consist of numeric, statistically-derived ranges of FQA scores for a given wetland community type, with each range corresponding to a narrative ranking category (e.g. "Excellent" to "Very Poor") along a gradient of ecosystem disturbance—generally following the Biological Condition Gradient approach promoted by the U.S. Environmental Protection Agency (US EPA).

During an earlier phase of the project, FQA Benchmarks were created for the US EPA Omernik Level III Northern Lakes and Forests Ecoregion, which is detailed in a separate report, *Northern Lakes and Forests Inland Wetland Survey: Relationships between Floristic Quality Assessment and Anthropogenic Stressors – 2012- 2014* by Hlina et al. (2015) of the Lake Superior Research Institute. The current study used a consistent statistical methodology similar to Hlina et al. to generate FQA Benchmarks from timed-meander survey data for the 3 remaining primary Omernik Level III Ecoregions of Wisconsin: North Central Hardwood Forests, Southeast Wisconsin Till Plains, and the Driftless Area. Tables 5, 8, and 11 (respectively) at the end of the Executive Summary contain the resulting suggested provisional Benchmarks based on cover-weighted Mean Coefficient of Conservatism ($w\overline{C}$) scores, including narrative condition rankings, for common wetland plant communities for each Ecoregion.

Based on both Hlina et al. (2015) and information gathered through this study, Benchmarks based on $w\overline{C}$ scores from timed-meander surveys were found to be the most appropriate FQA metric for Benchmark development because of their ability to discriminate the ecological condition of sites along a gradient of human disturbance, whereas other metrics that include measures of species richness in their calculation such as the Floristic Quality Index (*FQI*) and cover-weighted Floristic Quality Index (*wFQI*) were not. Community diversity and the effects of overall cover of individual plant species are captured using $w\overline{C}$, resulting in a more ecologically and statistically defensible assessment metric and corresponding set of Benchmark criteria for comparison. Based on these factors, we suggest that $w\overline{C}$ Benchmark criteria are used as the primary provisional Benchmarks whenever possible when attempting to apply Benchmark criteria for a project. Additionally, Benchmarks are based on use of the WDNR Timed-Meander Survey Protocol (Trochlell 2015), thus this protocol is recommended for wetland plant community survey efforts. However, in the instance that only limited data from plant inventories are available (i.e. a plant species list without cover percentage estimates), preliminary $\overline{C}$ Benchmarks based on Overall Disturbance (Tables II, VII, and XIV in Appendix 2) may be applied with the understanding of their potential limitations.

Benchmarks have numerous potential applications to meet the objectives of the Clean Water Act, including:

- the creation of numeric Tiered Aquatic Life Use criteria to formulate numeric water quality standards as either stand alone or additional/supportive criteria;
- the assessment of the natural quality of sites;
- the assessment of plant community response to restoration, management, and permitting actions;
- aiding in elucidating the relationship between wetland condition and wetland ecosystem functions.

It is emphasized that provisional FQA Benchmarks are an initial step towards evaluating wetland ecosystem condition for Wisconsin and there are a number of associated caveats and limitations for their full implementation:

- Wetland condition and function are two different concepts. Users should realize that even a wetland with in poor plant community condition may still provide some ecosystem functions and services dependent upon the context in which they are considered (e.g. landscape, watershed, wetland complex).
- Plant communities are one biotic community present in wetland ecosystems. FQA-based condition metrics are a promising start towards more complete wetland ecological assessment, but further work is needed. Other biotic and abiotic components (i.e. diatom communities, bryophyte communities, water chemistry, soil physicochemsitry, and soil microbial communities/enzyme activity) may deserve further consideration as indicators to assess ecosystem condition in relationship to anthropogenic stress.
- The Benchmarks in this study and Hlina et al. (2015) should be considered *provisional*, but they may be immediately applied with the understanding that they may be improved over time. Further refinements and investigation strategies are detailed in "Discussion" section.

- FQA surveys using the WDNR Timed-Meander Survey Protocol require substantial taxonomic expertise, as surveys conducted by those with lesser expertise are likely to miss some species and misidentify some species. One option for WDNR to build capacity for implementation could be to use state and regional aquatic monitoring funds to hire wetland assessment experts (e.g. Regional Wetland Ecologists or Wetland Botanists) that could specialize in FQA in addition to other wetland ecology-specific needs.
- Some wetland community types had no or limited Benchmarks. These communities may require more fieldwork efforts and/or data analyses or may require alternative statistical approaches.

Fieldwork and data from this study, combined with that of Hlina et al. (2015), have also generated a number of other valuable applications, including generation of a wetland reference network for Wisconsin for long-term wetland monitoring and assessment (O'Connor and Doyle 2017). This study was also able to statistically evaluate the "distinctness" of a select number of wetland communities as classified by the WDNR Natural Heritage Conservation Program. Furthermore, the plant community and disturbance data gathered for 1,100 wetland assessment areas will surely have many future applications for wetland monitoring and assessment. These data will also support the creation of target species planting lists based on the community composition to inform wetland restoration and mitigation efforts.

The provisional Benchmarks constitute a solid starting point for application of the WFQA as a statistically-valid, cost-effective, repeatable approach that will allow for relative comparisons across sites and time at most scales of interest. Understanding and documenting wetland condition, as well as the stressors likely driving condition, will allow for enhanced management and restoration opportunities of wetlands while also allowing for protection of wetlands already in excellent condition.

REFERENCES AND ADDITIONAL REQUIRED RESOURCES:

[1]O'Connor, R. 2018. Key to Wetland Communities of Wisconsin. Wisconsin Department of Natural Resources Natural Heritage Conservation Bureau. Accessed January 01, 2019. https://dnr.wi.gov/topic/EndangeredResources/documents/KeyToWetlandCommunities.pdf

[2]Trochlell, P.A. 2016. Timed-Meander Sampling Protocol for Wetland Floristic Quality Assessment. Wisconsin Department of Natural Resources. Accessed January 01, 2019. https://dnr.wi.gov/topic/Wetlands/documents/TimedMeanderSamplingProtocol.pdf

[3]WDNR. 2017. The Wisconsin Floristic Quality Assessment Calculator (October 2017). Wisconsin Department of Natural Resources. Accessed January 01, 2019. https://dnr.wi.gov/topic/Wetlands/documents/WDNR_FQA_CALCULATORv1.10.17.xlsx

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Provisional Wetland Floristic Quality Benchmarks for Wetland Monitoring and Assessment in Wisconsin

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For questions, etc. please contact Aaron Marti, WDNR Wetland Assessment Research Scientist, aaron.marti@wisconsin.gov

What are Floristic Quality Benchmarks?

Benchmarks are a means to assess the "health"—specifically, the *biological condition*— of a given wetland using the Wisconsin Floristic Quality Assessment Method. For more information, please read the "Executive Summary" at the beginning of this handout and/or the full report listed in the Disclaimer.

What are some applications of Benchmarks? (a non-exhaustive list)

Assessing the condition of wetlands—natural, restorations, or mitigations. Setting targets for wetland restoration or mitigation. Evaluating effectiveness of management actions or effects of potential disturbances to wetlands (pre- and post-monitoring).

What is the process to use the Benchmarks listed in the tables?

1) ID your area(s) of interest. ID the wetland natural communities within using the [1]*Key to the Wetland Communities.* Each community identified is a separate Assessment Area (AA).
2) Have an **experienced botanist** conduct surveys at each AA following [2]*WDNR's Timed Meander Protocol.* Ensure AAs meet criteria specified in the protocol before beginning.
3) For each AA surveyed, use [3]*WDNR's Floristic Quality Assessment Calculator*. Note the *weighted mean coefficient of conservatism score-all species* ($w\overline{C}_a$) for each AA.
4) Identify which ecoregion your survey(s) was conducted in. Find the corresponding table and search for the applicable community type. Use the $w\overline{C}_a$ score to identify condition.

Citations

[1,2,3]Please see citations after Executive Summary in the front portion of this handout.

Table 1: Hlina, P., N.P. Danz, K. Beaster, D. Anderson, and S. Hagedorn. 2015. Northern Lakes and Forests Inland Wetland Surveys: Relationship between Floristic Quality Assessment and Anthropogenic Stressors. Technical Report 2015-2, Lake Superior Research Institute, University of Wisconsin-Superior, Superior, WI.

Tables 2-4: Marti, A.M. and T.W. Bernthal. 2019. Provisional wetland Floristic Quality Benchmarks for wetland monitoring and assessment in Wisconsin. Final Report to US EPA Region V, Grants # CD00E01576 and #CD00E02075. Wisconsin Department of Natural Resources. EGAD # 3200-2020-01.

Table 1. Provisional Weighted Mean C ($w\overline{C}$) Condition Benchmarks for **Northern Lakes and Forests** Ecoregion Wetlands

		Condition Category				
		Least Disturbed				Most Disturbed
	Natural Community:	Excellent	Good	Fair	Poor	Very Poor
Emergent	Emergent Marsh	> 7.1	5.3 - 7.1	2.8 - 5.2	0.7 - 2.7	< 0.7
	Northern Sedge Meadow	> 7.1	5.3 - 7.1	3.5 - 5.2	< 3.5	
Shrub- Scrub	Shrub Carr	> 5.1			3.9 - 5.1	< 3.9
	Alder Thicket	> 5.3	4.6 - 5.3	4.1 - 4.5	3.8 - 4.0	< 3.8
	Open Bog	> 8.9	8.0 - 8.9	< 8.0		
	Muskeg	> 8.5	7.9 - 8.5	< 7.9		
Forested	Black Spruce/ Tamarack Swamp	> 7.9	7.5 - 7.9	6.7 - 7.4	5.7 - 6.6	< 5.7
	Cedar Swamp (NWMF)	> 7.4	6.9 - 7.4	< 6.9		
	Northern Hardwood Swamp	> 6.2	5.8 - 6.2	3.9 - 5.7	2.5 - 3.8	< 2.5

Provisional Wetland Floristic Quality Benchmarks for Wetland Monitoring and Assessment in Wisconsin

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For questions, etc. please contact Aaron Marti, WDNR Wetland Assessment Research Scientist, aaron.marti@wisconsin.gov

Table 2. Provisional Weighted Mean C (wC̅) Condition Benchmarks for **North Central Hardwood Forest** and **Western Corn Belt Plains** Wetlands

	Natural Community:	**Condition Category:** Least Disturbed: Excellent	Good	Fair	Most Disturbed: Poor	Very Poor
Emergent	Emergent Marsh	> 6.6	5.2 - 6.6	3.1 - 5.1	0.8 - 3.0	< 0.8
	Southern Sedge Meadow	> 6.0	5.0 - 6.0	2.7 - 4.9	1.9 - 2.6	< 1.9
	Northern Sedge Meadow	> 7.0	5.9 - 7.0	2.8 - 5.8	1.4 - 2.7	< 1.4
Shrub	Shrub Carr	> 5.7	4.9 - 5.7	2.0 - 4.8	1.6 - 1.9	< 1.6
Forested	Northern Hardwood Swamp	> 6.1	5.0 - 6.1	2.7 - 4.9	2.5 - 2.6	< 2.5
	Cedar Swamp (NWMF)	> 7.1		6.8 - 7.1	< 6.8	
	Northern Tamarack Swamp	> 7.1	6.7 - 7.1	5.7 - 6.6	4.5 - 5.6	< 4.5

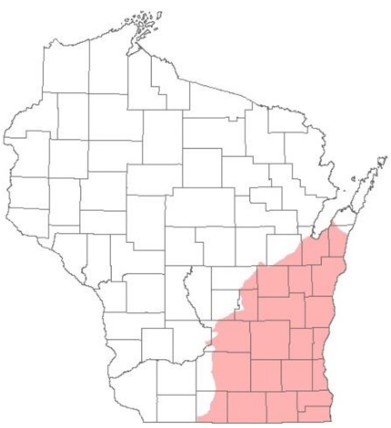

Table 3. Provisional Weighted Mean C (wC̅) Condition Benchmarks for **Southeast WI Till Plains** and **Central Corn Belt Plains** Wetlands

	Natural Community:	**Condition Category:** Least Disturbed: Excellent	Good	Fair	Most Disturbed: Poor	Very Poor
Emergent	Emergent Marsh	> 5.7	4.1 - 5.7	2.1 - 4.0	1.0 - 2.0	< 1.0
	Southern Sedge Meadow	> 6.3	5.6 - 6.3	3.8 - 5.5	1.0 - 3.7	< 1.0
	Wet-Mesic Prairie	> 5.5	4.6 - 5.5	3.1 - 4.5	1.9 - 3.0	< 1.9
	Calcareous Fen	> 7.0	6.2 - 7.0	3.6 - 6.1	2.2 - 3.5	< 2.2
Shrub	Shrub-Carr	> 5.1	4.7 - 5.1	3.2 - 4.6	2.3 - 3.1	< 2.3
Forested	Northern Hardwood Swamp	> 6.2	5.4 - 6.2	3.6 - 5.3	3.4 - 3.5	< 3.4
	Southern Hardwood Swamp	> 4.7	4.0 - 4.7	2.9 - 3.9	2.0 - 2.8	< 2.0
	Cedar Swamp (NWMF)	> 6.5	6.5	5.8 - 6.4	5.3 - 5.7	< 5.3
	Floodplain Forest	> 4.0	3.4 - 4.0	2.3 - 3.3	2.2	< 2.2

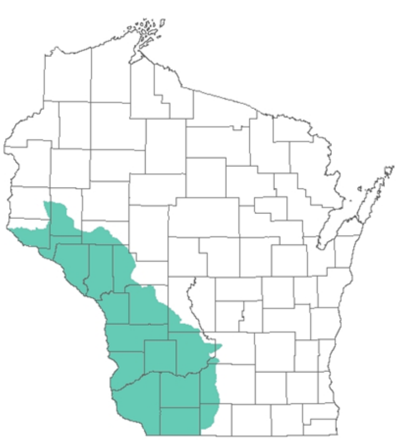

Table 4. Provisional Weighted Mean C (wC̅) Condition Benchmarks for **Driftless Area** Ecoregion Wetlands

	Natural Community:	**Condition Category:** Least Disturbed: Excellent	Good	Fair	Most Disturbed: Poor	Very Poor
Emergent	Emergent Marsh	> 5.2	4.8 - 5.2	3.4 - 4.7	1.6 - 3.3	< 1.6
	Southern Sedge Meadow	> 5.7	5.0- 5.7	3.0 - 4.9	1.1 - 2.9	< 1.1
Shrub	Shrub-Carr	> 5.5	4.4 - 5.5	2.6 - 4.3	1.8 - 2.5	< 1.8
	Alder Thicket	> 4.9	4.5- 4.9	3.8 - 4.4	3.1 - 3.7	< 3.1
Forested	Floodplain Forest	> 4.4	3.5 - 4.4	2.7 - 3.4	2.2 - 2.6	< 2.2