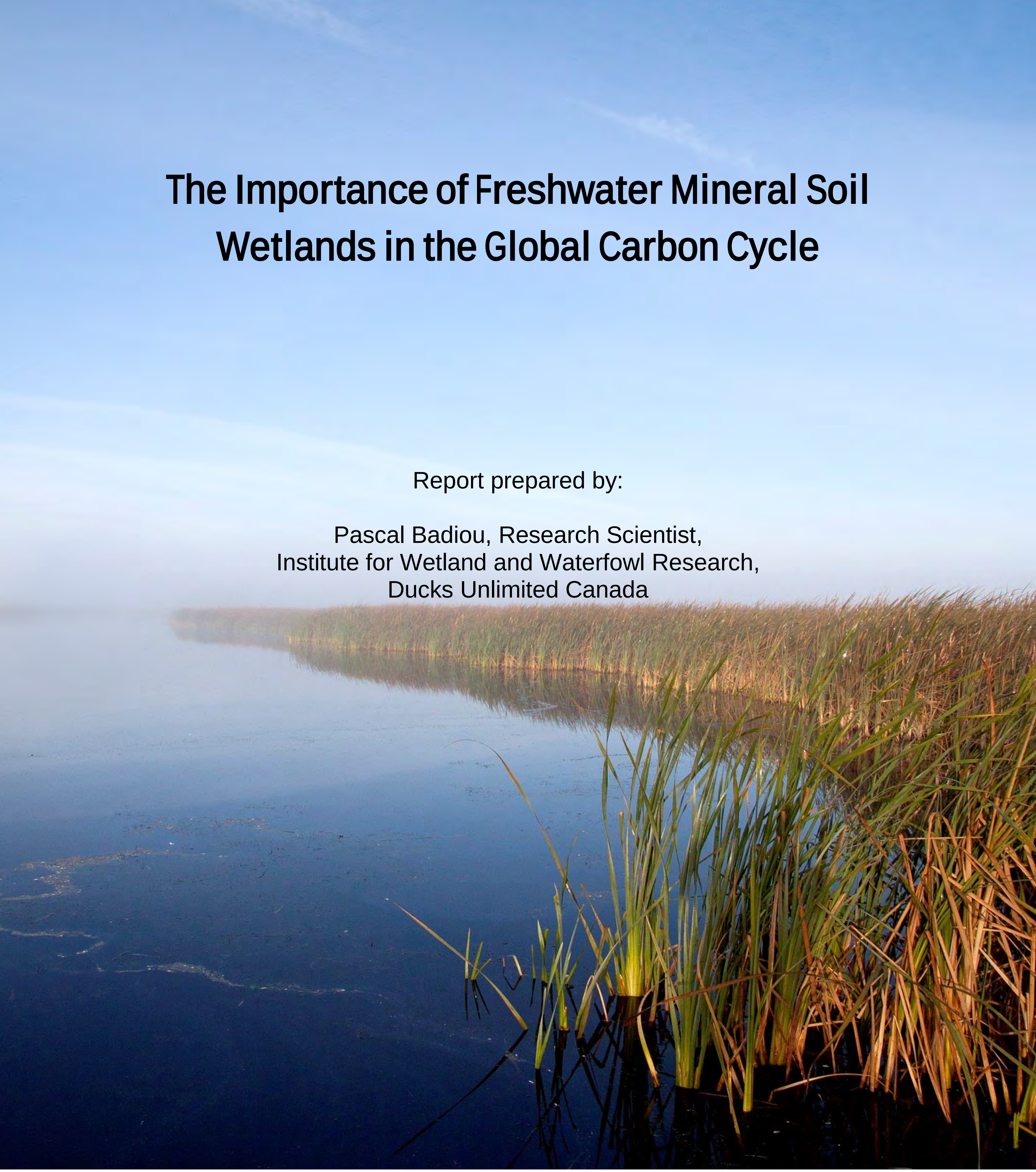

The Importance of Freshwater Mineral Soil Wetlands in the Global Carbon Cycle

Report prepared by:

Pascal Badiou, Research Scientist,
Institute for Wetland and Waterfowl Research,
Ducks Unlimited Canada

Table of Contents

INTRODUCTION

This document focuses on freshwater mineral soil wetlands (FWMSWs) and in particular on freshwater emergent marshes. Historically, FWMSWs have been the focus for conversion but are now commonly targeted for restoration and/or management in Canada.

FWMSWs account for approximately 39% of total wetland area globally, and approximately 12% of total wetland area within Canada. Peatlands account for the majority of wetland area in Canada, and are the most important wetland carbon stores both within Canada and globally. However, in terms of landscape change, an estimated 20 million ha of FWMSWs have been lost in Canada since settlement (~1800), compared to 1.4 million ha of peatlands (National Wetlands Working Group (NWWG) 1988). It is therefore likely that the conversion of FWMSWs is more significant in terms of total carbon emissions, relative to those associated with peatland conversion.

Since FWMSWs are often targeted for conversion, there is ample opportunity to investigate options and impacts for conservation and restoration, in order to deliver natural, cost-effective climate change adaptation and mitigation.

WETLANDS AND THE GLOBAL CARBON CYCLE

Why do wetlands accumulate carbon?

FWMSWs, especially freshwater marshes, are very productive ecosystems dominated by large emergent vegetation. The soils in FWMSWs are typically anoxic, and consequently, decomposition is reduced compared to aerobic environments. This results in an imbalance between the rate at which organic carbon is produced and accumulated, and the rate of decomposition. This leads to high rates of Net Ecosystem Exchange, or sequestration, of CO_2 into FWMSWs, producing a net sink for CO_2. As a result, organic carbon densities are typically 2 times higher in intact FWMSWs relative to their surrounding upland environments. Recent investigations of carbon density associated with FWMSWs in Canada have found comparable values for soil organic carbon (SOC) densities for wetlands in the Prairie Pothole Region: 205 Mg C ha^{-1} (Badiou et al. 2011) and 175.1 Mg C ha^{-1} (Bedard-Haughn et al. 2006).

Carbon sequestration in Freshwater Mineral Soil Wetlands:

Due to growing concerns regarding global climate change, research has been undertaken to determine the capacity of ecosystems such as wetlands to act as biological sinks for atmospheric CO_2. To date, much of the literature has focused on carbon sequestration within peat-forming wetlands such as bogs and fens, given their expansive extent and deep organic, carbon-rich, soils. However, there is a growing body of research that demonstrates the importance of intact and restored FWMSWs in temperate regions of the world as effective biological sinks for CO_2. It is estimated that, globally, carbon sequestration in established temperate FWMSWs averages in the range of approximately 100-250 gC m^{-2} yr^{-1} (Bernal and Mitsch, 2012; Zhang et al., 2016; Lu et al., 2017). Applying these rates to total FWMSW area in Canada, 15.9M-39.8M tonnes of carbon may be sequestered by FWMSWs annually (Bridgeham et al., 2006). This is equivalent to the emissions from 12.3M-30.8M passenger vehicles, or the electricity used by 8.6M-21.5M homes (US EPA, 2016).

Restored FWMSWs seem to be particularly proficient at sequestering carbon. Various studies have shown that restored wetlands sequester carbon at a much faster rate than intact natural wetlands in the same region. Research into restored wetlands in the prairies has shown carbon sequestration rates in the range of 270-305 gC m^{-2} yr^{-1} (Euliss et al., 2006; Badiou et al., 2011). The high productivity and low decomposition rates created by anoxic conditions in FWMSW ecosystems are the main reasons for high carbon sequestration rates.

Comparing carbon sequestration rates for natural and restored FWMSWs, with those for other land management techniques, shows the importance of FWMSWs as carbon sinks and stores. In terms of annual carbon sequestration rates, one hectare of mature trees can sequester approximately 170 gC m^{-2} yr^{-1}, similar to the sequestration rate observed for many FWMSWs (Canadian Council of Forest Ministers, 2003). Since many FWMSWs have been converted for agricultural land use, comparison of the sequestration rates of agricultural practices with FWMSWs is more pertinent. Changes in agricultural land management from conventional to no-till management is estimated to sequester 11.2 gC m^{-2} yr^{-1} in the Dry Prairie, and 16.1g C m^{-2} yr^{-1} in the Parkland ecoregions in Alberta (Alberta Agriculture and Forestry, 2012). Changing land use from cropland to grassland in the Canadian Prairies is estimated to sequester between 43-94g C m^{-2} yr^{-1} (Eagle et al., 2010). Therefore the conversion of cropland to FWMSWs results in much higher net changes in soil organic carbon stocks (270-305 gC m^{-2} yr^{-1}) compared to other changes in land use.

Wetlands and greenhouse gas (GHG) emissions: Methane and Nitrous Oxide

Although FWMSWs are important stores of carbon and CO_2 sinks, the emissions of other more potent GHGs, such as methane (CH_4) and nitrous oxide (N_2O), need to be considered to understand the overall impact of wetlands on radiative forcing (the net impact on climate change through GHG emissions and sequestration). Wetlands, particularly the riparian areas, are important to the cycling of GHGs due to their involvement in carbon and nitrogen cycles (Merbach et al. 2003).

The same high productivity and anaerobic conditions that favor the accumulation of carbon, and that supress CO_2 production in wetlands, also enhance the production and release of CH_4. As a result, FWMSWs typically have high rates of methanogenesis (methane production), and although they only occupy a small fraction of the Earth's surface area, are considered one of the most important natural sources of CH_4 emissions. This is an important fact given that CH_4 has a global warming potential 25 times more than CO_2. While FWMSWs are generally important sources of

CH_4, emission rates vary dramatically both spatially and temporally, and have been related to various hydrologic and climatological controls such as temperature, soil moisture, and degree of inundation (Crill et al. 1991, Altor and Mitsch 2008, Batson et al. 2015). Other factors such as the trophic state of a wetland, the quality of substrate, sulfate concentrations, and vegetation community also play an important role in regulating the production and release of methane from wetland systems (Pennock et al. 2010, Batson et al. 2015, Segarra et al. 2015). While increasing methane emissions are of concern for global climate change, according to Neubauer (2014) no natural wetlands older than ~250 years can be considered net sources of radiative forcing as these should be considered part of the pre-industrial baseline conditions. Furthermore, there is growing evidence of the importance of anaerobic oxidation of methane in freshwater wetland environments. A recent study by Segarra et al. (2015) found that anaerobic methane oxidation in temperate wetlands may consume up to 30 times the annual methane emissions from these systems.

Like CH_4, nitrous oxide (N_2O) is a very potent GHG with 310 times the global warming potential of CO_2. The water-saturated and anoxic environment found in FWSMWs mean that N_2O emissions are typically a minor component of the overall GHG emissions from these systems (Blais et al., 2005). However, less permanent wetlands that alternate between wet and dry cycles, such as the ephemeral and seasonal wetlands found in the Prairie Pothole Region of North America, can emit substantial amounts of N_2O (Pennock et al., 2010; Tangen et al., 2015). Short bursts of N_2O emissions seem to occur when water-filled pore space in the soil decreases from approximately 80% to 60% (Pennock et al., 2010).

Land use influences may also play an important role in determining if FWMSWs are major sources of N_2O emissions. Tangen et al. (2015) found that N_2O emissions were significantly higher in wetlands with catchments dominated by croplands when compared to those dominated by grassland, likely due to the presence of nitrogen fertilizers.

Net carbon sequestration in freshwater mineral soil wetlands and implications for radiative forcing

In order to fully understand the role of wetlands with respect to climate change, it is essential to account for both changes in carbon sequestration, and emissions of GHGs such as CH_4 and N_2O. Unfortunately, research quantifying these parameters is limited, and where they have been quantified there is some debate regarding the assumptions of the standard global warming potentials that have been applied. It is clear from research and wetland modelling that wetlands eventually become net radiative sinks through carbon sequestration (Mitsch et al., 2013; Neubauer, 2014). However, the length of time it takes for a wetland to change from a net source of GHGs to a net radiative sink, the "switchover time", is less clear.

Despite difficulties in modelling switchover times, freshwater temperate marsh type wetlands have some of the shortest switchover times and become net radiative sinks within a range of 60 to 130 years (Neubauer, 2014). The complexity associated with GHG emissions and carbon sequestration means that caution needs to be exercised when restoring or creating freshwater wetlands with the goal of mitigating atmospheric GHG concentrations. Without proper management, they may have a net warming effect on climate for decades or longer (Bridgham et al., 2014; Neubauer, 2014). However, this should not discourage accounting of the carbon sequestered, or GHGs emitted, by such projects, given the important role of wetlands in the global carbon cycle.

Wetland restoration should also be assessed against the radiative forcing of the pre-restoration land use. In agricultural environments, drained wetlands (concave and depressional landscape features) can act as hotspots for N_2O emissions (Corre et al., 1996) and surface drainage ditches can act as hotspots of methane emissions (Schrier-Uijl et al., 2011). Restoration of wetlands and the subsequent alteration of hydrology in this type of environment may substantially increase the capacity of a wetland restoration project to reduce net radiative forcing on a landscape scale, and decrease the amount of time for a given restoration project to become a net radiative sink.

While determining the value of an individual restored wetland can be challenging, wetland restoration protocols should focus on a landscape approach involving upland restoration as well as wetland restoration. There are many other forms of management that can be applied to wetlands and their surrounding landscape to reduce GHG emissions from wetlands while increasing carbon sequestration. For example, conversion of cropland to grassland in association with the creation of a wetland, can reduce wetland GHG emissions by increasing the hydroperiod (time between drying and rewetting), decreasing nutrient loading and the risk of eutrophication and N_2O emissions, and reducing the likelihood of tillage of depressional areas associated with N_2O emissions.

THE IMPACT OF WETLAND LOSS ON CARBON STORES AND GHG EMISSIONS

Drainage of FWMSWs is a common practice in the preparation of land for agriculture, grazing, and forestry (Intergovernmental Panel on Climate Change (IPCC) 2013). It is estimated that approximately 20 million ha of FWMSWs have been altered in Canada since European settlement, primarily in the agricultural regions of southern Ontario and the prairie Provinces (NWWG, 1988). A recent analysis of wetland loss in Ontario revealed that since settlement over 85% of wetlands have been converted to other uses (Ducks Unlimited Canada (DUC) 2010). The same study also found that, between 1982 and 2002, an estimated 70,854 ha of wetlands were lost in southern Ontario, equivalent to 3,543 ha yr^{-1}, indicating that wetland loss is an ongoing issue in this region. Similarly, there has been renewed efforts to drain remaining wetlands to increase agricultural production in the Canadian Prairies (Watmough and Schmoll 2007), despite evidence that 70% of prairie wetlands have been drained or altered historically (NWWG, 1988). DUC estimates that more than 500,000 ha of wetlands have been lost in the Canadian Prairies and southern Ontario since the 1950s, and that nationally ~11,800 ha are lost on an annual basis.

The ongoing drainage and conversion of wetlands for agriculture, forestry, construction, and other land uses, has serious implications for the carbon stored within these systems. Conversion of FWMSWs results in the loss of soil organic carbon by mineralizing organic matter that was previously protected under anaerobic conditions (Mitra et al. 2003). This represents a loss of carbon to the atmosphere that has been accumulated in wetlands over centuries if not millennia, not dissimilar to the burning of fossil fuels. Research has shown that wetland conversion has resulted in a loss of approximately 87-89 Mg C ha^{-1} in the Canadian Prairies, and approximately 117 Mg C ha^{-1} in southern Ontario (Badiou et al., 2011; Bedard-Haughn et al., 2006; Cebulski, 2016).

Applying the rate of wetland loss and region-specific changes in soil organic carbon density, we estimate that approximately **5.4 million tonnes of CO_2** is being reintroduced to the atmosphere annually from the Canadian Prairies and southern Ontario. This is equivalent to 7% of GHG emissions from the agriculture sector in Canada.

WETLAND MANAGEMENT: IMPLICATIONS FOR GHG EMISSIONS AND CARBON SEQUESTRATION

In terms of carbon offset markets and GHG accounting of wetland projects, the focus of many programs is on restoration and/or retention (i.e. avoided conversion). However, there are a number of different land management practices that can affect carbon sequestration and GHG emissions from wetlands.

Many wetlands are managed hydrologically with the goal of either removing excess top-water from watercourses or agricultural land during periods of high water, for example to increase land base available for cropping, or to hold additional water during droughts to maintain a preferable water table. This is important when one considers that hydrology is the main driver controlling the carbon balance and GHG emissions of wetlands (Malone et al. 2013). Wetlands that are subject to more dynamic hydrological regimes with alternate wetting and drying events, are more prone to increased CH_4 emissions than those with a more constant water level. Significant N_2O emissions are also associated with rapid drying of wetlands. Ephemeral prairie pothole wetlands are therefore much more prone to significant GHG emissions than larger, semi-permanent or permanent wetlands (Kannenberg et al. 2015; Pennock et al., 2010; Hartwig, 2006). Management to remove top waters from more permanent and semi-permanent wetlands creates similar drying and rewetting conditions experienced by ephemeral wetlands, often resulting in impartially drained wetland basins and enhanced GHG emissions. This type of management also decreases the ability of wetlands to alleviate flood risk and recharge groundwater systems. Similarly, the changing weather patterns and more extreme seasonal cycles that are being experienced, and are predicted to continue in Canada, could instigate even more dynamic hydrological cycles and greater GHG emissions from wetlands.

Coupled with hydrology, water quality can have important implications for GHG emissions from FWMSWs. When appropriate management activities are applied to wetland basins and contributing areas, landscape conditions can be managed to optimise GHG emission reductions. A number of studies have documented the inhibition of wetland CH_4 emissions in the presence of elevated SO_4 concentrations (Pennock et al. 2010, Pester et al. 2012). It may therefore be possible to target wetland conservation and restoration projects to areas with elevated SO_4 concentrations in order to reduce methane emissions and radiative switchover time.

The trophic state and nutrient concentrations in wetlands also play an important role in regulating GHG emissions. Numerous studies have documented relationships between increased phosphorus and nitrogen loading, and increases in CH_4 and N_2O emissions (Sanchez-Carrillo et al., 2010; Gonzalez-Valencia et al., 2014; Silva et al., 2016). While wetlands have a greater ability to sequester nutrients than other aquatic ecosystems, employing management activities that limit nutrient loading to wetlands can produce significant and measurable GHG benefits. This is particularly relevant in agricultural regions of Canada where a large number of wetlands now exist in cropland environments and are subject to increased nutrient loads.

Prescribed burning is another restoration and management tool which may have important implications for GHG emissions from wetlands. Data on the benefits of prescribed burning on wetland management is limited, but a study of controlled wetland burns in Florida found that the combustion products of biomass burning in wetlands exerted a soil fertilizing effect resulting in the increased CH_4 and N_2O emissions (Levine et al., 1990). However, this management activity is typically only applied to eradicate undesirable and/or invasive vegetation. Invasive plant species can often alter carbon sequestration and the ecosystem-atmosphere exchange of CH_4 and N_2O, resulting in increased GHG emissions from wetlands (Yuan et al., 2014; Xiang et al., 2015; Mueller et al., 2016). As a result, prescribed burning may be warranted to avoid the long-term GHG emissions associated with the proliferation and establishment of invasive plant species, but should not be employed as standard practice.

Lastly, there has been increasing interest in harvesting cattails from wetland environments as a potential source of biofuel (Suda et al., 2009). While this practice of paludiculture in peatland systems has been used for some time in Europe as a fuel source, application in North American FWMSWs is relatively new. Early work conducted by the International Institute for Sustainable Development (IISD) demonstrated how the harvesting of cattails can yield multiple private and public benefits, including: water quality and nutrient management benefits; biomass and bioenergy supply benefits; habitat improvement; and phosphorus recovery for fertilizer use and water quality trading (Dohan 2012).

Vegetation plays a major role in regulating GHG emissions from wetlands; the harvesting of wetland vegetation has the potential to alter ecosystem-atmosphere exchange of GHGs. There have been very few studies that have investigated the effects of wetland vegetation harvesting on GHGs. Juransinski et al., (2013) found that CH_4 emissions in wetlands where *Phragmites*, *Typha*, and *Carex* plant species were harvested were the same or lower compared to emissions from non-harvested control sites. Ongoing (as yet unpublished) research being conducted by IISD and DUC in south-western Manitoba has found that harvested wetland areas are large net sinks for CO_2 and produce less CH_4 relative to un-harvested control sites.

THE IMPACT OF WETLANDS ON REGIONAL CLIMATE

Inland waters such as lakes and wetlands are important geographical features in many parts of North America, and can comprise significant portions of regional landscapes, for example in the Great Lakes region, the Boreal region, and the Prairie Pothole Region. While wetlands play an important role in global climate as a result of their influence on the global carbon cycle and GHG emissions, it is also important to consider that the presence of wetlands on the landscape can moderate local and regional climate. Inland waters likely play an important role in regional climate regulation as a result of differences in albedo, heat capacity, topography and vegetation, and energy exchange (Bonan 1995). Such influences on local climate and climate adaptation should also be considered when determining the impacts of wetlands and wetland restoration on climate change.

There have been a number of studies that have demonstrated the importance of wetlands in regional climate regulation. A study by Krinner (2003) found that wetlands and lakes in the Boreal region had strong impacts on climate simulations, particularly in the summer, and that wetlands played a more important role relative to lakes in cooling the boreal region in summer and humidifying the atmosphere. Research has also shown negative correlations between wetland abundance and regional air temperatures, and has demonstrated that wetland conversion can reverse the cooling/humidifying effect of wetlands on regional climate, increasing maximum and minimum air temperatures. (Yunlong et al. 2011, Bai et al. 2013, Yan et al. 2015).

While restored and created FWMSWs might require upwards of a century to become net radiative sinks, they will have an immediate impact on local and regional climates through cooling and humidification of the atmosphere. This will alleviate some of the immediate regional impacts of climate change, and offset some radiative forcing associated with wetland restoration, prior to becoming net radiative sinks.

WETLAND PROTOCOLS

As a result of internationally developed carbon offset programs, there has been increased emphasis on developing guidelines and requirements for carbon offset programs and GHG emissions reporting associated with land use change. Following an invitation from the United Nations Framework Convention on Climate Change (UNFCCC) to "undertake further methodological work on wetlands, focusing on the rewetting and restoration of peatland, with a view to filling in the gaps in the *2006 IPCC Guidelines for National Greenhouse Gas Inventories"*, the IPCC developed the *2013 Supplement to the IPCC Guidelines on National Greenhouse Gas Inventories: Wetlands*. This supplement provides nation-level guidance on inventorying methods for soil organic carbon (SOC) and GHG sinks and sources from various wetland types, including FWMSWs.

The IPCC's Guidelines for National GHG Inventories identifies three tiers of methodological approaches for determining GHG emissions and removal estimates for Agriculture, Forestry and Other Land Use (AFOLU) activities. The three tiers are hierarchical in terms of complexity and accuracy; Tier 1 methodologies are the simplest and Tier 3 are the most complex. Methodologies are selected based on available information; the IPCC recommends higher tiered approaches to be used where possible to increase the accuracy GHG emission and removal estimates. The methodologies to estimate inland mineral wetland 0 – 30 cm SOC stock and stock changes are described in section 5.2.1.2 of Chapter 5 of the *2013 Supplement to the 2006 IPCC Guidelines for National GHG Inventories: Wetlands.*

The Verified Carbon Standard (VCS) has advanced carbon offsets protocols for wetlands considerably, having produced methodologies for the development of wetland retention and restoration projects. However, the initial guidance from the IPCC in 2006 largely focused on peatlands, and to date the majority of protocols have focused on these systems. While there are currently no approved methodologies for the conservation and restoration of FWMSWs, there has been interest in forwarding such a protocol.

A wetland restoration protocol was developed and submitted to the Alberta Carbon Offset System but has never been approved due to potential conflicts with Alberta's provincial wetland policy, as well as concerns regarding the science supporting the role of FWMSWs in carbon sequestration and GHG emissions. Given the focus of accepted carbon offset methodologies on large project areas, it may be more efficient to develop wetland restoration and conservation protocols that can be incorporated into applicable existing protocols, such as management of agricultural and grassland systems where FWMSWs are particularly abundant and at risk of conversion.

CO-BENEFITS ASSOCIATED WITH WETLAND RETENTION AND RESTORATION:

Wetland retention and restoration provides many additional co-benefits that traditional carbon offset methodologies do not. Importantly, FWMSWs can help mitigate some of the impacts predicted to occur as a result of climate change. For example, droughts and flooding are predicted to worsen in the future due to climate change. Wetlands, have the potential to moderate extreme drought and flood conditions and while sustaining base flows instream.[1]

In addition to regulating flows, wetlands are equally important in their role of nutrient sequestration, and mitigation of non-point source pollution at a watershed scale. Research suggests that nutrient loading in aquatic ecosystems can be an important driver of GHG flux, while the presence of submerged macrophytes can reduce CH_4 emissions (Davidson et al., 2015). It therefore stands to reason that the presence of FWMSWs, particularly in agricultural regions, can reduce GHG emissions on a landscape scale, as an integral part of a nutrient management plan. Excess nutrients sequestered by FWSMWs can limit the risk of eutrophication in open water bodies while excess nutrients are processed where abundant submerged macrophytes can limit CH_4 emissions. This is particularly important as it has been suggested that eutrophication in freshwaters may be a significant feedback mechanism (Moss et al., 2011). However, it should be stressed that wetlands do not have the ability to sequester significant excess nutrients and should be employed as part of a holistic nutrient management plan.

CONCLUSIONS AND RECOMMENDATIONS:

Wetlands play a major role in the global carbon cycle due to high productivity rates, enhanced carbon sequestration and potential for significant methane fluxes. The ability of a given wetland to affect climate change through changes in radiative forcing is related to the balance between cooling due to long-term carbon sequestration, and warming due to GHG emissions over decades (for CH_4) to centuries (for N_2O) (Neubauer and Megonigal, 2015). However, intact natural wetlands older than ~250 years, are likely net radiative sinks, and should not be considered sources of net radiative forcing as their emissions are included within the pre-industrial baseline (Neubauer 2014; Neubauer and Megonigal 2015).

However, conversion and/or degradation of intact wetland ecosystems can result in significant losses of soil organic carbon, resulting in these systems becoming net radiative sources. In many cases, the net radiative cooling effect associated with intact wetlands that may have taken centuries

---

[1] Under extreme snowmelt and precipitation conditions, wetlands can store water and reduce the speed at which water is transmitted through a watershed, thereby reducing peak flows and flood risk. Similarly wetlands have the potential to store large amounts of water and aid groundwater recharge during periods of drought.

to millennia to achieve can be reversed in a few short years. As such, ongoing conversion and degradation of wetland ecosystems should be captured in jurisdictional carbon accounting systems. Accounting for changes in carbon stores associated with wetland conversion and/or degradation should be embedded in existing carbon protocols, in particular those targeted to agricultural regions where wetland carbon stores are at risk. This would ensure that, if a proponent is being financially rewarded for sequestering carbon through a given management practice, while at the same time converting and/or degrading wetland systems, the resulting carbon credits and financial compensation are adjusted for wetland carbon losses.

Accounting for the net radiative forcing and potential direct climate change mitigation benefits associated with wetland restoration and/or creation is more challenging due to the wide range and potentially high methane emissions associated with such projects. However, numerous studies have demonstrated that restored and created wetlands are net radiative and carbon sinks. The issue is the length of time that is require for these systems to produce a net cooling effect. While this can be thousands of years for certain wetland types, FWMSWs in temperate regions demonstrate some of the highest carbon sequestration rates and shortest radiative switchover times (~20-150 years). Current land management practices should also be considered when assessing switchover times for wetland restoration or creation projects; where current land use causes net radiative forcing, switchover times should be reduced using current land use and management as the baseline.

Regardless of the amount of time required for a wetland restoration and/or creation project to produce a net radiative cooling effect based on its carbon balance, these activities have the ability to produce immediate environmental co-benefits such as water quality enhancement and mitigation against floods and droughts. Wetlands also have the ability to reduce the GHG flux of other aquatic ecosystems on a landscape scale by reducing nutrient loading. There is also increasing evidence that expansive wetland areas have a cooling and humidifying effect on regional climates. As a result, some of the early radiative forcing that may result from wetland restoration and/or creation may be offset and result in reduced radiative switchover times.

REFERENCES

Alberta Agriculture and Forestry. 2012. Quantification Protocol for Conservation Cropping. Available at: http://aep.alberta.ca/climate-change/guidelines-legislation/specified-gas-emitters-regulation/documents/ProtocolConservationCropping-Apr2012.pdf

Altor, A.E., and W.J. Mitsch. 2008. Methane and carbon dioxide dynamics in wetland mesocosms: effects of hydrology and soils. Ecological Applications 18: 1307-1320.

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

Bai, J., Q. Lu, Q. Zhao, J. Wang, and H. Ouyang. 2013. Effects of alpine wetland landscapes on regional climate on the Zoige Plateau of China. Advances in Meteorology 2013: 972430.

Batson, J., G.B. Noe, C.R. Hupp, K.W. Krauss, N.B. Rybicki, and E.R. Schenk. 2015. Soil greenhouse gas emissions and carbon budgeting in a short-hydroperiod floodplain wetland. Journal of Geophysical Research 120: 77-95.

Bedard-Haughn, A., F. Jongbloed, J. Akkerman, A. Uijl, E. de Jong, T. Yates, and D. Pennock. 2006. The effects of erosional and management history on soil organic carbon stores in Ephemeral wetlands of hummocky agricultural landscapes. Geoderma 135: 296-306.

Bernal, B. and W.J. Mitsch. 2013. Carbon sequestration in two created riverine wetlands in the Midwestern United States. Journal of Environmental Qualit 42: 1236-1244.

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

Bonan, G.B. 1995. Sensitivity of a GCM simulation to inclusion of inland water surfaces. Journal of Climate 8: 2691-2704.

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

Bridgham, S.D., T.R. Moore, C.J. Richardson, and N.T. Roulet. 2014. Errors in greenhouse forcing and soil carbon sequestration estimates in freshwater wetlands: A comment on Mitsch et al. 2013. Landscape Ecology 29: 1481-1485.

Canadian Council of Forest Ministers. 2003. Canada's forests: CO_2 sink or source. Canadian Council of Forest Ministers, Ottawa, Ontario, Canada.

Cebulski, A. 2016. Quantifying the carbon sequestration of restored wetlands in the agricultural landscape of southern Ontario. Honours Thesis, Western University, London, Ontario, Canada.

Corre, M.D., C. van Kessel, and D.J. Pennock. 1996. Landscape and seasonal patterns of nitrus oxide emissions in a semiarid region. Soil Science Society of America Journal 60: 1806-1815.

Davidson, T.A., J. Audet, J-C. Svenning, T.L. Lauridsen, M. Søndergaard, F. Landkildehus, S.E. Larsen, and E. Jeppesen. 2015. Eutrophication effects on greenhouse gas fluxes from shallow lake mesocosms override those of climate warming. Global Change Biology 21: 4449-4463.

Dohan, R., D. Roy, R. Grosshans, P. Gass, M. McCandless, and H.D. Venema. 2012. Cattail (Typha spp.) harvesting in Manitoba: A legislative and market analysis for operationalization and carbon emission offsets. IISD Report October 2012

Ducks Unlimited Canada. 2010. Southern Ontario wetland conversion analysis. Ducks Unlimited Canada, Barrie, Ontario, Canada.

Eagle, A., L. Olander, L.R. Henry, K. Haugen-Kozyra, N. Millar, and G.P. Robertson. 2012. *Greenhouse Gas Mitigation Potential of Agricultural Land Management in the United States: A Synthesis of the Literature*. Report NI R 10-04, Third Edition. Durham, NC: Nicholas Institute for Environmental Policy Solutions, Duke University.

Euliss Jr., N.H., R.A. Gleason, A. Olness, R.L. McDougal, H.R. Murkin, R.D. Robarts, R.A. Bourbonniere, and B.G. Warner. 2006. North American prairie wetlands are important nonforested land-based carbon storage sites. Science of the Total Environment 361: 179-188.

Gonzalez-Valencia, R., A. Sepulveda-Jauregui, K. Martinez-Cruz, J. Hoyos-Santillan, L. Dendooven, and F. Thalasso. 2014. Methane emissions from Mexican freshwater bodies: Correlations with water pollution. Hydrobiologia 721: 9-22.

Intergovernmental Panel on Climate Change. 2014. Supplement to the 2006 IPCC guidelines for national greenhouse gas inventories: Wetlands. T. Hiraishi, T. Krug, K. Tanabe, N. Srivastava, J. Baasansuren, M. Fukuda, and T.G. Troxler, editors. Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Intergovernmental Panel on Climate Change. 2006. 2006 IPCC guidelines for national greenhouse gas inventories. H.S. Eggleston, L. Buendia, K. Miwa, T. Ngara, and K. Tanabe, editors. Institute for Global Environmental Strategies, Kanagawa, Japan.

Jurasinski, G., A. Günther, V. Huth, S. Glatzel, and J. Couwenberg. 2013. GHG emissions from peatlands: Effects of rewetting and land use. Reeds as a Renewable Resource. Greifswald, Germany.

Kannenberg, S.A., S.T. Dunn, S.M. Ludwig, S.A. Spawn, and J.D. Schade. 2015. Patterns of potential methanogenesis along soil moisture gradients following drying and rewetting in Midwestern prairie pothole wetlands. Wetlands 35: 633-640.

Krinner, G. 2003. Impact of lakes and wetlands on boreal climate. Journal of Geophysical Research 108: 4520.

Levine, J.S., W.R. Cofer, D.I. Sebacher, R.P. Rhinehart, E.L. Winstead, S. Sebacher, C.R. Hinkle, P.A. Schmalzer, and A.M. Koller. 1990. The effects of fire on biogenic emissions of methane and nitric oxide from wetlands. Journal of Geophysical Research 95: 1853-1864.

Liu, Y., L. Sheng, and J. Liu. 2015. Impact of wetland change on local climate in semi-arid zone of northeast China. Chinese Geographical Science 25: 309-320.

Lu, W., J. Xiao, F. Liu, Y. Zhang, C. Liu, and G. Lin. 2017. Contrasting ecosystem CO_2 fluxes of inland and coastal wetlands: A meta-analysis of eddy covariance data. Global Change Biology: In Press.

Merbach, W., T. Kalettka, C. Rudat, and J. Augustin. 2002. Trace gas emissions from riparian areas of small eutrophic inland waters in northeast-Germany. Pages 235-244 *in* G. Broll, W. Merbach, and E.-M. Pfeiffer, editors. Wetlands in Central Europe. Springer, Berlin, Germany.

Mitsch, W.J. and J.G. Gosselink. 2000. The value of wetlands: Importance of scale and landscape setting. Ecological Economics 35: 25-33.

Mitsch, W.J., B. Bernal, A.M. Nahlik, Ü. Mander, L. Zhang, C.J. Anderson, S.E. Jørgensen, and H. Brix. 2013. Wetlands, carbon, and climate change. Landscape Ecology 28: 583-597.

Moss, B., S. Kosten, M. Meerhoff, R.W. Battarbee, E. Jeppesen, N. Mazzeo, K. Havens, G. Lacerot, Z. Liu, L. De Meester, H. Paerl, and M. Scheffer. 2011. Allied attack: climate change and eutrophication. Inland Waters 1: 101-105.

Mueller, P., R.N. Hager, J.E. Meschter, T.J. Mozdzer, J.A. Langley, K. Jensen, and J.P. Megonigal. 2016. Complex invader-ecosystem interactions and seasonality mediate the impact of non-native Phragmites on CH4 emissions. Biological Invasions doi 10.1007/s10530-016-1093-6.

National Wetlands Working Group. 1988. Wetlands of Canada. R. Tabacnik and J. Wilson, editors. Ecological land classification series, no. 24, Environment Canada, Ottawa, Ontario, Canada.

Neubauer, S.C. 2014. On the challenges of modeling the net radiative forcing of wetlands: Reconsidering Mitsch et al. 2013. Landscape Ecology 29: 571-577.

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

Pester, M., K.-H. Knorr, M.W. Friedrich, M. Wagner, and A. Loy. 2012. Sulfate-reducing microorganisms in wetlands – famleless actors in carbon cycling and climate change.

Frontiers in Microbiology 3: 72.

Sánchez-Carrillo, S., D.G. Angeler, M. Álvarez-Cobelas, and R. Sánchez-Andrés. 2010. Freshwater wetland eutrophication. Pages 195-210 *in* A. Ansari, S. Singh Gill, G.R. Lanza, and W. Rast, editors. Eutrophication: Causes, consequences and control. Springer, New York, New York, USA.

Schrier-Uijl, A.P., A.J. Veraart, P.A. Leffelaar, F. Berendse, and E.M. Veenendaal. 2011. Release of CO_2 and CH_4 from lakes and drainage ditches in temperate wetlands. Biogeochemsitry 102: 265-279.

Segarra, K.E.A., F. Schubotz, V. Samarkin, M.Y. Yoshinaga, K.-U. Hinrichs, and S.B. Joyce. 2015. High rates of anaerobic methane oxidation in freshwater wetlands reduce potential atmospheric methane emissions. Nature Communications 6: 7477.

Silva, J.P., T.R. Canchala, H.J. Lubberding, E.J. Peña, and H.J. Gijzen. 2016. Greenhouse gas emissions from a tropical eutrophic freshwater wetland. International Journal of Environmental, Chemical, Ecological, Geological, and Geophysical Engineering 10: 449-455.

Strachan, I.B., K.A. Nugent, S. Crombie, and M.-C. Bonneville. 2015. Carbon dioxide and methane exchange at a cool-temperate freshwater marsh. Environmental Research Letters 10: 065006.

Suda, K., A. Shahbazi, and Y. Li. 2009. The feasibility of using cattails from constructed wetlands to produce bioethanol. G.A. Uzochukwu et al. (eds), Proceedings of the 2007 National Conference on Environmental Science and Technology doi 10.1007/978-0-387-88483-7_2

Tangen, B.A., R.G. Fionocchiaro, and R.A. Gleason. 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: 391-409.

US EPA. 2016. Greenhouse Gas Equivalencies Calculator. Available at: https://www.epa.gov/energy/greenhouse-gas-equivalencies-calculator

Watmough, M.D. and M.J. Schmoll. 2007. Environment Canada's Prairie & Northern Region habitat monitoring program phase II: Recent habitat trends in the Prairie Habitat Joint Venture. Canadian Wildlife Service technical report series number 493. Environment Canada, Edmonton, Alberta, Canada.

West, T.O., and W.M. Post. 2002. Soil Organic Carbon Sequestration Rates by Tillage and Crop Rotation: A Global Data Analysis. Soil Sci. Soc. Am. J. 66:1930–1946

Xiang, J., D. Liu, W. Ding, J. Yuan, and Y. Lin. 2015. Invasion chronosequence of *Spartina* *alterniflora* on methane emission and organic carbon sequestration in a coastal salt marsh. Atmospheric Environment 112: 72-80.

Yuan, J., W. Ding, D. Liu, H. Kang, C. Freeman, J. Xiang, and Y. Lin. 2014. Exotic *Spartina alterniflora* invasion alters ecosystem-atmosphere exchange of CH_4 and N_2O and carbon sequestration in a coastal slat marsh in China. Global Change Biology doi:10.1111/gcb.12797.

Yunlong, Y., Y. Hongxian, L. Xianguo, and W. Lei. 2011. The impact of wetland cultivation on the temperature based on remote sensing – a case study in Naoli watershed of Sanjiang Plain, northeast China. Physics Procedia 22: 622-626.

Zhang, Z., N.E. Zimmermann, J.O. Kaplan, and B. Poulter. 2016. Modeling spatiotemporal dynamics of global wetlands: Comprehensive evaluation of a new sub-grid TOPMODEL parameterization and uncertainties. Biogeosciences 13: 1387-1408.