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PEAT VOLUME LOSS IN THE DITCHES OF NORTHERN MINNESOTA

Liam Krause

Michigan Technological University, lmkrause@mtu.edu

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PEAT VOLUME LOSS IN THE DITCHES OF NORTHERN MINNESOTA

By

Liam Krause

A REPORT

Submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

In Forest Ecology and Management

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This report has been approved in partial fulfillment of the requirements for the Degree of
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College of Forest Resources and Environmental Science

Report Co-Advisor: *Evan Kane*

Report Co-Advisor: *Erik Lilleskov*

Committee Member: *Randall Kolka*

Committee Member: *Rodney Chimner*

College Dean: *Andrew Storer*

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Abstract

Peatlands play an important role in carbon (C) storage and are estimated to contain 30% of global soil C, despite occupying only 3% of global land area. Historic management of peatlands has led to widespread degradation and loss of important ecosystem services including C- and fresh water storage. Legacy drainage features in the peatlands of northern Minnesota were studied to assess the volume of peat that has been lost in the ~100 years since drainage. Using high-resolution Light Detection and Ranging (LiDAR) data, we measured elevation changes along the margins of legacy ditches to model pre-ditch surface areas, which were used to calculate peat volume loss. We established relationships between volume loss and site characteristics from existing Geographic Information Systems (GIS) datasets and used those relationships to scale volume loss to the length of peatland ditches in northern Minnesota. It is estimated that 165.3 ± 8.6 million m^3 of peat have been lost throughout peatland ditches that extend almost 4,000 km. Peat loss on the upslope side of the ditch was significantly less than peat loss on the downslope side of the ditch ($P < 0.001$) and mean width of the entire ditch effect zone was 333 ± 8.32 m. Using our volume loss estimates, literature estimates of oxidation, and mean bulk density and peat C% values from Minnesota peatlands, this volume loss represents a total historic loss of 3.847 ± 0.364 Tg C. Assuming a constant oxidation rate during the 100 years since drainage, euic and dysic peatlands within the ditch effect zone lose 0.26 ± 0.08 and 0.40 ± 0.13 Mg C $\text{ha}^{-1} \text{yr}^{-1}$, respectively. Our framework can be used as a decision support tool to guide preliminary management decisions with the objective of protecting C on natural Minnesota landscapes.

1 Introduction

Land management with a focus on the retention of carbon (C) in natural landscapes is widely recognized for its potential to play an important role in greenhouse gas emissions (GHG) mitigation. A number of “Natural Climate Solutions” (NCS) (Griscom et al., 2017) have been identified as potential approaches land managers can adopt to support GHG emissions reduction goals. Among these are the avoidance of impacts on peatland ecosystems and, where disturbance has occurred, active peatland restoration (Griscom et al., 2017).

Global inventories suggest that peatland ecosystems represent only 3% of terrestrial land area (Xu et al., 2018), but store 30% of the organic C found in global soils (Page and Baird, 2016). However, historic and continued peatland drainage has impacted over 65 million hectares worldwide (Kaat and Joosten, 2008). Peatland drainage activities precipitate changes in the biogeochemical processes that regulate long-term C storage (Drexler et al., 2009), shifting peatland ecosystems from net C sinks to net C sources. Surface drainage ditches in peatlands have been widely used as a means to facilitate agricultural and agroforestry production in saturated soils, but have altered the hydrology, carbon and nutrient dynamics, and physical structure of the landscapes surrounding drainage ditches. Without intervention, it is estimated that by 2100, CO₂ emissions from degraded global peatlands will account for 12-41% of the remaining global CO₂ emissions budget required to keep global warming under the 1.5 – 2° C threshold (Leifeld et al, 2019). In Minnesota, saturated organic soils within the state’s c. 24,000 km²

peatland area (MNDNR, 1984) have experienced extensive ditching and degradation following drainage.

Systematic drainage in Minnesota began with the state's inception and reached a zenith between 1900 and 1930 (Wilson, 2016). After a second spike following World War II, drainage activities in Minnesota began to decline, precipitated by a growing recognition of the ecological importance of wetland areas and federal and state policies (Wilson, 2016).

Today, although the initiation of new peatland drainage is infrequent in Minnesota, the legacy of historic drainage efforts during the 19th and 20th-centuries can still be seen in a gridded ditch network that patterns many of the state's northern counties. It is estimated that over 33,000 km of surface drainage ditches and channelized waterways have been installed across the state (Hanson, 1987) in addition to an unknown number of privately installed drainage features.

While ditching projects of the 20th century in Minnesota's peatlands largely failed to facilitate agricultural production (Bradof, 1992), they have a continued effect on the structure and functions of the peatland landscape (Gorham and Wright, 1979). Legacy ditches alter peatland hydrology and lower water tables, increasing air-filled pore-space in the upper strata of the peat matrix, in turn promoting rapid decomposition rates of the organic peat material with the onset of aerobic microbial processing (Clymo, 1983). The removal of water from the pores in the peat matrix also initiates compaction associated with the decreased buoyancy of the acrotelm (Hooijer et al., 2012). The cumulative effect

of the chemical and physical changes results in land subsidence in the areas adjacent to drainage features.

Given wide variations in the structure and function among peatland types, it is not surprising that their response to ditching would be similarly varied. The response of a peatland to drainage is influenced by peatland soil structure and the hydraulic conductivity of the peat matrix (Boelter, 1972), peatland slope (Stewart and Lance, 1991), degree of humification of peat material (Boelter, 1968), ditch orientation, and climatic conditions (Braekke, 1983). Among the ways in which peatland ecosystems, and thus their response to drainage, can differ is the extent to which they are hydrologically connected to groundwater sources (Moore and Bellamy, 1974). Ombrotrophic bogs are hydrologically isolated from groundwater and receive the majority of their moisture and mineral inputs from precipitation. Their soils are acidic and contain poorly-humified peat, supporting vegetation communities dominated by *Sphagnum* mosses, ericaceous shrubs, and *Picea mariana*. On the other end of the nutrient gradient are minerotrophic fens, which are characterized by hydrologic connectivity with groundwater and are thus more nutrient rich and less acidic than bogs. These peatlands tend to support higher productivity graminoid plant communities and contain more highly humified peat material.

The bulk density values and degree of peat humification characteristic of the two broad peatland types are highly influential with regards to potential subsidence. Bog peat, being comprised primarily of recalcitrant *Sphagnum* material, is less-well decomposed and generally has a lower bulk density and larger pore spaces than that from fen peatlands

(Minkinen and Laine; 1998, Bridgham et al., 1998; Hill et al., 2016). The mechanical settling of the peat during subsidence will be greater in a matrix that is composed of larger pore sizes. Additionally, lower bulk density permits greater saturated hydraulic conductivity (Rezanezhad et al., 2016). As such, ditch effectiveness increases with the increase in water mobility through the large pore sizes in surface peat (Boelter, 1972). The slope and position of the ditch on the landscape can compound ditch effectiveness by capturing upslope runoff, isolating peatland areas on the downslope side of the ditch from water inputs.

Our goal was to assess the historical impact of legacy drainage ditches on the peatlands of northern Minnesota. An increased understanding of the distribution of degraded peatlands in the state of Minnesota will help land managers make decisions regarding the protection and restoration of these important ecosystems. Our subsequent investigations into the peatland characteristics that help predict volume and C loss will further assist in the prioritization of restoration activities, especially within the scope of NCS strategies.

We hypothesized that: 1) humified peat, as classified through the Soil Survey Geographic database (SSURGO) (SSURGO, 2019) taxonomic suborder and soil carbonate reaction classifications, will reveal a less pronounced response to ditching than acidic, fibric peat given an inherently higher bulk density in minerotrophic sapric peat; 2) the downslope side of a ditch would experience more loss than the upslope side of a ditch due to the diversion of water fed from the upslope to the downslope side; 3) volume change will exhibit a positive relationship with slope.

To accomplish our goal, we estimated the volume of peatland loss attributable to the ditch network, which can be measured using high resolution Light-Detection and Ranging (LiDAR) digital elevation models (DEMs) (MNDNR, 2017). Lacking historical datasets to describe pre-ditch elevation levels, it was necessary to create a model to estimate the pre-ditch surface level. This we did using two independent surface modeling approaches. We calculated volume change by finding the difference between modeled surfaces and real elevation data from 1 m bare-earth DEMs. To test the hypotheses that volume change is related to peatland characteristics, we performed site level analyses of the predictors of ditch impacts on peat volume loss, and used GIS to estimate the impact this had on peatlands in all of northern Minnesota.

2 Materials and Methods

2.1 Study Area

The study was conducted in two ecological provinces within northern Minnesota, the Laurentian Mixed Forest Province and the Tallgrass Aspen Parklands Province (MNDNR, 2003; MNDNR 2005a). The Laurentian Mixed Forest covers over 93,100 km² of northeastern Minnesota and is characterized by mixed deciduous and coniferous forests, glacial deposits, and large areas of poorly drained patterned peatland. Mean annual precipitation (MAP) ranges from 53-81 cm across a west to east gradient, while from north to south, mean annual temperatures (MAT) range from 1°C to 4°C (MNDNR, 2020a). The Tallgrass Aspen Parklands Province is characterized by much more extreme temperatures, reaching extreme lows down to -40°C and spans a 12,100 km² extent of northwestern Minnesota. The flat plains and poorly drained soils in this Province left by Glacial Lake Agassiz form an ecotone between the arid grasslands to the west and the more humid Laurentian Mixed Forest to the east. Mean annual precipitation is between 51-56 cm (MNDNR, 2020b).

In our analysis, we excluded two ecological provinces within the state, the Prairie Parklands and Eastern Broadleaf Forest Provinces. Whereas peatlands cover c. 21% of the combined area of the Laurentian Mixed Forest and the Tallgrass Aspen Parklands provinces, development and agricultural practices following European settlement have significantly reduced the area of peat-forming ecosystems in the Prairie Parklands and Eastern Broadleaf Provinces (MNDNR, 2005a, MNDNR, 2005b), and the residual peat areas are generally small and fragmented, comprising less than 2% of the updated

Minnesota National Wetlands Inventory (NWI)-classified peatlands within the state (MNDNR, 2019). The discontinuous areas of the ditched peat-forming ecosystems in the two excluded provinces precluded the application of our analysis, which requires 500 m of continuous peat surface on either side of a ditch.

Peat depth varies across the study area, with depths of 40 cm to over 7 m having been recorded throughout the state (MNDNR, 2007). Ditching in these areas occurred primarily during the early 20th century. Most of the ditches were dug to depths of 1-2 m (Averell and McGrew, 1929). Widths at the top of the ditches generally ranged from 3.5-6 m, while the width at the bottom of the channel was 1-3 m (Averell and McGrew, 1929). Ditch spacing varied by location, but generally adopted a gridded configuration and was often found to be approximately 1600 m by 1600 m or 1600 m by 3200 m (Averell and McGrew, 1929; MNDNR, 2016). General practice at the time of installation was to place the excavation spoils on the downflow side of the ditch to facilitate the natural flow of upslope water into the ditch, although this was not a universal practice (Averell and McGrew, 1929). Most peatland ditches in northern Minnesota have not been maintained regularly since installation (Bradof, 1992).

2.2 Site Selection

All geospatial analyses were conducted in Arc Pro 2.5.1 and ArcGIS Desktop 10.8.1. To generate an estimate of peatland distribution within the two ecological provinces that were included in our study, we used data from the updated Minnesota NWI and the SSURGO database. From SSURGO, we extracted all polygons in which at least 85% of

the polygon area was classified with the taxonomic soil order Histosol (classified as a soil with a surface organic layer depth over 40cm (Kolka et al., 2016)), which represented 67% of the Histosol-classified area within the study area. In our analysis, we assumed that all of this Histosol class that was also classified as wetlands from the NWI could reasonably be classified as peatlands, irrespective of whether peat was indicated in that NWI wetland classification. The intersection of the SSURGO and NWI layers included 86% (12,143 km²) of all NWI features classified with coding for peatlands in the study area. All data used in this analysis were projected to the NAD 83 UTM zone 15N.

We used the Minnesota Buffer Protection layer (MNDNR, 2016) to delineate the locations of ditches in the Laurentian Mixed Forest and the Tallgrass Aspen Parkland provinces. To exclude features such as channelized streams and other altered natural watercourses, we extracted all features with the descriptor "Public Ditch". We calculated an intersection of the Public Ditch features with the peatland layer, as described above, for a total estimate of public ditches that exist within peatlands in the study area.

A total length of 114 km of ditch segments East of Sax-Zim Bog was manually added to this layer to account for an area of ditches that were recently restored (Myers, 2015) and subsequently removed from the DNR ditch layer. This was done to include ditch volume loss that could be measured using the pre-restoration LiDAR data we used for this analysis.

We used the MNDNR NWI Wetland Finder (Kloiber, et al., 2019) and the MNDNR topography tool, MN TOPO (MNDNR, 2014), to examine summer imagery and contour

lines across the ditch network to select 71 sites based on visual estimations of the surrounding continuity of the peatland. This was done to avoid the influence of topographic variation caused by interspersed upland features on the model. Of these 71 sites, 69 were retained for analysis, with two of the original 71 identified sites dropped due to bad edge-matching between DEM patches. Fourteen are control sites with no ditches present (henceforth referred to as “control”), 33 are single ditch sites (“across”), and 22 are sites in which two ditches intersect (“4-corner”).

We accessed LiDAR elevation data from the MnGeo FTP server. Statewide LiDAR data were collected at a 1 m horizontal resolution between 2005 and 2012 by MNDNR and Woolpert, Inc. (MNDNR, 2017).

For each site identified, we established a site center point. In the case of across site types, site center points were established on the centerline of a single ditch, avoiding areas with adjacent upland features. Four-corner site center points were placed in the middle of the intersection of two intersecting ditches. Control site center points were placed in unditched peatland areas.

Our analysis utilized two separate measures of elevation change along the ditch network. The first (transect) approach uses 1 m LiDAR elevation data to model pre-ditch surfaces using the elevation profile of a 1000 m linear feature drawn perpendicular to the ditch. The second (surface trend) approach uses 1 m LiDAR elevation data to model a surface trend over an area of interest (AOI) and was used as a validation tool to assess the predictive power of the scaled transect approach.

For the transect approach in the across sites, four 1000 m transects were established perpendicular to the single ditch. Where possible, the two transects on either extreme end were placed midway between the site center point and a ditch junction or ditch endpoint. The inner across site transects were evenly positioned about the site center point between the two end transects. This spacing was generally approximately 450 m. Where topographic variation did not permit uniform spacing, placement was adjusted to avoid upland features while attempting to maintain a uniform spacing. In the 4-corner sites, four 1000 m transects were established on each intersecting ditch, two on each of the four arms of the ditch intersection, for a total of eight transects per 4-corner site. Each of these eight transects was placed perpendicular to the respective ditch segment. The four outermost transects were placed approximately at the midpoint between the site center point in the ditch intersection and the adjacent ditch intersection midpoint. The four innermost transects were located midway between the site center point and the outer transect. For both the across and 4-corner transects, the transect midpoint was located at the centerline of the ditch, with 500 m extending out perpendicular to the centerline in both directions. Because some sites were immediately adjacent to each other in a gridded ditch formation, 14 transects found at the midpoints between two sites were not duplicated and are shared between sites.

Because the surface trend analysis required a larger area of continuous landscape uniformity than does the transect approach, it was not possible to perform this process on all 69 sites. We therefore conducted the surface trend analysis on 38 sites, 14 of which were control sites for comparison with the transect approach.

AOI sizes for each site were determined by the distance between the center point location of each site and the midpoint between the site center point and any adjacent ditch intersections. It was necessary to restrict the size of the AOI to isolate the effect of a single ditch while excluding the potential elevation change generated by neighboring ditches. If distance between ditch intersections was at least 1500 m, 1500 x 1500 m was used as the default site dimension, otherwise sizes were reduced to 1000 x 1000 m. Twelve sites were assigned a size of 1000 x 1000 m, while the remaining 57 sites were assigned 1500 x 1500 m.

2.3 Analysis of Elevation Change

2.3.1 Transect Approach

Using 3D Analyst tools and 1 m LiDAR data in ArcGIS, an elevation profile was created for each transect and the data were exported to an Excel file. For each transect, we created a scatterplot of elevation vs. distance. We inspected each elevation profile to determine a point on either side of the ditch midpoint that deviated from the trend of the surrounding landscape. The area between the two deviation points is referred to as the ditch effect zone. Using only the elevation data from beyond the ditch effect zone, we tested linear, second-order, and third-order polynomial fits of a line to model the hypothesized pre-ditch elevation above the ditch effect zone.

Using the regression equation output from the modeled fit of the line, we found the predicted elevation value for all points within the ditch effect zone. To calculate total volume change along the length of a transect, we subtracted the observed elevation value

from the predicted elevation value. These residuals were summed for a total volume change estimate for each transect, which represent the total volume change given one meter of ditch length. Given the similarity between the values measured with the three tests (Table 1), the dominance of concave or convex land surfaces (e.g., Fig. 1), and the convention of using the lowest-order polynomial fit to limit bias, we ultimately chose the second-order polynomial fit of a line to estimate all volume change with the transect method.

Figure 1. Transect approach. Four 1000 m transects are centered perpendicular to each ditch within a site AOI. From 1 m elevation data, elevation profiles of the ditch were created. A second-order fit of a line is fit to the elevation points from the undisturbed distal segments (orange) of the transect. Using the line of best fit, volume change is calculated by subtracting the real elevation data of the ditch effect zone (blue) from the elevation of the modelled surface. (Graphic created in Arc Pro 2.5.1 and Microsoft Excel. Elevation data was obtained from the Minnesota Geospatial Commons and manipulated. Composite aerial imagery and ditch layers obtained from the Minnesota Geospatial Commons with full attribution in Appendix A.)

2.3.2 Surface Trend Approach

The surface trend analysis uses elevation data derived from random point sampling of the real DEM to create a new surface DEM. For each of the 38 sites, 200 randomly generated points were used to sample the real elevation data from a 1 m LiDAR DEM. To exclude the ditch effect zone in the 4-corner sites, random point sampling in 1000 x 1000 m AOI sites was constrained to four areas 250 x 250 m, one in each of the corners. From each 250 m by 250 m corner location, 50 points were randomly sampled. Four-corner sites with AOIs of 1500 x 1500 m were similarly sampled, using 50 points in four areas 450 x 450 m in each corner. Point sampling of the DEMs for the across site type, all of which had an AOI of 1500 x 1500 m, was constrained to two 450 x 1500 m areas beyond the ditch effect zone, one on either side of the ditch. Each area was sampled using 100 randomly generated points. Using this method, points were randomly selected ten separate times using the Generate Random Points tool in ArcGIS and the resulting 10 DEMs were averaged to provide a better estimate of the pre-ditch surface. Both the 4-corner and the across trend surface processes were tested in control sites.

Once each of the 38 sites was fitted with a pre-ditch surface using this approach, we calculated total volume change using the cut-fill analysis tool in ArcGIS. This tool subtracts the real elevation data values from the elevation values from a generated DEM surface and returns a total volume difference. Given the similarity between the values measured using the three fits of a surface (Table 1), the volume change values we used for the surface trend approach were ultimately derived from the third-order polynomial fit

of a plane based on review of the fits indicating that a third-order fit would be more representative of the curved surface of each AOI than would a linear or quadratic fit.

2.4 Analysis of Wetland Type

To assess any relationship between total volume change along the elevation profile transects and wetland type, we used existing GIS layers describing wetland type, soil characteristics, and ditch stream order. To assign Cowardin wetland classes to each transect, we intersected transects with the updated Minnesota NWI. We assigned to each transect the Cowardin "class" that intersected with >50% of the length beyond the ditch effect zone (>150 m on either side of the ditch). These were either FO, SS, or EM representing a forested, scrub/shrub or emergent class in a Palustrine system, respectively (Kloiber, et al., 2019).

We conducted a similar process to intersect each transect with the SSURGO to identify the soil taxonomic suborder and the carbonate reaction class. Each transect was assigned the taxonomic suborder and carbonate reaction class (SSURGO, 2019) that intersected with >50% of the length beyond the ditch effect zone. The Histosol taxonomic suborders found in the study area describe a gradient of organic material humification, i.e., sapric > hemic > fibric. The carbonate reaction class relates to the pH level within a soil polygon, dysic (< 4.5) and euic ($\geq$ 4.5). These classifications were used as broad estimates of ombrotrophic and oligotrophic (dysic) vs. more minerotrophic (euic) peat types.

Additionally, we joined the Strahler stream order attributes as an ordinal variable to each transect based on the stream order value at the point the transect was coincident with the

ditch, using data from the Stream Routes with Strahler stream order dataset (MNDNR, 2013).

2.5 Analysis of Climatic and Slope Attributes

At each transect center point, we extracted mean annual temperature and mean annual precipitation data from the PRISM dataset (PRISM, 2019). These data were accessed at a spatial resolution of approximately 800m. Slope characteristics at the midpoint of each transect were extracted using both 10 m and 30 m elevation data.

2.6 Statistical Analysis

We performed all data analyses in the statistical programming environment R (R Core Team, 2020). To determine which factors from the transect-level data were predictive of peat volume loss and could be used to scale to the map level, we implemented multi-level unbiased recursive partitioning (Hothorn et al., 2006, Fokkema, et al., 2018) through the R package glmertree (Fokkema et al., 2020) to create a generalized linear mixed-model (GLMM) tree of volume change of peat. Initial examinations of residual plots indicated some heteroskedasticity within the data, prompting us to use a square root transformation of the volume loss response variable measured from the second order polynomial transects.

This method allows for the inclusion of a random effects term and is not prone to continuous variable selection bias as is the case with other regression tree approaches (Hothorn et al., 2006). The use of statistical significance testing for variable selection limits the likelihood of an overfit tree model (Hothorn et al., 2006) and the methods and

resulting tree model are not obscured by the “black box” of ensemble learning methods like random forests (Fokkema et al., 2020).

In our GLMM tree, we used a restricted maximum likelihood estimation of a random intercept for the transect site and of the following fixed effect parameters: soil taxonomic suborder, carbonate reaction, NWI Cowardin class, Strahler stream order value, slope degree, and PRISM mean annual precipitation and mean annual temperature. Variable selection parameters were set at $\alpha=0.05$ with Bonferroni-corrected p values and a minimum node size of 40 observations. This conservative method prevented against potential overfits of the GLMM tree model.

We assessed the predictive power of the tree model using 10-fold cross validation (Hastie et al., 2009). This process splits the dataset into 10 bins and predicts a response for each bin given its exclusion using the predictor variables of the remaining 9 bins.

We further investigated the predictive capabilities of the GLMM tree model by classifying all ditches within the entire study area, including AOIs, using the three predictor variables from the terminal nodes of the GLMM tree (Figure 3, Table 3). SSURGO carbonate reaction class was applied to ditch segments by first generating 1000 m transects orthogonal to the ditch at 5 m intervals along its entire length. Each of these transects was assigned the majority reaction class that intersected with the segment of the transect beyond the ditch effect zone (150 m on either side of the ditch). The reaction class for each transect was used to determine the reaction class assigned to its respective 5 m ditch segment. Slope derived from 10 m elevation data and MAP data were

intersected with the location of the ditch. We then compared our volume loss predictions from GLMM tree ditch classifications with the AOI volume loss observations from surface trend analysis. Within each AOI, we multiplied the length of each GLMM tree ditch classification by the median volume loss value reflected in each classification's respective terminal node. We summed the volume loss values returned by this approach within each site AOI for a predicted areal estimate of volume change and regressed those values against volume change observed with the surface trend analysis.

The same ditch classification approach described above was adopted to scale volume change estimates to the entire study area. A map that includes the peatland area and GLMM tree-classified ditches can be found in the supplemental material.

To estimate C loss represented by volume loss, we assumed the percent of subsidence attributable to oxidation was $51\% \pm 7.83$ (mean $\pm$ se), using an estimate derived from an equation predicting relative contribution of oxidation to subsidence from years since drainage, described in Pronger et al. (2014). We used two different datasets to obtain average bulk density values for both euic and dysic peat types, both collected from various locations in Minnesota. The first dataset contains averaged peatland bulk density values from within the top 25 cm of peat reported in Bridgham et al. (1998), from which we excluded the observations of one mineral site type. From the second dataset, we averaged peatland bulk density values from the 35-50 cm depth collected as part of a statewide peat inventory from Minnesota peatlands (MNDNR, 2007), excluding samples that were missing data or were taken within 200 m of a ditch. Observations from both datasets were separated by pH value into dysic and euic classes and the bulk density

values were averaged for $0.09 \pm 0.003 \text{ g cm}^{-3}$ and $0.13 \pm 0.004 \text{ g cm}^{-3}$ for dysic and euic, respectively. These estimates are similar to other Minnesota peat bulk density measurements described in the literature (Boelter, 1968). Total volume loss for each ditch classification based on GLMM tree nodes was then multiplied by 0.51 ± 0.078 to represent the estimated volume loss attributable to oxidation. Estimated dry mass loss was derived from the resulting value, by multiplying the average bulk density value for dysic and euic, respectively. For a final calculation of C loss, we found the estimated C content of the peat dry mass by multiplying the total dry mass value by $0.426 \pm .004$ and $.424 \pm 0.016$ for dysic and euic, respectively (Bridgham et al., 1998).

To estimate C losses per hectare per year, we multiplied total ditch length per node by average ditch effect width to calculate the area of disturbed peatlands and assumed a period of 100 years of drainage. We then divided the total historic C loss values as described above by disturbed area and total elapsed time.

3 Results

3.1 Peatland Area and Ditch Length

Using our assumed peatland area based on coincidence between SSURGO Histosols and NWI layers, the total peatland area in the two ecological provinces is 22,105 km². Subdivided by soil taxonomic suborder, 445 km² were classified as fibrists, 12,382 km² were classified as hemists, and 9,278 km² were classified as saprists. Classifying by SSURGO carbonate reaction class (dysic < 4.5 pH, euic ≥ 4.5 pH), 6,671 km² were classified as dysic and 15,254 km² were classified as euic (180 km², or 1% of the peatland area, had no designation). Ninety-eight percent of the peatland area was classified in the Minnesota NWI by three Cowardin classes: forested=9,273 km², scrub/shrub=8,532 km², and persistent emergent=3,963 km². Total ditch length within the study area was 3,948 km, with the majority of ditches occurring in the larger Laurentian Mixed Forest province.

3.2 Observed Volume Change

3.2.1 Transect Approach

Mean volume change across all ditched peat transects measured using a second-order polynomial fit of a line (n=294) was 59.80 ± 2.95 m³ per meter of ditch length. Mean volume change from control sites was 1.16 ± 1.23 m³ per transect (n=56). Mean R² of the line fitted to elevation data from the undisturbed segments of the transect was 0.78 ± 0.01 m³. This value is slightly depressed given that so many of our sites were at near-zero surface slope. Volume loss values estimated by the three best fit methods were

comparable (Table 1). In support of hypothesis 2, ditch-effect width and volume loss were both larger on the downslope side of the ditch ($P<.001$; Table 2). Mean width of the entire ditch effect zone was 333 ± 8.32 m.

Table 1. Mean volume change per m ditch length for both transect and surface trend methods using linear, second- and third-order polynomial fits.

Fit Method	Transect (m ³) n=294	Surface Trend (m ³) n=24
Linear	59.69 ± 3.09	62.50 ± 9.17
Second-order polynomial	59.80 ± 2.95	56.47 ± 7.06
Third-order polynomial	58.93 ± 2.94	56.46 ± 7.02

Table 2. Volume change and ditch effect width $\pm$ standard error measured on both the upslope and downslope side of the ditch. Paired t-tests between both upslope and downslope volume change were both significant ($P<0.001$).

Characteristic	Mean $\pm$ se (m ³)	Min (m ³)	Max (m ³)
Downslope ditch effect width (m)	190.88 ± 5.85	9	410.59
Upslope ditch effect width (m)	142.15 ± 4.90	2	411.37
Downslope volume change (m ³)	34.18 ± 1.85	-13.16	212.75
Upslope volume change (m ³)	25.78 ± 1.48	-22.21	142.72

3.2.2 Surface Trend Approach

Using the surface trend approach, mean volume loss in control sites, standardized by AOI side length for direct comparison to ditched site AOI volume loss, was $2.48 \pm 2.30 \text{ m}^3$ per AOI (n=14). This is equal to $0.001 \pm 0.002 \text{ m}^3 \text{ m}^{-2}$ in each control AOI.

Volume loss was significantly greater in 4-corner sites than in across sites ($P=0.001$). Standardizing surface trend analysis volume change measurements by meter ditch length within the AOI, mean per-meter ditch length volume loss was $56.46 \pm 7.02 \text{ m}^3$ (Table 1). While comparable to the volume loss per meter ditch length measured with the transect method, there was a significant difference in volume loss per meter ditch between the two methods ($P=0.005$) (Table 1).

Assessing the relationship between volume change measured using the surface trend approach and the transect approach, we found that, while the transect method had a slight tendency to under-predict volume change relative to the surface trend method, there was a positive relationship between the two measured values ($R^2=0.70$) (Figure 4a).

3.3 Volume Change Relationship to Site Characteristics

Soil carbonate reaction class and soil taxonomic suborder from the SSURGO dataset and the Cowardin class from the updated Minnesota NWI dataset all exhibited significant relationships with volume change (Figure 2). As expected, a distinct gradient can be seen in the response to soil taxonomic suborder, with mean volume loss increasing with decreasing initial soil humification. The more acidic, lower bulk density ombrotrophic and oligotrophic *Sphagnum* peatland types with the dysic designation exhibited more loss

than did their higher pH minerotrophic counterparts. Forested wetland types exhibited higher volume losses than did either open emergent or scrub shrub classes.

Of the continuous predictor variables tested, two were positively correlated with volume loss: ditch slope degree at a 10 m ($R^2=0.15$), and MAP ($R^2=0.15$). Our initial analysis of slope utilized elevation data collected at a 30 m horizontal resolution, which was significant at $\alpha=0.05$, but explained very little of the variability in volume change ($R^2=0.01$). The relationship improved when we increased the resolution of the elevation data to 10 m ($R^2=0.15$, $P<0.001$). MAT was also significant but explained very little of the variability in volume change ($R^2=0.03$). Strahler stream order was not significant at $\alpha=0.05$.

Figure 2. Volume change per meter ditch length by soil and wetland characteristics (N=294). EM1 = persistent emergent, FO = forested, SS = scrub/shrub.

3.4 Volume Change Predictions

The recursive partitioning analysis selected one categorical variable and two continuous variables based on statistical significance tests (Figure 3). Soil carbonate reaction class

was the most significant partitioning variable ($P < 0.001$), followed by slope degree using the 10m resolution elevation data ($P < 0.001$). At both nodes in the tree where slope was the variable selected, the higher slope node showed higher volume loss. A final split of node 3 was selected using MAP as a partitioning variable ($P < 0.045$). Volume loss was higher in the high precipitation node than the low precipitation node. The resulting tree displays the volume loss per meter of ditch length based on transect-level characteristics, classified into five terminal nodes (Figure 3).

Figure 3. Volume change values classified by site characteristics as determined using multi-level unbiased recursive partitioning. The y-axis in each terminal node represents the volume change in $\text{m}^3 \text{m}^{-1}$ of ditch length (also see Table 3). Carbonate reaction class is from the SSURGO dataset, slope degree is derived from 10 m resolution elevation data, and mean annual precipitation is from the PRISM dataset.

Table 3. Actual volume change $\pm$ standard error of the mean per meter of ditch length from each category as classified by terminal nodes of GLMM tree. Ditch length is the length of ditch within the study area, classified by terminal nodes of GLMM tree.

GLMM tree node	Mean $\pm$ se (m^3)	Median (m^3)	Ditch length (km)
Node 4 - Dysic, low slope, low MAP	60.89 $\pm$ 4.88	55.52	568.47
Node 5 - Dysic, low slope, high MAP	87.78 $\pm$ 5.77	78.75	357.8
Node 6 - Dysic, high slope	120.08 $\pm$ 9.90	106.01	328.79
Node 8 - Euic, low slope	24.86 $\pm$ 2.60	17.42	1479.53
Node 9 - Euic, high slope	48.24 $\pm$ 5.01	36.78	1213.15

Table 4. Average volume loss scaled by total ditch length for each GLMM tree node. Estimates of C loss calculated using volume loss, an assumption of 51% oxidation, average bulk density values, and average C content. C loss per year ($\pm$ se) assumes constant rate of C loss for 100 years since drainage.

GLMM tree node	Total volume loss (m^3)	Total C loss (Mg)	Mean ditch effect width (km)	C loss (Mg $\text{ha}^{-1} \text{yr}^{-1}$)
Node 4 - Dysic, low slope, low MAP	31561454	628137	0.35	0.32 $\pm$ 0.06
Node 5 - Dysic, low slope, high MAP	28176750	560774	0.43	0.36 $\pm$ 0.06
Node 6 - Dysic, high slope	34855028	693686	0.41	0.51 $\pm$ 0.09
Node 8 - Euic, low slope	26086973	724930	0.25	0.19 $\pm$ 0.04
Node 9 - Euic, high slope	44619657	1239934	0.31	0.33 $\pm$ 0.07

Using 10-fold cross validation, mean aggregated R^2 from the 10 folds was 0.50 with a mean aggregated RMSE of the square-root-transformed response was 2.31.

Predicted volume change values within each site based on the classification of each ditch by the GLMM tree nodes were also correlated with transect averages from each AOI, scaled to AOI by ditch length ($R^2=0.74$) (Figure 4b). There was also a positive relationship between predicted values and volume change values measured using the surface trend method ($R^2=0.62$) (Figure 4c).

Figure 4. Volume loss within each AOI between: A) both volume change measurement methods (transect and surface trend) in the sites in which the surface trend analysis was conducted (N=24); B) Average AOI transect volume change measured and predicted volume change values derived from the GLMM tree classifications (N=55); C) Volume change measured in each AOI through the surface trend analysis and predicted volume change values derived from the GLMM tree classifications (N=24). Units in $1000 m^3$.

3.5 Total Peat Volume and Carbon Loss Estimations

Total estimated volume loss in the ditches within the 22,105 km² study area, calculated by multiplying the median volume change value for each terminal node of the GLMM tree by the respective ditch length, is approximately 165.3 million $\pm$ 8.6 million m³ (Figure 5). Despite making up only 32% of ditch length, dysic-classified ditches account for 57% of total volume loss.

Total estimated historic C loss throughout the study area was approximately 3.847 $\pm$ 0.364 Tg C, which is roughly 1/2000th of the total estimated C pool found in peatlands in the conterminous United States (Kolka et al, 2018). Due to higher average bulk densities, the proportion of C loss to volume loss is higher in euic types than in dysic types, though the elevated volume loss from dysic types means that C losses per meter ditch length are highest in dysic types (Table 4, Table 5).

Figure 5. Map of peatland distribution within the study area. Inset: Ditches color-coded by classification within a terminal node of the GLMM tree. (Graphic created in Arc Pro 2.5.1. Carbonate reaction class graphics were obtained from the SSURGO database and manipulated. Ecological provinces, composite aerial imagery, and state outline obtained from the Minnesota Geospatial Commons with full attribution in Appendix A).

Table 5. C loss estimates based on volume loss, average ditch effect width, average bulk density, dry mass percent C, and years since drainage. Losses from our study are compared with losses estimated in IPCC (IPCC, 2013).

Characteristic	Affected area (ha)	Mg C ha ⁻¹ yr ⁻¹	95% Confidence Interval		Mg CO ₂ ha ⁻¹ yr ⁻¹
Dysic	48636.24	0.40	0.15	0.65	1.46
Euic	75465.84	0.26	0.10	0.42	0.95
IPCC estimate (<i>Drained boreal peatlands</i>)		0.37	-0.11	0.84	1.36

4 Discussion

The primary objective of this study was to develop an estimate of total peat volume loss throughout the ditched peatlands in northern Minnesota and determine an approximation of the historic loss of C that it represents. Our results show that peat volume has not yet recovered along the extensive ditch network, despite the lack of ditch maintenance in many areas for a century. Both the volume loss and the associated C loss evident in our results have important implications for land managers.

4.1 Volume Loss

As hypothesized, dysic peatland types, which tend to be more fibric, displayed a more pronounced volume loss in response to drainage than did the more sapric, euic peatlands. This response can likely be explained largely by a difference between the bulk densities of the two peatland types and the effect of bulk density on peatland subsidence. Peatland subsidence is the effect of both mechanical and chemical mechanisms (Ewing and Vepraskas, 2006), beginning with the onset of drainage and the settling of peat material and its consolidation following the drawdown of water table levels (Wösten, 1997). Where the peat profile is saturated, positive porewater pressure alleviates effective stress on soil particles (Terzaghi, 1943), but as the water table is lowered, the large pores of the less decomposed, fibric peat are quickly evacuated of water with an increase in soil tension (Kennedy and Price, 2005), while water held in smaller pores of sapric peat is maintained under equal soil tension. Given the volume reduction potential determined by initial peat matrix pore sizes, it is expected that fibric peat would be more susceptible to

volume loss following drainage than would sapric peat, a pattern that is borne out in our results (Figure 2b).

Interestingly, volume loss had a positive relationship with MAP. This is contrary to what might be expected given the mitigating effects of increased precipitation inputs on the water loss induced by drainage. One explanation of this observation is the effect of increased precipitation on erosion losses along the ditches (Li et al., 2018). Liu et al. (2017) demonstrated that degradation of the peat matrix is related to an increase in preferential flow paths. The erosion of peat sediment can also be enhanced by cracking due to peat desiccation following drainage and is understood to increase exponentially with time (Holden et al., 2006), leading to an increase in suspended sediments to connected water bodies (Marttila and Klove, 2010). Dissolved organic C, an important component of peatland C balances (Roulet et al., 2007), has also been shown to increase following drainage (Evans et al., 2014). Export of particulate organic matter represents an additional pathway of C loss from the peatlands in the study area and has important implications with regards to the export of nutrients and mercury into Minnesota waterways (Kolka et al., 1999; Nieminen et al., 2017).

The ditch effect on peat volume loss on the downslope side of the ditch was significantly more than was the ditch effect on the upslope side for both euic and dysic classes ($P < 0.001$). Lateral flow of surplus water in a peatland is significantly less constrained in the more permeable acrotelm than in the deeper, compacted catotelm, meaning that the majority of horizontal water movement happens at the peat surface (Damman, 1986). A ditch, especially one that runs perpendicular to the slope of the landscape, will intercept

water from the upslope side and translocate it through the ditch network, starving the downslope side of water inputs. This new flow path not only causes enhanced volume loss but also limits peatland hydrological services. Peatland subsidence has been associated with a decrease in saturated hydraulic conductivity (K_{sat}) in the surface layers of peat due to the reduction of large pore sizes (Silins and Rothwell, 1998). Decreased K_{sat} limits horizontal movement of water through the peat, which increases preferential flow and degrades important ecosystem services provided by peatlands, including water storage, groundwater protection, and solute filtration (Liu et al, 2017; Lennartz and Lui, 2019).

It has been found that the proportion of subsidence due to mechanical processes decreases exponentially with time (Pronger et al., 2014). This is due to the rapidity with which initial mechanical densification of the peat matrix occurs during the first 25 years after drainage (Pronger et al, 2014) as well as a number of hydrological feedbacks that limit continued structural changes in an altered peatland (Waddington et al., 2015). Thus, the primary driver of continued subsidence in the study area is likely to be microbial oxidation.

4.2 Carbon Loss

The oxidative component of peatland subsidence describes the loss of C mass due to increased oxidation rates of organic material in the deeper aerobic layers of peat following drainage (Clymo, 1983; Drexler et al., 2009). The extent to which volume loss in subsided peat can be attributed to carbon mass loss from peat oxidation can vary

widely between peatland ecosystems (Couwenberg et al, 2010). Estimating subsidence attributable to oxidation as a percentage of total measured volume loss is one method that has been employed to approximate C loss (Grønlund et al., 2008; Leifeld et al, 2011). We assume a value of 51% of subsidence due to oxidation that was derived using a regression equation presented in Pronger et al. (2014) and assuming 100 years since drainage. Pronger et al. (2014) used years since drainage to predict the proportion of subsidence due to oxidation based on a synthesis of the results of five separate studies that demonstrated its strong relationship with time since drainage. Because few studies exist that directly estimate the proportion of subsidence attributable to oxidation, our assumption of 51% should be interpreted with the understanding that there is wide variability in the hydrological and biogeochemical dynamics that drive C mass loss in peatland ecosystems, and further research is warranted to decrease the range of possible estimates. A study of the changes in peat bulk density and C content along the transects that were used in our analysis would help narrow this variability and strengthen our preliminary estimates.

Based on predicted volume loss, average peatland bulk density values, ditch effect area, and oxidation percent of subsidence, and assuming a constant rate of C loss over the course of the 100 years since drainage, our C loss estimates are similar to estimated annual CO₂-C emissions from drained boreal peatlands described by the Intergovernmental Panel on Climate Change (IPCC, 2013) (Table 5). It should be noted that our C loss estimates are partially dependent on average bulk density values, which vary spatially, both horizontally and with depth. In using an average bulk density value

for two broad peatland types, we likely overestimate C loss from ditches in which volume loss has occurred primarily in the top 0-25 cm of peat, while underestimating C loss where volume loss has extended into deep peat layers.

4.3 Environmental Management Implications

Given the need to identify strategies for the retention and restoration of C in natural landscapes to mitigate GHG emissions (Griscom, 2017), peatland restoration in Minnesota presents itself as an important pathway to protect what has recently been termed "irrecoverable carbon" (Goldstein et al., 2020). The dense C stores and slow C accumulation rates (Loisel et al., 2014) mean that continued C emissions from these ecosystems represent C losses unlikely to be restored in a timescale relevant to the remaining global C budget described in IPCC projections (IPCC, 2018; Goldstein et al., 2020). This suggests that the efforts of land managers should be prioritized toward management activities that limit further C losses from peatlands. According to our estimates, degraded Minnesota peatlands in our study area lose c. 38,000 Mg C per year, roughly equivalent to the average annual CO₂ emissions from 31,000 personal vehicles (EPA, 2018). One strategy to mitigate this C loss is to restore the natural hydrology of a peatland by blocking or filling drainage ditches. Effective hydrological restoration raises water tables and restores biogeochemical processes that protect C stores and facilitate continued peat accretion. It has also been shown to restore the hydrological buffer function of surface peat layers, reducing the connectivity of surface water runoff to groundwater (Ahmad et al, 2020). Land managers are encouraged to explore resources

that detail restoration strategies, goals, and outcomes relevant to individual sites and management objectives (Landry and Rochefort, 2012).

While numerous studies have found that peatlands revert to C sinks in the years following peatland restoration (Wilson et al., 2016, Nugent et al., 2018, Günther et al., 2020, Ratcliffe et al., 2020), a number of uncertainties persist, including the dynamics of other GHGs, especially methane (CH₄) and nitrous oxide (N₂O). It is generally understood that CH₄, a GHG with a global warming potential (GWP) 28-34 times more than that of CO₂ (Myhre et al., 2013), decreases with drainage (Glenn et al., 1993). Conversely, N₂O has a GWP almost 300 times higher than that of CO₂ (Myhre et al., 2013), and has been documented increasing following drainage (Salm et al., 2012; Liu et al, 2020). Peatland restoration as a GHG mitigation strategy must therefore be assessed to determine the net radiative forcing effect on Minnesota landscapes. One common practice has been to assess gas exchanges with eddy covariance studies (D'Acunha, et al., 2019). Such ecosystem-level studies will be key in helping identify the most effective natural climate solution strategies available to land managers in Minnesota and could be used to improve the preliminary findings of this study.

Land management strategies to mitigate continued emissions will depend on an assessment of the management objectives and unique characteristics for each site, but our preliminary results can be used to guide initial efforts aimed at prioritizing peatland restoration activities. Assuming similar oxidative subsidence in both peatland types, our results indicate that, per meter ditch length, dysic peatlands are expected to lose more C than euic peatlands, despite higher nutrient availability and C densities in the latter.

Higher volume losses in dysic peatlands, however, have the potential to complicate restoration efforts. In-filling of drainage ditches paired with reinforcing check dams is a common restoration strategy, but sites with elevated volume loss may require more fill material, raising restoration costs. Access to a site and availability of fill material is another important consideration. Some ditches, for example those north of Upper Red Lake, may prove to be all but inaccessible to the machinery that would be required for large-scale restoration. Trees that have colonized the de-watered margins of ditches have recently been used as ditch fill (Myers, 2015), presenting a readily available solution to the issue of sourcing and transport of fill material, which can be a significant cost in peatland restoration (Chimner et al., 2018). Using landscape- and site-level indicators to identify and prioritize peatland restoration opportunities, land managers will be able to explore cost effective mitigation strategies (Griscom et al., 2017).

Our analysis contributes to the understanding of the effects of legacy drainage features in Minnesota and demonstrates a significant loss of wetland volume and a potential area for GHG emission mitigation on Minnesota's natural landscapes. While further research is needed to determine local drivers of peatland degradation and the net radiative forcing effect of peatland restoration in Minnesota, the results of this study can be used to help natural resource managers gauge the estimated C cost of the continued preservation of legacy ditches on the landscape as they exist today.

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Figure 1. Minnesota IT Service, Geospatial Information Office. LiDAR elevation data for Minnesota. Scale Not Given. <https://www.mngeo.state.mn.us/chouse/elevation/lidar.html>. Accessed October, 2019.

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