FINAL REPORT

Submitted March 2020

Project No.: 69-3A75-16-031

Feasibility of a Greenhouse Gas Protocol for Restoration and Avoided Drainage of Wetlands in Agricultural Landscapes of the Prairie Pothole Region

Coauthors:
Billy Gascoigne, M.S., Ellen R. Herbert, Ph.D., Sarah K. Mack, Ph.D., Robert R. Lane, Ph.D., and Tim Pearson, Ph.D.

Co-Investigators:
Stephen M. Ogle, Ph.D., Kyler Sherry, M.S.

Partners:

TIERRA RESOURCES

DUCKS UNLIMITED

WINROCK INTERNATIONAL

Project leader:
Billy Gascoigne
bgascoigne@ducks.org

Funding provided by USDA Natural Resources Conservation Service, Conservation Innovation Grant.

USDA United States Department of Agriculture
Natural Resources Conservation Service

Cover photos: michaelfurtman.com

EXECUTIVE SUMMARY

Extensive wetland drainage for agricultural production has occurred across much of the Northern Great Plains and continues at elevated rates in many locales. This analysis investigated the potential to utilize burgeoning carbon markets to financially incentivize wetland retention and/or restoration in the Prairie Pothole Region (PPR) and preserve the array of valuable ecosystem services these systems provide. The primary requirement for carbon market engagement is an industry approved methodology (i.e. protocol) by which carbon accruals and emissions are accounted. The foundation of all protocols is robust greenhouse gas (GHG) estimation of a 'baseline' scenario versus that of a proposed 'project' scenario that assumingly results in GHG savings. Freshwater wetlands—especially temperate seasonal wetlands that are found throughout much of the region and prone to drainage—are inherently dynamic and difficult to quantify. This effort elicited expert opinions to identify market opportunities and inherent challenges, and statistically analyzed a suite of datasets and peer-reviewed literature to assess protocol feasibility.

Biogeochemical models are valuable to a protocol's ultimate success given they replace costly and time-consuming data collection on each individual site, thereby allowing projects to scale. Significant effort was put forth to parameterize the well-established DayCent model for wetland loss and restoration activities; something that had never been attempted. The resulting model, however, had limited predictive power. As a result, we employed a more generalized approach of estimating emission reductions from the scientific literature. The paucity of data on carbon cycling in PPR wetlands (both native and altered) resulted in large uncertainties in the scenarios reviewed. Such uncertainties usually translate into large discounts to a project's credits or even complete ineligibility, making this result valuable in and of itself. The analysis also identified genuine concerns with the quality of existing methane emission measurements in these wetland systems and a growing debate within the climate community on how to measure its true influence. Unlike coastal wetlands that have limited methane emissions due to the presence of saltwater, freshwater wetlands in the PPR do emit methane to various degrees. When applying a standard global warming potential, methane emissions led to low (~1 $MtCO_2e$/acre/year) or negative estimates of net GHG sequestration potential, depending on the data used.

Given the general lack of data, large degrees of uncertainty around emission rates, the inability to parameterize a biogeochemical model, and the effect of methane and debate on proper climate influencing, we conclude that a carbon market protocol for wetland preservation and/or restoration in the PPR is not justified at this time. It should be noted that advancements in GHG data capture—particularly for methane fluxes and advanced aging techniques for comparability—are taking place with additional investment warranted. More research is needed on the sequestration and long-term emissions of seasonal non-drained wetlands and the resulting impacts of eliminating cultivation in that zone of the catchment. Carbon sequestration rates for restored seasonal wetlands have not been reported in the literature. More research is also needed as to whether preserving or restoring surrounding upland grasslands would significantly reduce nutrient loading to the prairie potholes and result in decreased emissions. A biogeochemical model that can be parameterized to handle various site characteristics and negate year-to-year emission variability is likely required for commercial scalability. Lastly, our determination that a wetland protocol is not ripe for success at this time is not synonymous with the conclusion that freshwater wetlands are unimportant from a GHG standpoint. Contrarily, these dynamic systems have long been significant carbon sinks and we need to better understand their role in the quest to mitigate the growing carbon imbalance.

INTRODUCTION

Freshwater wetlands provide numerous ecosystem services that are inherently valuable to individuals, businesses, and society as a whole. The suite of benefits associated with wetlands includes water purification, retention of nutrients and sediments, flood abatement, wildlife habitat suitability, and climate regulation, among others. Historically, freshwater mineral soil wetlands such as the prairie pothole wetlands of the Northern Great Plains stored massive amounts of carbon and exerted a net cooling effect on climate (Euliss et al., 2006). However, large-scale drainage of these systems has released a significant portion of these carbon stores into the atmosphere (Brigham et al., 2006; Euliss et al., 2006; Follett et al., 2001). While carbon sequestration in grasslands, rangelands, and agricultural soils has been commonly promoted for mitigating greenhouse gas (GHG) emissions (Bedard-Haughn et al., 2006; Lal, 2010), there has been relatively little focus on the millions of wetlands imbedded in the agricultural landscapes of the U.S. All the while, freshwater mineral soil wetlands are estimated to store 17% (36 Gt) of the total North American wetland carbon pool (215 Gt; Bridgham et al. 2006) and sequester soil organic carbon at a rate approximately five times faster than restored grasslands (Euliss et al. 2006).

It is widely recognized that economics are a primary driver of wetland drainage, as producers understandably look to expand their total production base. However, this might not be optimal even from an economic standpoint when foregone ecosystem services are considered (Gascoigne et al., 2013), nor when drained wetlands remain too wet to produce a yield sufficient of offsetting that grower's cost (e.g. Fey et al., 2016). For many reasons, landowners may look to restore or retain wetlands on their property and this project aims to expand innovative market-based mechanisms to compensate producers for the provision of ecosystem services from their lands; namely, the feasibility of an isolated freshwater wetland-based carbon market methodology, or "protocol," for the Prairie Pothole Region (PPR).

While seven protocols exist for coastal and peatland (organic soil) wetland restoration, no methodology exists for inland freshwater mineral soils wetlands even though these wetlands hold nearly ten-times the carbon as their coastal counterparts in the U.S. (Nahlik and Fennessy, 2016; Sapkota and White, 2020). Given that drained or otherwise disturbed wetlands contain approximately half of the carbon compared to those that are relatively undisturbed (Nahlik and Fennessy, 2016), wetland drainage and the disturbance of surrounding native grasslands continues to be a major source of GHG emissions associated with the agricultural sector.

The success of any protocol inherently comes down to three attributes:

1. **GHG savings and the state of the science**: The amount of carbon (tons/acre/year) attributed to the practice and the certainty of science underlying emission reductions across a project area.

2. **Practicality & scalability**: The extent to which there is carbon project potential under the proposed protocol in terms of eligible acres and ease of adoption, including socioeconomic factors that might hinder/promote such adoption or reverse landowner behavior.

3. **Cost-effectiveness of credit generation**: The balancing of costs associated with credit development, monitoring, transactions, risk management, and the revenue potential from credits sales.

While these three components should be assessed in concert, there is little value to doing so if a robust accounting framework cannot be derived that results in GHG savings based on defensible science (i.e.

attribute 1). As such, this analysis focuses on GHG accounting of various wetland scenarios pertinent to the PPR, while providing prospective on the other two components.

Study Area and Wetland Dynamics

The PPR within the Northern Great Plains is an area encompassing 177 million acres of the United States and Canada that is characterized by abundant shallow palustrine wetlands within a historical grassland landscape. Pre-settlement, wetlands made up more than 49 million acres or roughly 23% of total land area of the PPR (Gleason et al., 2005). The region has experienced significant land conversion to cropland and deepwater basins ranging upwards of 90% in Iowa to 27% in Montana with annual net losses of 6,200 acres/year (Dahl, 1990; Dahl, 2014). Various policy measures in the Farm Bill have been designed to incentivize wetland retention (e.g. Swampbuster) and restoration (e.g. Conservation Reserve Program and Wetland Reserve Program), with some success between 1997 and 2009 when loss rates slowed and an estimated 87,690 acres of wetland restoration occurred (Gleason, 2011). However, the continuing net loss of wetlands in the PPR indicates that additional incentives (e.g. revenue from potential carbon offsets) are needed to retain these important systems on the landscape.

The wetlands in the PPR range from deeper, permanent systems that in most years retain water throughout the year, to shallow, ephemeral wetlands that are often dry seasonally and during below-average moisture years (Cowardin et al., 1979). Kantrud et al. 1989 estimate that most (>90%) of PPR wetlands have temporary or seasonal water regimes. Semi-permanent and seasonal wetlands experience the highest rate of conversion for crops of any wetland class given they are often dry during the growing season and do not require additional drainage infrastructure to cultivate crops (Renton et al., 2004), and thus are the focus of this protocol feasibility analysis. Semi-permanent wetlands exhibit deeper marsh emergent zones and have the potential for areas of open standing water with submerged aquatic vegetation and zones of emergent vegetation around the edge throughout the entire growing season. Semi-permanent wetlands have historically been drained using various degrees of surface drains and the use of drain tile is expanding throughout the PPR (Johnston, 2013; North Dakota State Water Commission, 2015). It is important to note that all PPR wetlands are highly dynamic and wetland basins expand and contract over time in response to precipitation and evapotranspiration, making demarcating geographic boundaries difficult; an issue that had muddied the regulatory status of these systems for decades.

Carbon Markets and Considerations for PPR Wetland Retention/Restoration

A carbon offset, or 'credit', that is transacted in a carbon market represents the avoidance or removal of one metric tonne (Mt) of carbon dioxide equivalent (CO_2e). There are two general types of carbon markets: the compliance market tied to mandated emission reductions (e.g. cap and trade), and the voluntary market sustained by willing buyers and project developers. Both are supported by industry-standard groups known as registries and/or standards that oversee protocol development, project compliance, and credit issuance. As of this report[1], no wetland protocols have been approved by the oversight committees of the compliance market and seven wetland-based protocols have been approved into the voluntary markets. Zero certified offsets have been issued under a wetland-specific protocol to date[4].

[1] 3/12/2020

Whereas regulations steer market activity in the compliance markets, the voluntary market is driven by private purchasers, primarily midsize to large corporations. The total volume of carbon offsets ($MtCO_2e$) issued worldwide in 2017 was 62.9 million (Hamrick and Gallant, 2018) and a record 42.8 million were retired. Historically, approximately 40% of the offsets generated go unsold. The average price of a voluntary carbon offset has remained relatively stable since 2014. Average prices have been reported between \$3.00-\$4.50, but with prices varying from as low as \$0.10/tCO_2e to as high \$70.00/$MtCO_2e$ (Hamrick and Gallant, 2018). Much of this variance is driven by the uniqueness of the project and its associated environmental and socioeconomic benefits, in which wetland projects would rank high. Both the price variance and the percent of unsold credits highlight the inherent competitiveness existing in the voluntary marketplace. A project type that is likely to have high costs associated with credit generation (i.e. high costs for land acquisition or long-term conservation easements, management, measurement and verification, among other costs) and therefore high revenue needs, must account for these dynamics when assessing market potential.

It is often necessary to take advantage of economies of scale in addressing inherent transaction costs for generating carbon credits from nature-based solutions. Figure 1 below shows the 5.7 million acres of semi-permanent, seasonal, and temporary wetlands throughout the U.S. portion of the PPR in the late 1990s[2]. The abundance of these wetlands on the landscape and the scalability of the carbon protocol concept is illustrated by their areal coverage, historical and current loss rate, and scale of current restoration efforts detailed previously in this report, yielding high potential for scaling avoided loss or restoration credits.

[2] There are ongoing remap efforts to update this geospatial database.

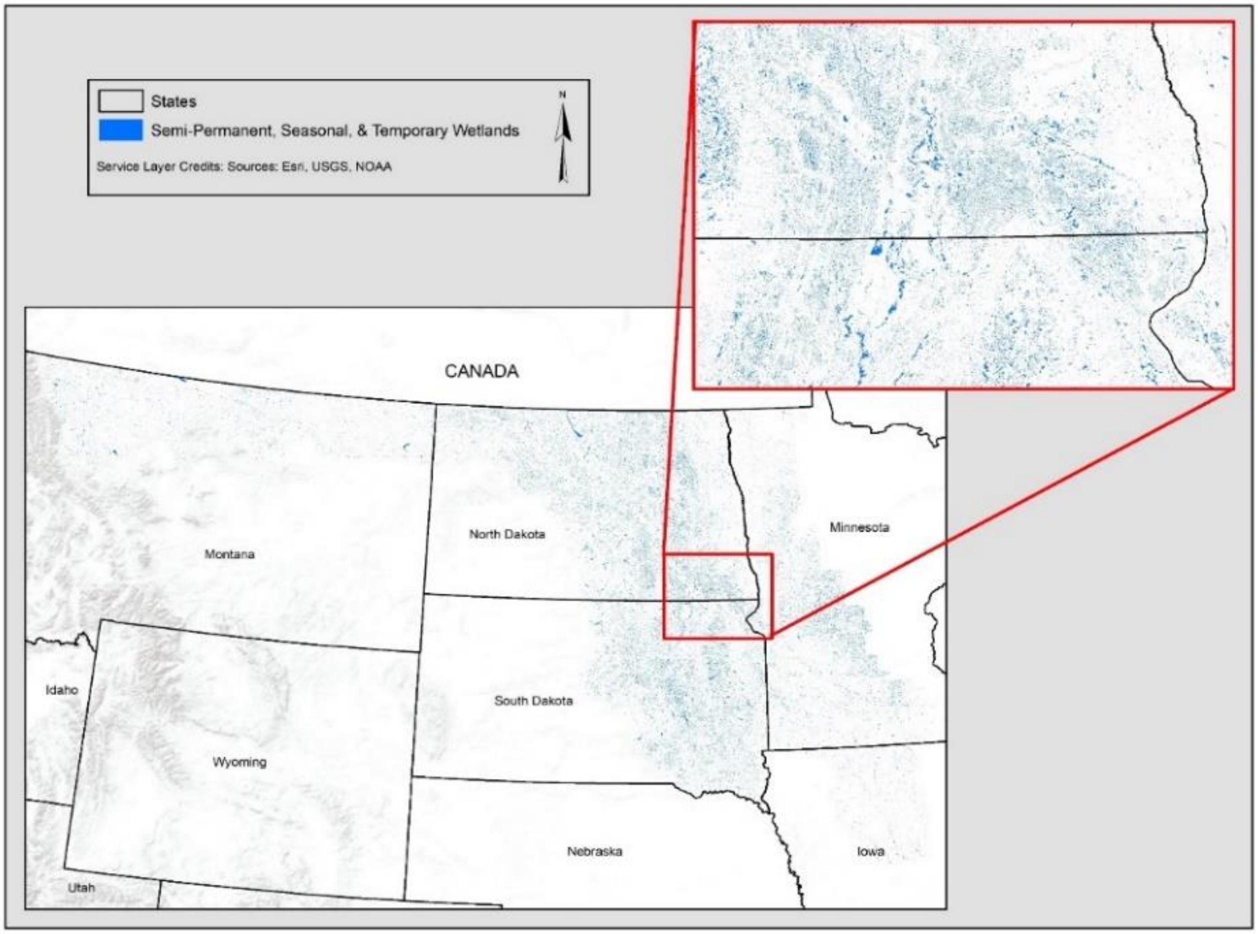

Figure 1. Distribution of semi-permanent, seasonal, and temporary wetlands in the PPR of the U.S in 1997. Developed by the Habitat and Population Evaluation Team of the U.S. Fish and Wildlife Service, summarized in Johnson and Higgins (1997).

PPR Wetland Carbon Cycling

Freshwater wetlands, such as those in the PPR, are highly dynamic systems when it comes to GHG emissions. PPR wetlands and adjacent uplands contain significant quantities of soil organic carbon that have been preserved by slow decomposition in the anaerobic conditions of wetland environments. Carbon is sequestered when wetland plants fix CO_2 during photosynthesis and store the carbon in biomass, which is then transferred to soil as roots and shoot senesce and decay. Negligible amounts of carbon are sequestered in conventionally managed croplands due to oxidation and soil aggregate distribution during tillage and the removal of standing biomass during harvest (Nelson et al., 2008).

When wetlands are drained and converted to cropland a large amount of the stored soil organic carbon is released to the atmosphere, contributing to anthropogenic impacts on the climate (Mitsch and Gosselink, 2007; Pendleton et al., 2012). Research by Euliss et al. (2006) estimates that the drainage and subsequent conversion of a wetland to cultivated cropland will lead to the loss of an average of 15 $MtCO_2$ per acre to the atmosphere over several decades. Cumulatively, the historical conversion of 9.4 million acres of wetlands to croplands in the PPR has led to the loss of an estimated 77 $MtCO_2$ (Euliss et al., 2006). While soil organic carbon stocks can recover to pre-disturbance levels within as few as four years of restoration of semi-permanent wetlands, researchers could not detect an increase in SOC in semi-permanent wetlands post-restoration (Euliss et al. 2006). Carbon emissions related to management practices, such as fuel used for equipment, is also reduced with restoration from cropland or avoided conversion.

Management practices in and around these wetlands can influence not only carbon dioxide (CO_2), but also methane (CH_4) and nitrous oxide (N_2O). Methane is naturally produced in all wetland systems during anaerobic decomposition. Methane rates within wetlands are determined in large part by length of water inundation, water depth, water chemistry, vegetation, temperature, and amount of water-filled pore space in soils (Bansal et al., 2016; Beeri and Phillips, 2007; Pennock et al., 2010). High salinity or alkalinity conditions associated with elevated sulfate concentrations (SO42-), can completely negate methane emissions as sulfate reducing bacteria can outcompete methanogens for substrate due to the higher energy yield of sulfate reduction (Poffenbarger et al., 2011; Herbert et al., 2015). Interestingly, sulfate concentrations in PPR wetlands can vary three orders of magnitude from rainwater (<10 mg/L) to higher than ocean water (>2,000 mg/L; Pennock et al., 2010). Similar variability in the other noted variables has made methane research challenging and difficult to apply at scale.

Nitrous oxide emissions from embedded PPR wetlands are often a direct result of nitrogen fertilizer applications to the surrounding cropped uplands. Applied nitrogen fertilizer can leach via surface runoff or through groundwater that makes its way into the wetland zone of the catchment and accelerate mineralization of organic matter and accelerate emissions of N_2O and even CH_4 (Dunmola et al., 2010; Merbach et al., 2002). Thus, maintaining and/or restoring the upland zone of a wetland catchment to a perennial cover is often critical for lowering net carbon emissions within wetland systems (Gleason et al., 2011; van der Kamp et al., 2003).

As touched on, when assessing these GHGs collectively for a single carbon credit they are expressed as “carbon dioxide equivalents (CO_2e)” using multipliers that standardize their respective degree of heat-trapping influence over a 100-year period, known as “global warming potential (GWP).” For context, the International Panel on Climate Change suggests using a GWP multiplier of 28 for CH_4 and 265 for N_2O (IPCC, 2014). As such, it is quickly recognized that CH_4 and N_2O are vastly more impactive from a GWP standpoint and a relatively small amount of these gases can largely offset significant gains in CO_2 (which has a GWP of 1).[3]

APPROACH TO ASSESSING PPR WETLAND OFFSET FEASIBILITY

Generally speaking, a protocol’s principal objective is to make sure GHG emission reductions are real, additional, measurable, verifiable, and permanent. Registries/standards use protocols to outline criteria that accurately quantify net GHG emission reductions and/or sequestration rates from baseline and project scenarios. A *baseline* scenario is the land management practice(s) currently in effect or the expected land management without the intervention of the carbon project (i.e., status quo). The *project* scenario is the land management practice(s) to be implemented as part of the carbon project that results in GHG benefits.

[3] Summary statistics were calculated using JMP (Version 14) produced by SAS Institute, Inc. (Sall et al. 2017). Analysis of variance (ANOVA) was carried out to detect differences between means. Comparison of means with significant ANOVA tests were made using the Tukey-Kramer Honestly Significant Difference (HSD) test (Sall et al. 2017). All analyses were carried out using a p-value of 0.05 to determine significance.

The net GHG difference between the baseline and project scenario equates to the potential emission reduction 'credit'. Table 1 outlines the potential PPR wetland-related carbon offset projects explored in this report. Note that we differentiate the 'wetland' and 'upland' zones of the larger wetland catchment, as land-use practices of both zones can greatly impact the GHG accounting in the wetland itself. For the purposes of this study, the 'wetland' is being defined as the depressional area where water saturates the soil all year, seasonally, or periodically creating hydric soils. The 'upland' is being defined as the immediate agricultural area that drains into the wetland. The wetland and upland together are defined as the "wetland catchment."

Avoided Drainage/Restoration of Wetlands Carbon Offset Scenarios

Two baseline scenarios exist in the absence of a carbon market promoting the practice of avoided drainage and/or restoration of wetlands:

1. **Drained Wetland -** A wetland that is drained and converted to cropland.
2. **Non-Drained Wetland -** A wetland that is not deliberately drained; however, is tilled and cropped in dry years with no standing water, and has a cropped upland.

There are four accompanying project scenarios that are expected to have positive net GHG benefits in relation to the baseline condition.

1. **Native Wetland Catchment**– an at-risk wetland that historically has never been cultivated, and current and future drainage is avoided.
2. **Restored Drained Wetland Catchment-** A wetland that was previously cultivated using drainage infrastructure and is restored, and the upland is planted back to grass in project scenario.
3. **Restored Non-Drained Wetland** – A wetland that does not have drainage infrastructure but was historically cultivated in dry years, is restored with no further cultivation in all years; however, cultivation and cropping continue in upland zone.
4. **Restored Non-Drained Catchment** – A wetland, that does not have intentional drainage infrastructure, but was historically cultivated in dry years, is restored with no further cultivation in all years; the upland zone is restored back to a non-cultivated grassland condition.

These baseline and project conditions as they relate to potential wetland-based protocols are further outlined in Table 1 below:

Table 1. Baseline and Project land use management scenarios for various carbon offset projects considered by the study.

Project Title	Baseline Management	Project Management	Project Description
1. Avoided Drainage of Native Wetland Catchment	Drained wetland used for crop production	Native wetland catchment	Protection of an at-risk native wetland with grassland uplands
2. Restoration of Drained Wetland Catchment	Drained wetland used for crop production with drainage infrastructure	Restored wetland catchment	Drainage infrastructure is removed and restoration of wetland(s) that had been farmed, and uplands restored to grasslands
3. Restoration of Wetland from Cultivation in Drought Years	Non-drained wetland used for crop production in drought years	Restored non-drained wetland	Restoration of wetland(s) that had been cropped during drought, cultivation continues in the upland zone of the catchment
4. Restoration of Wetland Catchment from Cultivation in Drought Years	Non-drained wetland used for crop production in drought years	Restored non-drained wetland catchment	Restoration of a wetland that has been cropped in drought years and upland converted to grassland

Data Analysis

Measuring and monitoring GHG emissions by each individual wetland site would undoubtedly be too resource-intensive, as has been the case with most other land-based carbon protocols. As such, protocols look to establish quantification approaches that utilize models with defensible predictive abilities. Biogeochemical models provide a means to evaluate chemical, physical, geological and biological properties of an ecosystem. These biophysical models allow validated inputs to be applied to estimate the average impact of changes to land use, practices, and/or inputs based either on process-based deterministic or stochastic models. When this approach is not possible, a context-specific empirical model can be utilized if sufficient data exists and relationships among variables can be extrapolated. Empirically-based estimates can be evaluated with independent data to determine if the resulting models accurately predict annual 'fluxes' and/or cumulative 'stock' emissions.

Model Parameterization

The potential benefits of a biogeochemical model parameterized for PPR wetland scenarios warranted significant effort. As noted earlier, such a model would directly reduce costs and allow projects to scale, while also providing credible estimates that tease out site-to-site variability often observed with individual field measurements. However, after numerous modeling attempts with an array of data sources with the established DayCent model[4], accurate predictive capacity was not achieved. As an alternative, flux data

[4] DayCent is a daily time-step version of CENTURY biogeochemical model (Parton et al. 1994). Our attempt to parameterize it was based on a linear mixed-effect model that accounted for fixed and random effects. Fixed effects included temperature, water filled pore space, standing water, soil texture, pH, soil sulfate concentration, and bulk density. The random effects addressed dependencies in data collected from the same site and data from the same

from peer-reviewed literature was compiled to build an empirical model that could assess the potential GHG emission reductions between the baseline and project scenarios. We identified a limited number of studies that had measurements of carbon stock values from intact sites, carbon accumulation rates, and flux values post cultivation for the various wetland types and across all scenarios (Appendix 1).

Data synthesis for empirical modeling

Despite extensive literature review, it was determined there is not sufficient GHG data for the diversity of wetlands in the region and across all project and baseline scenarios. Furthermore, it was difficult to ascertain the history of the sites used for data collection in regard to our proposed scenarios, particularly drained wetlands. The most extensive and relevant emissions data is from an observational study spanning four years and extending from North Dakota to northern Iowa by Tangen et al. (2015)[5]. We choose to combine it with published data on estimates of carbon sequestration based on a chronosequence of restored wetlands (Euliss et al. 2006), from comparable systems in the same geography to develop a first approximation of outcomes.

EMPIRICAL MODEL RESULTS

Average cumulative annual emissions of methane plus nitrous oxide in CO_2e from Tangen et al. (2015) were calculated for the upland and wetland zones for each hydrologic wetland class (semi-permanent, seasonal) and land use (drained crop, non-drained crop, reference, restored) separately to obtain annual gross emissions in CO_2e. After calibration and data summation, however, we were left with data outcomes far contrasting of other peer-reviewed research. Although our calculations produced total GHG emissions for restored wetlands ranging from 0.58 ± 0.13 tCO_2e/ac/yr, which is similar to other studies (Figure 2a), we produced total GHG emissions of 427.25 ± 96.27 tCO_2e/ac/yr for restored semi-permanent wetlands; an estimate that is 4 to 100 times higher than reported in other regional studies (Figure 2b).

Much of this variance was caused by methane estimates. To evaluate the accuracy of our single data source (Tangen et al., 2015) for emissions modeling, we compared the range (5^{th} -95^{th} percentile) of cumulative annual emissions of methane plus nitrous oxide in CO_2e reported in the additional publications (Appendix 1) that reported emissions for one or more of the land-use categories in Tangen et al. (2015) for comparison (Figure 2a&b).

---

time series. The flux rates of methane and nitrous oxide had non-normal distributions and unequal variances, so a weighted regression method was applied to meet model assumptions.

[5] This dataset contained 152 observations of cumulative growing season fluxes from native wetlands, 62 from drained croplands, 94 from restored wetlands, 48 from non-drained croplands, and 54 observations from non-drained croplands.

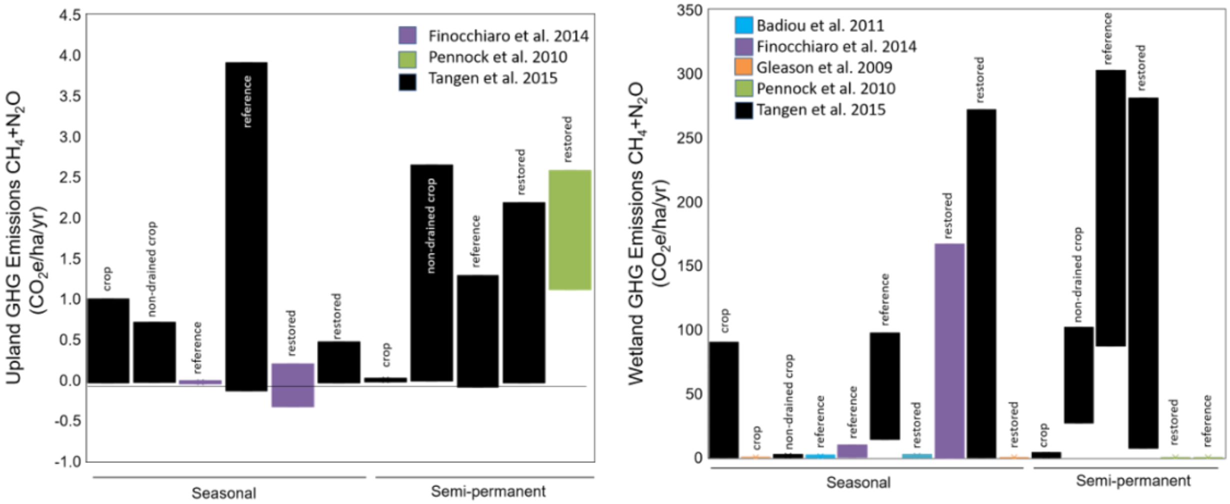

Figure 2. Comparison of the range (5th -95th percentile) seasonal and semi-permanent hydrologic classes for upland (a) and wetland (b) greenhouse gas emissions (cumulative annual emissions of methane plus nitrous oxide) in CO_2e for the prairie pothole region. Color of bars indicate the study and bar label denote land use type within the specified hydrologic class.

Figure 2 portrays the disparity. We posit that much of this result is driven by the fact that the data collection methods used by Tangen et al. were non-standard. Specifically, the researchers used a single-point gas flux method wherein the gasses accumulated in a closed chamber are only sampled one time at the end of an incubation and compared with a sample of ambient air. Standard flux methodology relies on collecting multiple samples (upwards of five) and calculating the flux rate from the linear portion of the curve. This is done because gas fluxes may be partitioned into diffusive versus ebullition (sporadic bubbles) fluxes. Ebullition is often triggered by the pressure of mounting the measurement chamber and thus greatly inflating the concentration of gas in the chamber. We are aware that Tangen et al. (2015) methodology is being compared to standard methodology to establish what, if any, bias this method may induce (per comm P. Badiou). Unfortunately, for our analysis a complete set of baseline and project scenarios were not available from other publications, however it is clear that if the Tangen et al. (2015) estimates used in our empirical estimation are indeed overestimates, the direction of the next fluxes from restored wetlands could change from net emissions to net sequestration. Other analyses of freshwater mineral soil wetland generally conclude there are positive net carbon fluxes (sequestration potential) between 1.4 and 18.48 $MtCO_2e$/ac/yr (Bernal and Mitsch 2012; Euliss et al., 2006 as cited in Alcock, 2017; Lu et al., 2017, Mitsch et al., 2013).

DISCUSSION

Challenges of Modelling GHG Emissions from the PPR Region

This effort revealed many challenges with modeling GHG emissions from PPR wetlands, namely the data collection itself. It is understandably difficult to have a large sample of reference native wetlands, those that have been degraded, and those that have been restored. Of those attempting to measure GHG sequestration and/or emission in restored wetlands, only one had included techniques such as chronosequencing to determine the time since restoration took place. It is easy to recognize how a restored wetland that is twenty years old would have different GHG dynamics than a wetland that was restored only two years ago. Furthermore, the naturally high variability of methane fluxes occurring in wetlands proposes another challenge. If not gathered in a comprehensive manner, data calculations can often be skewed by "hotspots" of methane and nitrous oxide emissions that occur commonly within small areas of wetlands. Similarly, the uneven distribution of sulfate can lead to GHG measurement inaccuracy (Dunmola et al., 2010). Consequently, these dynamics require intensive sampling across wetland sites to ensure high accuracy and precision of emission estimates. Unfortunately, the most relevant data for a potential wetland protocol has not been collected in such a manner to date.

Global Warming Potentials and Radiative Forcing Models—Adding further complexity, there is growing debate on how to equalize the various GHGs in terms of their anthropogenic global warming at a given time. This is required for carbon market protocols that guide the transaction of an offset associated with a specific impact at a specific point in time. Conventional approaches have used GWP multipliers to conflate all GHGs into CO_2e over 100 years. While methane is certainly more potent than carbon, it has a shorter atmospheric lifespan and its relative climate impact reduces significantly over time. Not only does GWP and net radiative forcing depend upon the lifetime of a particular gas in the atmosphere, it also depends upon whether the gas is emitted in a sustained flux or a pulsed flux (Balcomb, et al 2018; Neubauer and Megonigal, 2015). Recent modeling suggests using the conventional GWP approach does not accurately capture the different behaviors of both long-lived climate pollutants (like CO_2) and short-lived climate pollutants (like CH_4), and in fact misrepresents their impact on global temperature (Allen et al., 2016; Balcombe et al., 2018; Cain et al., 2019). Critical to freshwater wetlands and a potential protocol, these researchers posit that the conventional approach greatly overestimates the cumulative effects of methane.

Improving Quantification of Carbon Budgets in the PPR

The estimated net offsets from the empirical analysis are associated with large uncertainties, which would likely lead to high monitoring costs, large deductions in eligible offsets, or even invalidation of an entire project. The development of a process-based model could be an option for improving the precision of the estimates. Process-based models can predict emissions based on key driving variables and a more complex mathematical representation of structure underlying wetland ecosystems. The drivers include weather, edaphic characteristics, catchment topography, surrounding landscape conditions, and management. Successfully calibrating a process-based model to accurately represent wetland dynamics could reduce monitoring costs by limiting the need for an intensive measurement campaign to monitor and verify emission reductions.

In general, more research is needed on PPR wetlands. The final model used in this study was developed from the available scientific literature where sequestration and emission rates were never measured simultaneously, which is not optimal. More research is needed on the sequestration and long-term emissions of seasonal non-drained wetlands and the resulting impacts of eliminating cultivation in that zone of the catchment. Carbon sequestration rates for restored seasonal wetlands have not been reported in the literature. Proper aging of restoration activities would greatly improve comparisons and generate more realistic emission curves. More research is also needed as to whether preserving or restoring surrounding upland grasslands would significantly reduce nutrient loading to the prairie potholes and result in decreased emissions. Similarly, data collection efforts need to attempt to distinguish atmospheric GHG dynamics (i.e. Net Ecosystem Productivity), not just total system carbon given carbon protocols will often require a deduction of carbon additions originating from outside of the site.

Cost Considerations

Our analysis did not dive deep into cost considerations given the lack of GHG data fundamental to pursuing any protocol. However, cost per credit generated is one of three major attributes to a successful protocol, and as GHG data collection advances in the future, these considerations will become more imperative. Costs of project development are wide ranging, including but not limited to landowner engagement, landowner payments, buyer/broker dealings, modeling and report writing, third-party verification, legal contracting, registration fees, monitoring, marketing, and possibly debt-financing. Surmounting costs have crippled, if not greatly hindered, many protocols to date. While these costs categories are relatively fixed, significant strides have been made to lower actual costs incurred. Project aggregation of many participating sites has proven successful at lowering costs per project and would likely be required for a wetland protocol. In order to do so, however, quantification methods that handle the inherent variability across wetlands must be devised.

As noted throughout, a model that can be parameterized to handle various site characteristics and negate year-to-year emission variability is likely required. Without it, data collection and annual monitoring costs would be excessive. Uncertainty around GHG data in and of itself represents a significant cost within one's business planning because it typically results in discounts to your credit volume. That is, project developers are required to be conservative and uncertainly in any form often requires you to deduct a proportional percentage of your credits. The adoption and scaling of many protocols have been hindered by this. The ability to stack other conservation outcomes resulting from prairie wetland conservation—like water quality credits—could impact the revenue opportunities, but would require additional investigation.

CONCLUSIONS

The development of a successful carbon market methodology is dependent on significant GHG savings between baseline and project scenarios, scalability of offsets, and ability for offsets to be sold under current market demands. This feasibility analysis assessed all three components, with an emphasis on defensible GHG estimates for potential wetland project scenarios. Unfortunately, we were not able to parameterize a biogeochemical model that had predictive capacity, nor rely on the available peer-reviewed literature for a more generalized empirical approach. Data gathering for the proposed carbon project scenarios is difficult, and the state of the science does not seem sufficient for project adoption and scalability at this time.

The socioeconomic value of PPR wetlands is immense and potential financing options to compensate private landowners for the public goods provide by these systems are needed. This is ever more important as increased demands are placed on land resources and wetland drainage is the result. The charismatic nature of freshwater wetlands and the array of ancillary ecosystem services provided could be attractive to the private sector interested in nature-based climate solutions. Carbon markets have been a vehicle to link these investors with conservation concerns; however, certain conditions must be met for protocols to be successful. At the forefront, there must be robust science behind proposed carbon project concepts and generally limited uncertainly around what they mean in terms of climate change mitigation.

After much effort, we uncovered challenges with modelling, the available scientific data, and continued debate within the science community on how to handle the climate influence of short-lived gases like methane; wetland's biggest question mark at this point.

This feasibility analysis was proposed as a first step given the known dynamics within freshwater wetland systems and the number of carbon protocols that have been developed yet never used. While it was determined that a carbon protocol for PPR wetlands is not ripe for success at this time, this conclusion is very valuable in and of itself. The analysis also identified many gaps within the science and ways in which data collection can improve in a manner conducive to carbon credit generation in the future. The scale of these wetlands, their rate of loss, and potential for restoration within working agricultural operations warrants more investment in the science. Even if sequestration gains are limited within wetlands, they could be additive to existing protocols based on avoided grassland conversion (i.e. uplands) that currently require wetland acres to be omitted. If the GHG data improves and process-based models can be derived, the potential for a protocol should be revisited. These advancements along with a new consensus around methane's climate influence could prove enough to have an impactful, scalable, and cost-effective methodology in the future.

ACKNOWLEDGEMENTS

The authorship team would like to acknowledge the support provided by all individuals that participated in the early working group, especially Sheel Bansal, Brian Tangen, and Pascal Badiou. Our sincere thanks to our funding partner, the Natural Resources Conservation Service, and the forward-thinking team behind the Conservation Innovation Grant Program.

LITERATURE CITED

Alcock, J. 2017. Wetland and Grassland Retention as and Effective Carbon Management Strategy in Alberta. A report to Ducks Unlimited Canada.

Allen, M.R., Fuglestvedt, J.S., Shine, K.P., Reisinger, A., Pierrehumbert, R.T., Forster, P.M. 2016. A new use of Global Warming Potentials to compare cumulative and short-lived climate pollutants. *Nature Climate Change*, 6: 773-776.

Badiou, P., Mcdougal, R., Pennock, D., Clark, B. 2011. Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian prairie pothole region. *Wetland Ecology Management*, 19: 237–256.

Balcomb, P., Speirs, J.F., Brandon, N.P., Hawkes, A.D. 2018. Methane emissions: choosing the right climate metric and time horizon. *Environmental Science: Processes Impacts*, 20: 1323.

Bansal, S., Tangen, B., Finocchiaro, R. (2016). Temperature and hydrology affect methane emissions from prairie pothole wetlands. *Wetlands*, 36(2): 371-381.

Bedard-Haughn, A., Matson, A.L., Pennock, D.J. 2006. Land use effects on gross nitrogen mineralization, nitrification, and N_2O emissions in ephemeral wetlands. *Soil Biology Bioche*. 38: 3398-3406.

Beeri, O., Phillips R.L. 2007. Tracking palustrine water seasonal and annual variability in agricultural wetland landscapes using Landsat from 1997 to 2005. *Global Change Biology*, 13: 897-920.

Bernal, B., Mitsch, W.J. 2012. Comparing carbon sequestration in temperate freshwater wetland communities, *Global Change Biology* 18: 1636-1647.

Bridgham, S.D., J.P. Megonigal, J.K. Keller, N.P. Bliss, and C. Trettin. 2006. The carbon balance of North American wetlands. *Wetlands*, 26: 889-916.

Cain, M., Lynch, J., Allen, M.R., Fuglestvedt, J.S., Frame, D.J., Macey, A.H. 2019. Improved calculation of warming-equivalent emissions for short-lived climate pollutants. Climate and Atmospheric Science, 29, 9p.

Cowardin, L.M., Carter, V., Golet, F.C., LaRoe, E.T. 1979. Classification of wetlands and deepwater habitats of the United States. U.S. Fish and Wildlife Service report FWS/OBS-79/31. http://www.fws.gov/wetlands/Documents/classwet/index.html

Dahl, T.E. 1990. Wetlands losses in the United States, 1780’s to 1980's. Report to the Congress. United States.
Dahl, T.E. 2014. Status and trends of prairie wetlands in the United States 1997 to 2009. U.S. Department of the Interior; Fish and Wildlife Service, Ecological Services, Washington, D.C.
Dunmola, A. S., Tenuta, M., Moulin, A. P., Yapa, P., Lobb, D. A. 2010. Pattern of greenhouse gas emission from a Prairie Pothole agricultural landscape in Manitoba, Canada. *Canadian Journal of Soil Science,* 90: 243-256.
Euliss, N. H., Gleason, R. A., Olness, A., Mcdougal, R. L., Murkin, H. R., Robarts, R. D., Bourbonniere, Warner, B. G. 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. *Science Total Environmental*, 361: 179–188.
Fey, S., Kyveryga, P., Connor, J., Kiel, A., Muth, D. 2016. Within-Field Profitability Assessment: Impact of Weather, Field Management and Soils. A paper from the Proceeding of the 13th International Conference on Precision Agriculture, July 31-August 4, 2016, St. Louis, MO, USA.
Follett, R.F., Pruessner, E.G., Samson-Liebig, S.E., Kimble, J.M., Waltman, S.W. 2001. Carbon sequestration under the Conservation Reserve Program in the historic grassland soils of the United State of America. In: Lal, R. (Ed.), Carbon sequestration and greenhouse effect. *Soil Science Society of America*, Special Publication No. 57: 27-40.
Gascoigne, W.R., Hoag, D.L.K., Johnson, R.R., and Koontz, L.M., eds. 2013. Dynamics of land-use change and conservation in the Prairie Pothole Region of the United States—Environmental and Economic implications with linkages to rural community well-being. U.S. Geological Survey Professional Paper 1800, 65p.
Gleason, R. A., Euliss Jr, N. H., McDougal, R. L., Kermes, K. E., Steadman, E. N., & Harju, J. A. 2005. Potential of restored prairie wetlands in the glaciated North American prairie to sequester atmospheric carbon. USGS Northern Prairie Wildlife Research Center, 92.
Gleason, R. A., Euliss, N. H., Tangen, B. A., Laubhan, M. K., & Browne, B. A. 2011. USDA conservation program and practice effects on wetland ecosystem services in the Prairie Pothole Region. *Ecological Applications* 21: S65-S81.
Hamrick, K., Gallant, M. 2018. Voluntary Carbon Markets: Outlooks and Trends. Forest trends.
Herbert, E.R., Boon, P., Burgin, A.J., Neubauer, S.C., Franklin, R.B., Ardon, M., Hopfensperger, K.N., Lamers, L.P.M., Gell, P. 2015. A global perspective on wetland salinization: Ecological consequences of a growing threat to freshwater wetlands. Ecoshere 6(10): 1-43.
IPCC. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups, I, II, and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.
Johnson, R.R., Higgins, K.F. 1997. Wetland Resources of Eastern South Dakota. Brookings: South Dakota State University. 102pp.
Johnston, C. A. 2013. Wetland losses due to row crop expansion in the Dakota Prairie Pothole Region. Wetlands, 33(1), 175-182.
Kantrud, H.A., Krapu, G.L., Swanson, G.A. 1989. Prairie basin wetladns of the Dakotas: a community profile. Biological Report 85. U.S. Fish and Wildlife Service, Washington, D.C., USA.
Lal, R. 2010. Managing Soils and Ecosystems for Mitigating Anthropogenic Carbon Emissions and Advancing Global Food Security. *BioScience*, 60(9): 708-721.
Lu, W., Xiao, J., Liu, F., Zhang, Y., Liu, C.A., Lin, G. 2017. Contrasting ecosystem CO2 fluxes of inland and coastal wetlands: a meta-analysis of eddy covariance data. *Global Change Biology* 23: 1180-1198.
Merbach, W., Kalettka, T., Rudat, C., Augustin, J. 2002. Trace gas emissions from riparian areas of small eutrophic inland waters in Northeast-Germany. In: Broll, G., Merback, W., Pfeiller, E.-M. (Eds.), Wetlands in Central Europe: Soil Organisms, Soil Ecology Processes and Trace Gas Emissions. Springer-Verlag, Berlin, Germany, pp. 235-244.
Mitsch, W. J., & Gosselink, J. G. 2007. Wetlands. Hoboken. ed: John Wiley & Sons, Inc.
Mitsch, W.J., Bernal, B., Nahlik, A.M., et al. 2013. Wetlands, carbon, and climate change. *Landscape Ecology*, 28(4): 583-597.
Nahlik, A.M., Fennessy, M.S. 2016. Carbon storage in US wetlands. Nature Communications, 7, article no. 13835, p. 9.
Nelson, J.D.J., Schoenau, J.J., Malhi, S.S., 2008. Soil organic carbon changes and distribution in cultivated and restored grassland soils in Saskatchewan. *Nutr. Cycl. Agroecosyst*. 82, 137–148.
Neubauer, S.C., Megonigal, J.P. 2015. Moving beyond global warming potentials to quantify the climatic role of ecosystems." *Ecosystems*, 18(6): 1000-1013.

North Dakota State Water Commission. 2015. Trends in North Dakota Field Water Management. The Oxbow Publication: 16-18.

Parton, W.J., Schimel, D.S., Ojima, D.S., Cole, C.V. 1994. A general model for soil organic matter dynamics: sensitivity to litter chemistry, texture and management. *In* Quantitative Modeling of Soil Forming Processes, Soil Science Society of America, by Bryant, R.B., Arnoldm, R.W. (eds.). Madison, USA, pp. 147-167.

Pendleton, L., D.C. Donato, B.C. Murray, S. Crooks, W.A. Jenkins, S. Sifleet, C. Craft, .W. Fourqurean, J.B. Kauffman, N. Marbà, P.k Megonigal, E. Pidgeon, D. Herr, D. Gordon, and Alexis Baldera. 2012. Estimating global "blue carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLoS One 7(9): e43542.

Pennock, D., Yates, T., Bedard-Haughn, A. Phipps, K., Farrell, R., McDougal, R. 2010. Landscape controls N_2O and CH_4 emissions from freshwater mineral soil wetlands of the Canadian Prairie Pothole region, *Geoderma*, 155: 308-319.

Poffenbarger, H., Needelman, B., Megonigal, J. 2011. Salinity influence on methane emissions from tidal marshes. *Wetlands*, 31: 831–842.

Renton, D.A., Mushet, D.M., DeKeyser, E.S. 2015. Climate change and prairie pothole wetlands—Mitigating water-level and hydroperiod effects through upland management: U.S. Geological Survey Scientific Investigations Report 2015-5004, 21 p.

Sall, J., M. Stephens, A. Lehman, and S. Loring. 2017. JMP Start Statistics: A guide to statistical and data analysis using JMP, sixth edition. Cary, NC: SAS Institute Inc. 660p.

Sapkota, Y., White, J.R. 2020. Carbon offset methodologies applicable for coastal wetland restoration and conservation in the United States: A review. *Science of The Total Environment*, 701(20), p. 134497.

Tangen, B. A., Finocchiaro, R. G., Gleason, R. A. 2015. Effects of land use on greenhouse gas fluxes and soil properties of wetland catchments in the Prairie Pothole Region of North America. *Science of the Total Environment*, 533. 391-394

van der Kamp, G., Hayashi, M., Gallén, D. 2003. Comparing the hydrology of grassed and cultivated catchments in the semi-arid Canadian prairies. *Hydrological Processes*, 17: 559-575.Nahlik, A.M., Fennessy, M.S. 2016. Carbon storage in US wetlands. Nature Communications, 7, no. 13835, p. 9.

Appendix 1.

Available flux data for PPR wetlands in the US and Canada.

Reference	***Upland/ Wetland***	***Associated Wetland Class***	***Landuse Type***	***PPR Geography***	***Linear Flux Model***	***CH_4 Emissions $tCO_2e\ ha^{-1}\ yr^{-1}$***		***N_2O Emissions $tCO_2e\ ha^{-1}\ yr^{-1}$***		***Carbon Accumulation Mg C $ha^{-1}\ yr^{-1}$***	
						low	***high***	***low***	***high***	***low***	***high***
Tangen et al. 2015	upland	seasonal	crop	US	single point	-0.04	1.00	0.29	3.67	na	na
Tangen et al. 2015	upland	seasonal	non-drained crop	US	single point	-0.03	0.71	0.47	1.17	na	na
Finocchiaro et al. 2014	upland	seasonal	reference	US	multipoint	-0.05	-0.01	0.01	0.03	na	na
Tangen et al. 2015	upland	seasonal	reference	US	single point	-0.13	3.91	0.07	0.16	na	na
Finocchiaro et al. 2014	upland	seasonal	restored	US	multipoint	-0.33	0.20	0.04	4.12	na	na
Tangen et al. 2015	upland	seasonal	restored	US	single point	-0.04	0.47	0.07	0.24	na	na
Tangen et al. 2015	upland	semi-permanent	crop	US	single point	-0.02	0.02	0.29	1.13	na	na
Tangen et al. 2015	upland	semi-permanent	non-drained crop	US	single point	-0.01	2.65	0.58	1.21	na	na
Tangen et al. 2015	upland	semi-permanent	reference	US	single point	-0.09	1.29	0.07	0.27	na	na
Tangen et al. 2015	upland	semi-permanent	restored	US	single point	-0.04	2.18	0.07	0.27	na	na
Pennock et al. 2010	upland	semi-permanent	restored	CAN	multipoint	1.11	2.58	0.09	0.97	na	na
Tangen et al. 2015	wetland	seasonal	crop	US	single point	-0.01	90.24	0.47	1.30	na	na
Gleason et al. 2009	wetland	seasonal	crop	US	single point	0.12		0.05		na	na
Tangen et al. 2015	wetland	seasonal	non-drained crop	US	single point	0.02	2.49	0.59	1.65	na	na
Badiou et al 2011	wetland	seasonal	reference	CAN	multipoint	0.00	1.93	0.01	0.03	na	na
Finocchiaro et al. 2014	wetland	seasonal	reference	US	multipoint	0.17	9.93	0.04	0.06	na	na
Tangen et al. 2015	wetland	seasonal	reference	US	single point	14.32	97.48	0.22	0.89	na	na
Badiou et al 2011	wetland	seasonal	restored	CAN	multipoint	0.03	2.66	0.00	0.15	2.5	6.1
Finocchiaro et al. 2014	wetland	seasonal	restored	US	multipoint	-0.27	166.91	0.06	0.94	na	na
Tangen et al. 2015	wetland	seasonal	restored	US	single point	0.01	271.84	0.16	1.20	na	na
Gleason et al. 2009	wetland	seasonal	restored	US	single point	0.04		0.05		na	na
Tangen et al. 2015	wetland	semi-permanent	crop	US	single point	-0.02	4.05	0.83	1.70	na	na
Tangen et al. 2015	wetland	semi-permanent	non-drained crop	US	single point	26.85	101.74	0.31	0.54	na	na
Tangen et al. 2015	wetland	semi-permanent	reference	US	single point	87.33	302.55	0.08	1.15	na	na
Tangen et al. 2015	wetland	semi-permanent	restored	US	single point	7.12	280.87	0.00	1.75	na	na
Pennock et al. 2010	wetland	seasonal	restored	CAN	multipoint	0.01	0.02	0.31	0.41	na	na
Pennock et al. 2010	wetland	semi-permanent	reference	CAN	multipoint	0.00	0.04	-0.13	0.30	na	na