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ORIGINAL RESEARCH

Automated Techniques to Identify Lost and Restorable Wetlands in the Prairie Pothole Region

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Abstract Wetland loss in the Prairie Pothole Region has been substantial, and automated techniques to estimate wetland loss and identify priority wetlands for restoration are crucial if important ecosystem services provided by wetlands are to be maintained. A suite of automated methods was developed to establish a historical wetland inventory and to identify the proportion of permanently and temporarily lost wetlands in a prairie pothole watershed in Alberta, Canada. A power law analysis of area vs. frequency of historical wetlands provided estimates of permanently lost wetlands. Combining the historical wetland inventory with an inventory of existing wetlands provided estimates of temporarily lost wetlands. 22,204 historical wetlands comprising 12,431 ha were estimated in the watershed. Permanently lost wetland number and area were estimated as 11.1% and 0.6% respectively, and temporarily lost wetland number and area were estimated as 61.1% and 78.3% respectively. Existing wetlands represented only 27.8% of the total historical number and 21.2% of the total historical area. 1,588 ditch-drained (relatively easy restore) wetlands were identified from the inventory of temporarily lost wetlands using digital terrain analysis, representing a potential recovery of 7.2% of the historical wetlands by number and 9.8% by area.

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Introduction

Wetlands are among the most valued ecosystems in terms of ecosystem services (Costanza et al. 1997); however, loss rates are exceptionally high in many regions (Zedler and Kercher 2005). Within the Prairie Pothole Region of North America, the majority of wetlands have been lost to agriculture by filling and draining (Watmough and Schmoll 2007; Dahl 2014). Farmers alter wetlands to increase property access, cultivated area, crop yield, and the diversity of crop options (Van der Gulik et al. 2000; Blann et al. 2009). In the face of continued wetland loss, wetland policies are increasingly moving towards not only protection but also restoration to re-establish ecosystem service functions. Wetland restoration begins with understanding where lost wetlands have been located and which wetlands can be most efficiently restored. While recent progress has been made in the automation of wetland mapping (Lang et al. 2012; Tiner et al. 2015; Serran and Creed 2016), simple automated techniques to identify lost and restorable wetlands are still needed to support wetland management decision-making (Dahl and Watmough 2007; Clare and Creed 2014).

Automated approaches for wetland mapping take advantage of a large variety of remotely sensed data including aerial photography, light detection and ranging (LiDAR) data, and radar, multispectral and hyperspectral satellite imagery (Ozesmi and Bauer 2002; Baker et al. 2006). The increasing availability of fine resolution data creates further opportunities to improve automated wetland mapping techniques. In particular, object-based methods classify groups of pixels in order to take advantage of the spatial context in which individual pixels exist (Dronova 2015; Knight et al. 2015). Serran and Creed (2016) applied object-based methods to a depression probability surface derived from a fine resolution LiDAR digital elevation model (DEM) to map the locations and boundaries of remnant "historical" wetlands.

A historical wetland inventory can be used to identify wetland loss. Due to the fractal nature of natural waterbodies, wetland area vs. frequency plots follow a power law function; i.e., a negative linear relationship when plotted on logarithmic-logarithmic axes (Downing et al. 2006; Zhang et al. 2009; Seekell et al. 2013; Van Meter and Basu 2015; Serran and Creed 2016). When applied to historical wetland inventories, permanent wetland loss – wetlands whose basins are no longer detectable on the landscape (i.e., filled wetlands) – is estimated by the deviation from the power law. In addition, temporary wetland loss, wetlands whose basins are intact but that are not captured in existing wetland inventories, is estimated by comparing the historical wetland inventory to an existing wetland inventory. Together the delineation of historical wetlands and the estimate of permanent and temporary loss provides insight into the fate of historical wetlands on the landscape.

Mechanisms of wetland loss include filling and drainage associated with urbanization and agricultural cultivation (Gleason and Euliss 1998; Watmough and Schmoll 2007). Restoring filled wetlands requires the excavation of fill, recontouring of the wetland depression, and revegetation (Galatowitsch and van der Valk 1994). The high cost of restoring filled wetlands means that it is rarely pursued and filled wetlands are generally seen as a form of permanent loss. Drainage is one of the most common mechanisms of wetland loss (Government of Manitoba 1985; Watmough and Schmoll 2007; Blann et al. 2009). Surface drainage uses a drainage ditch to carry water away from a wetland; drainage ditches vary morphologically, but are usually between 1 and 10 m wide and up to 1 m deep. Subsurface or tile drainage uses a network of underground perforated pipes to divert water but is not common in the Canadian Prairies, likely due to higher costs compared to surface ditches as well as the potential for underground pipes to become blocked by ice, particularly in Alberta during winter Chinooks (Government of Manitoba 1985; Watmough and Schmoll 2007). The use of tile drains increases in the southern US portion of the Prairie Pothole Region (Dahl 2014).

Progress has been made in automating the delineation of wetlands; however, the identification of restorable wetlands remains a challenge. Each case of wetland restoration is unique; however, the wetlands that are easiest to restore are generally those that have minimal changes (Galatowitsch and van der Valk 1994). Ditch-drained wetlands can be restored by filling a portion of the drainage ditch, known as ditch plugging, while tile-drained wetlands can be restored by breaking and removing portions of the perforated pipes. Ditch-drained wetlands have an advantage for restoration because wetland plants often grow in the ditches, serving as seed banks for restored wetlands (Galatowitsch and van der Valk 1994) and therefore are considered a form of restorable wetland loss. Methods of identifying wetland loss do not currently target the mechanisms of wetland loss; however, this is changing with advancing ability to detect small features from high-resolution imagery. The ability to capture the mechanism of wetland loss varies with the footprint left by different wetland impacts. Drainage ditches leave visible concave features on the surface, while surficial evidence of a tile-drained wetland may consist only of an inlet pipe (Biebighauser 2007). Therefore, the growing availability of high-resolution spatial data holds promise for the identification of restorable ditch-drained wetlands.

This paper focuses on building a comprehensive wetland inventory from which the following research question can be asked: what are the magnitudes of existing, restorable (i.e., ditch-drained), permanently lost and temporarily lost wetlands within an agricultural watershed? The object-based method developed by Serran and Creed (2016) is applied to high-resolution elevation data to delineate historical wetlands in a prairie pothole watershed in Alberta, Canada. The power law relationship between area and frequency in the historical wetland inventory is analyzed to understand the extent of permanent wetland loss, and the historical wetland inventory is compared with the Canadian Wetland Inventory of existing wetland features to identify the temporary wetland loss that represents the potential for wetland restoration. An automated digital terrain analysis method to identify restorable wetlands is then developed to identify drainage ditches in historical wetlands. A better understanding of the magnitude of wetland loss as well as of the spatial distribution of restorable wetlands will aid in the restoration of the number, area and distribution of wetlands within watersheds, a common goal in wetland restoration efforts.

Methods

Test Area

The Prairie Pothole Region extends across central North America where the retreat of the Wisconsin glacier left behind millions of depressional wetlands in the fine-grained glacial till, known as prairie potholes (Johnson et al. 2008; Dahl 2014). Prairie potholes tend to be small (< 1 ha) (van der Valk and Pederson 2003; Watmough and Schmoll 2007) and shallow (< 1 m in depth) (Huang et al. 2011), and generally become dry through summer and fall (Winter 1989).

The Nose Creek watershed (51°16′57″N, 114°7′14″W) comprises 886 km^2 in the western portion of the Prairie Pothole Region along the northern edge of Calgary, Alberta

(Fig. 1). The watershed is characterized by a dry continental climate with mean annual temperature of 4.4 °C, mean annual precipitation of 418.8 mm/yr., and mean annual moisture deficit based on potential evapotranspiration (Hamon 1961) minus precipitation of −107.4 m for 1981–2010 (Environment Canada 2015). The watershed is underlain by the Paskapoo Formation which comprises sandstones, mudstones, and siltstones (Hamblin 2004) and is overlain by fertile Black Chernozemic soils (Agriculture and Agri-Food Canada 2016). The landscape consists largely of rolling and undulating plains with the topography ranging from 1336 m to 1048 m above sea level. The western half of the watershed is covered by aspen forests and willow scrublands mixed with grasslands, and the eastern half is covered by grasses (Natural Regions Committee 2006).

The Nose Creek watershed has been modified extensively by agricultural and urban development, leaving only small areas of native vegetation. Development covers approximately 12% of the watershed and is concentrated in the Town of Crossfield and the rapidly growing cities of Airdrie and Calgary. Agricultural activities occur over approximately 70% of the watershed (Agriculture and Agri-Food Canada

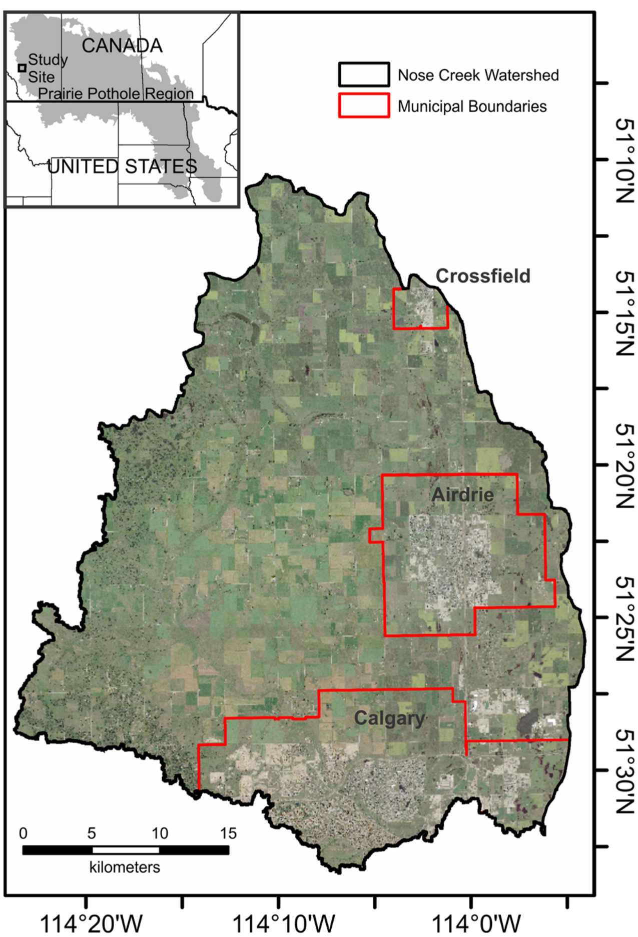


Fig. 1 Map showing the location of the Nose Creek watershed, Alberta, Canada. The watershed is dominated by agricultural activities

2013), taking advantage of some of the most productive croplands in Alberta (Natural Regions Committee 2006). The most common agricultural crops include canola, spring wheat, barley and alfalfa (Government of Alberta 2012). Where the terrain is not favourable to crops, grazing predominates. The dominance of agricultural activity in the watershed makes it ideal for the development of a method to identify wetlands altered due to agricultural activity, namely wetland drainage through surface ditches.

Historical Wetlands and Permanent and Temporary Wetland Loss

Definitions of existing, historical and restorable wetlands and permanent and temporary loss as used in this paper are summarized in Table 1.

Delineating historical wetlands consisted of four steps. First, a Monte Carlo approach was used to derive a depression probability (p_{dep}) surface from a 1-m bare earth LiDAR DEM acquired in October 14–17 2014 (absent deciduous canopy, during the driest part of the year, and during a drier than average year) and resampled using bilinear interpolation to 3-m pixel spacing. A distribution of random elevation errors with a standard deviation equal to the 15 cm vertical accuracy of the DEM was used to determine random error terms that were added to the DEM in 1000 iterations. Each error-added DEM was filled using the Planchon and Darboux (2001) depression filling algorithm and pixels that were filled were flagged as depressions. P_{dep} was calculated for each pixel as the number of times it was flagged as a depression divided by the number of iterations (Lindsay and Creed 2006). The digital terrain analyses were performed using the Terrain Analysis System version 2.0.9 software (Lindsay 2005).

Second, an object-based approach was used to segment and classify the p_{dep} surface. The multi-resolution segmentation algorithm of Baatz and Schäpe (2000) was used to merge adjacent pixels of relative p_{dep} homogeneity into image objects. Unitless segmentation scale parameters determining the average size of objects govern the degree of homogeneity allowed for pixel merging; a small scale parameter (2) was used to generate depression object "pieces" and a large scale parameter (20) was used to prevent fragmentation of larger depression objects. Segmentation was constrained by a road vector layer (Alberta Environment and Parks 2015) buffered 15 m on each side to prevent objects from crossing roads. Objects segmented using the small scale parameter with mean $p_{dep} \geq 0.52$ were classified as historical wetland objects; objects segmented using the large scale parameter with mean $p_{dep} \geq 0.45$ were classified as historical wetland objects. Adjacent classified historical wetland objects were then merged to create historical wetland features. Segmentation scale parameters and p_{dep} classification thresholds were selected based on previous work in a nearby watershed (Serran and

Table 1 Glossary of wetland terms

Term	Definition
Existing wetlands	Wetlands mapped in the Canadian Wetland Inventory (CWI) minus any wetlands identified as restorable
Historical wetlands	An estimate of the historical extent of wetlands including wetlands that are extant and remnant on the landscape as well as wetland number and area estimated from the power law trend extrapolated to the minimum size of wetlands
Permanent loss	Wetlands whose basins are no longer detectable on the landscape (i.e., filled or paved over); number and area of permanent loss is estimated from the area between the extrapolated linear power law and the observed deviation from the power law
Restorable wetlands/ Restorable loss	Temporarily lost wetlands with evidence of a drainage ditch
Temporary loss	Remnant wetlands on the landscape whose basins are still detectable but that are no longer detected in existing inventories; estimated by comparing the historical wetland inventory to the existing wetland inventory
Total wetland loss	Permanent loss plus temporary loss

Creed 2016). Segmentation and object classification and merging was performed in eCognition Developer software (Trimble Navigation Limited 2009). Following classification and merging, some historical wetlands included "tails" where drainage ditches existed. These were removed by simplifying the non-riparian historical wetland boundaries using the Clean tool from the ET Geowizards extension (Tchoukansi 2012) for ArcGIS, where non-riparian refers to wetlands that do not intersect riparian features.

Third, wetland features in the Canadian Wetland Inventory (CWI) were added to the inventory of historical wetland features, based on the assumption that if a wetland existed in the CWI, it should also be present in the historic wetland inventory. The CWI was delineated using stereo pairs of high-resolution panchromatic aerial photographs from 2006 to a minimum mapping unit (MMU; i.e., the smallest wetland that can be reliably mapped) of 0.02 ha (Ducks Unlimited Canada 2006). Wetland features in the merged historical inventory less than 0.02 ha area were removed.

Fourth, a piecewise linear regression was then applied to the historical wetland data to identify the power law line, which describes the trend of decreasing wetland number with increasing wetland area, and breakpoints in the power law line, which describe deviations from the wetland area vs. frequency relationship. The power law is based on the fractal nature of natural waterbodies (Downing et al. 2006); therefore, developed areas, where waterbodies are largely engineered, were removed from all analysis. Power law analyses are recommended for regional scales, such as watersheds, to establish a large enough sample size to clearly identify a power law trend (Serran and Creed 2016). Wetland inventories derived from high resolution data are also required, to reduce the amount of extrapolation that is sensitive to changes in power law parameters (McDonald et al. 2012; Muster et al. 2013), as well as to capture the small area breakpoint (usually <0.5 ha) (Serran and Creed 2016).

Wetlands were binned by area starting from the smallest wetland size of 0.02 ha. The bin increment was chosen objectively as the coarsest resolution of the data used to create the historical wetland inventory (0.0009 ha or 9 m^2). When applying a piecewise regression to the wetland area vs. wetland frequency data plotted on logarithmic-logarithmic scales, there are often two breakpoints, one breakpoint at a smaller wetland area, and a second breakpoint at a larger wetland area. A three-segment piecewise regression was first applied to identify the large area breakpoint. Data with an area above the large area breakpoint and data with a frequency below the large area breakpoint were removed (Serran and Creed 2016). Removal of data above the large area breakpoint isolates the power law trend in the data, removing the influence of large wetland areas where the frequency begins to be one. A two-segment piecewise regression was then run on the remaining data to define the power law line, the small area breakpoint, and the deviation of historical wetland data from the power law line. This power law relationship was extrapolated to the MMU (0.02 ha) and the deviation of the mapped historical wetland data from the power law line was assumed to represent permanently lost wetlands (Serran and Creed 2016). For each bin below the small area breakpoint, the number of permanently lost wetlands was calculated as the difference between the frequency estimated by the power law line and the frequency of the mapped wetlands, and the area of permanently lost wetlands was calculated as the difference between the area estimated by the power law line, and the area observed for mapped wetlands. Given that the historical wetland data are topographically-based, wetland loss estimates derived from deviation from the power law line represented wetlands that are no longer topographically detectable on the landscape; i.e., wetlands were filled or paved and are therefore permanently lost. The number and area of permanently lost wetlands was added to the number and area of the mapped historical wetlands to establish a final estimate of the historical wetland inventory. An existing wetland inventory was then created by using the CWI and removing any wetlands that were identified as restorable (method to identify restorable wetlands presented below).The difference between the historical wetland inventory and the existing wetland inventory was used to estimate temporarily lost wetlands (Van Meter and Basu 2015).

Restorable Wetlands

A flowchart of the method developed to identify restorable wetlands is shown in Fig. 2 and consists of curvature analysis, reclassification, noise removal, and location filters to produce a final inventory of restorable wetlands. Visual examples for each step are presented in Fig. 3. The identification of drainage ditches hinged on their topographic concavity properties. Curvature of the 1-m bare earth DEM surface was derived using the Curvature tool in ArcGIS which calculates the second derivative of the surface. A low pass filter was applied to smooth the data and remove noise. The smoothed curvature data were then reclassified to isolate the drainage ditches. Given that concave features have negative curvature, an upper threshold was set to separate potential drainage ditches from other features. Jenks classification, an iterative variance minimization classification (Jenks 1967), was used to identify an appropriate break in the data associated with drainage ditch features. The chosen threshold, −6.61, was a stable

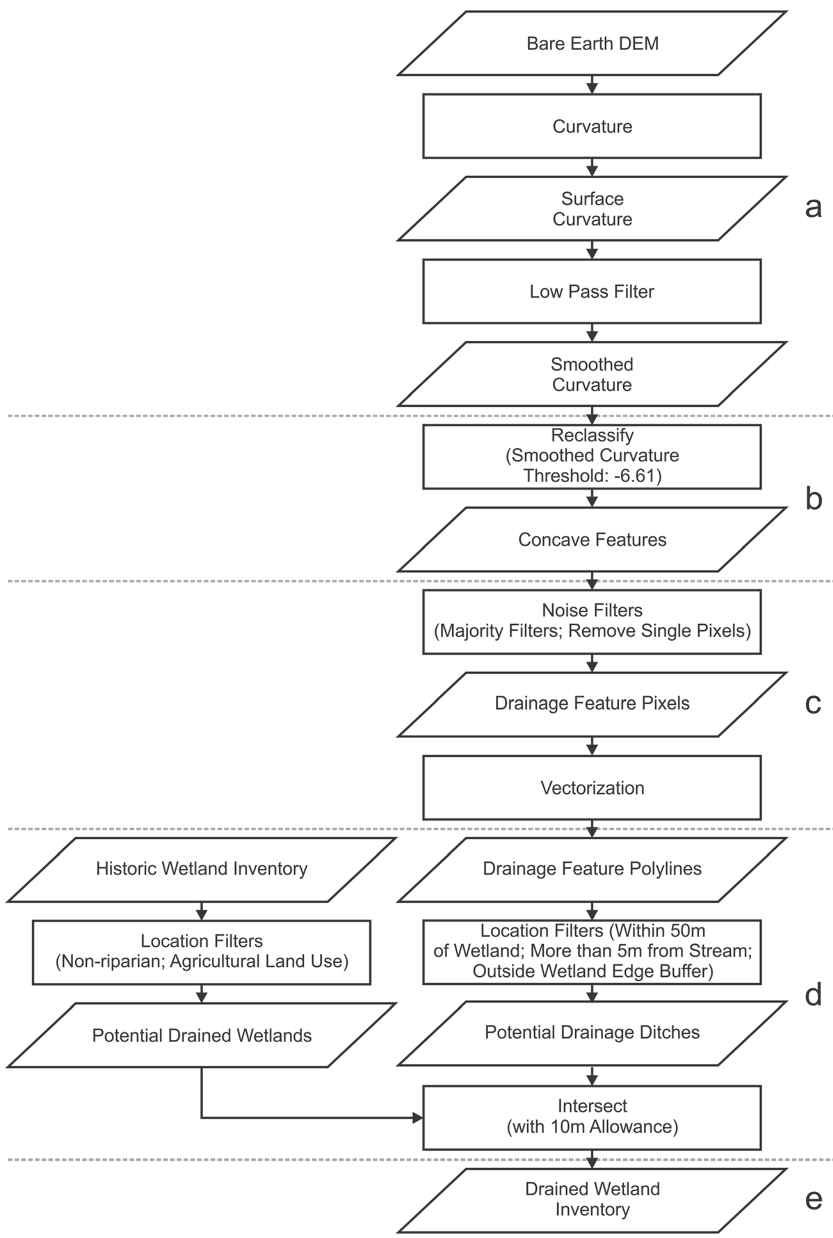


Fig. 2 Flowchart of steps to delineate ditch-drained wetlands: (**a**) the surface curvature quantifies the convexity or concavity of a surface; (**b**) curvature reclassification narrows down the area of interest to concave features; (**c**) noise filters remove features that are not of interest including single pixels and features resulting from roads and railways; (**d**) location filters reduce potential drainage ditch candidates to those that are near wetland boundaries; and (**e**) the intersection of drainage ditch candidates with non-riparian wetlands on agricultural land identifies ditch-drained wetlands

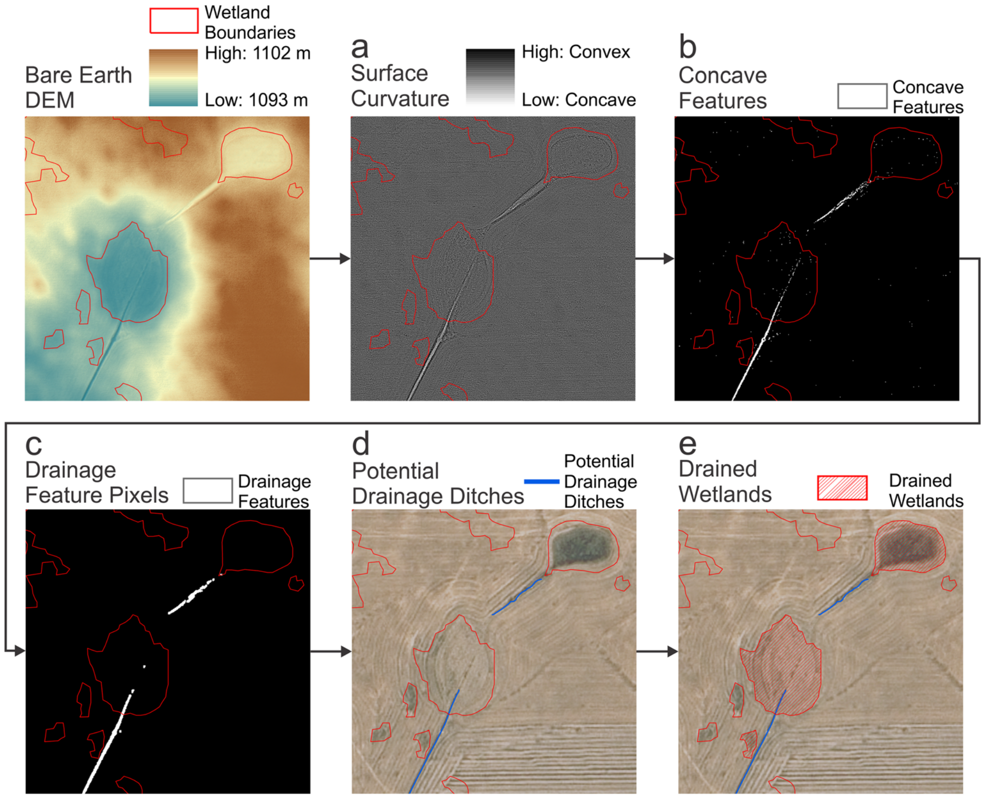


Fig. 3 Illustrations of the ditch-drained wetland mapping method. A 1-m bare earth DEM is used to calculate (**a**) surface curvature which quantifies the convexity or concavity of a surface; (**b**) reclassification narrows down the area of interest to concave features; (**c**) noise filters remove features that are not of interest (single pixels, convex features from roads and railways); (**d**) location filters narrow down the potential drainage ditches based on their spatial relationship with wetland boundaries; (**e**) the intersection of drainage ditch candidates with non-riparian wetlands on agricultural land identifies ditch-drained wetlands. Image centroid location: 51.2218°, −113.9186°

break point, consistently appearing when data were binned into four or more classes. The data were reclassified to a binary map with smoothed curvature values less than or equal to −6.61 (representing potential drainage ditches) reclassified as “1” and all other values (representing other surfaces) reclassified to “0”.

The potential drainage ditches were filtered to remove features that were not of interest in the context of restorable wetlands. Drainage ditches are often located along roads and railroads, and although these features may in fact be draining wetlands it is unlikely that these permanent infrastructure features would be altered to restore wetlands. Therefore, drainage ditches located along roads and railroads were removed from consideration by creating a 15 m buffer on each side of these linear features and re-assigning the ‘potential drainage ditch’ pixels within these buffers to the ‘other surface’ class. Noise in the form of single pixels was removed by applying a majority filter. Some drainage ditch features were broken up into pieces due to spatial variation in the smoothed curvature values along the feature; the Expand tool in ArcGIS was used to grow the potential drainage ditch features by 1 pixel (1 m) in each direction to create more continuous features. The potential drainage ditch features were then vectorized to allow for analysis of their spatial relation to wetland boundaries.

The potential drainage ditch lines were filtered based on their location. Drainage ditches are likely to cross or be adjacent to the boundaries of the wetlands they are draining, and so lines more than 50 m from a historical wetland were removed. To eliminate natural drainage features, lines completely contained within 5-m buffers of base stream flow features (Alberta Parks and Environment 2015) were also removed. The intersection of drainage ditch lines with wetland boundaries posed a challenge because the change in slope that can occur along the boundaries of a wetland depression resulted in concave features similar to drainage ditches. A 15 m buffer centered on the boundary of the historical wetlands was created; the 15 m buffer was chosen heuristically to balance the removal of concave features resulting from wetland edges while minimizing the elimination of drainage ditches. Drainage ditch lines completely contained within these boundaries were eliminated. The remaining curvature features were buffered by 5 m and the polygon buffers were converted to polylines.

Wetlands within agricultural land use areas were considered, as the practice of ditch-draining wetlands is largely associated with agriculture. The Agriculture and Agri-Food Canada (2013) annual crop inventory was used to identify agricultural land use. Riparian wetlands (i.e., wetlands that intersect streams) were removed as these wetlands are already connected to the drainage network and therefore not drained. Ditch-drained wetlands were identified as any non-riparian agricultural wetland within 10 m of a potential drainage ditch. The 10 m distance allowance served as a precautionary safety net to capture drainage ditches that lay just outside the periphery of wetland boundaries, and was chosen because initial accuracy assessment following the use of a simple intersect with no distance allowance resulted in some drained wetlands not being captured to an average distance of approximately 10 m (data not shown). Our wetland mapping method should capture the full extent of a wetland; however, if a wetland has been breached by a drainage ditch, the fill algorithm will have a lower spill elevation, which may result in a smaller extent being filled and would necessitate the use of the 10 m buffer.

Performance of Wetland Mapping Tools

One hundred random wetlands classified as ditch-drained (i.e., non-riparian agricultural wetlands within 10 m of a drainage ditch) and 100 random wetlands not classified as ditch-drained were sampled to assess the accuracy of the method for detecting ditch-drained wetlands. To avoid propagation of errors, only wetlands in the historical wetland inventory that showed evidence of being a wetland were included in the sample sets. Evidence included inundated areas or wetland vegetation patterns (i.e., concentric bands of vegetation varying with moisture) present at any point within 21 aerial and satelite images from 1949 to 2014. In addition, dugouts were manually excluded from the sample as these human made features are not the focus of wetland restoration efforts. The samples were visually assessed for the presence of a drainage ditch feature using the DEM and historical and contemporary imagery. The accuracy assessment was determined by generating a confusion matrix and calculating overall accuracy, producer's and user's accuracy, and Cohen's Kappa statistic (Congalton and Green 2008).

Results

Historical, Permanently Lost and Temporarily Lost Wetlands

Historical wetland number and area in the undeveloped areas of the Nose Creek watershed were estimated as 22,204 and 12,431 ha, respectively. These estimates were derived from (1) 19,753 features in the historical wetland inventory with a total area of 12,362 ha (i.e., 15.5% of the undeveloped watershed area) (Fig. 4a), and (2) the deviation from the power law which represented a permanent loss of 2,451 small (< 0.052 ha) historical wetlands with an area of 69 ha (Fig. 5). For the power law analysis, the large area breakpoint was 0.592 ha and the small area breakpoint was 0.052 ha. The difference between the historical wetland inventory and the existing wetland inventory (Fig. 4b), after accounting for permanent loss, represented the temporary loss of wetlands, which was 13,571 wetlands with an area of 9,732 ha.

The area vs. frequency distributions of the historical and existing wetland inventories are shown in Fig. 5. By considering the wetland inventories together, absolute and proportions of wetland loss were calculated (Fig. 6). Of the historical wetlands, only 27.8% of the number and 21.2% of the area remain on the landscape. The loss of wetlands is not distributed evenly across wetland sizes; there was substantial loss (> 75%) of both small (especially <0.32 ha) and large (> 0.82 ha) wetlands (Fig. 7). The least loss, about 50%, occurs for wetland sizes between 0.32 ha and 0.82 ha. Of the historical wetlands that were not completely lost, the number (Fig. 8a) and area (Fig. 8b) of existing wetland features within a historical wetland extent indicates that historical wetlands are breaking up into smaller wetland features, with the degree of fragmentation increasing with wetland size.

Restorable Wetlands

Of the 11,279 non-riparian agricultural historical wetlands with a total area of 3,060 ha, 1,588 wetlands with a total area of 1,220 ha were classified as ditch-drained (Fig. 4c); that is, 14.1% of the number and 39.9% of the area of the non-riparian agricultural historical wetlands were ditch-drained. This means there is an opportunity to increase existing wetland numbers by 25.7% (from 6,182 to 7,770 wetlands) and wetland area by 46.4% (from 2,630 to 3,850 ha) through restoration of ditch-drained wetlands. An accuracy assessment of the classification of restorable wetlands is presented in Table 2. The overall accuracy was 76.0% and the Kappa coefficient was 0.52. Of the wetlands classified as ditch-drained, 65.0% of the wetlands had a drainage feature present. In the remaining 35.0% of cases, misclassification was due to other concave features resulting from berms, the bottom of hills, furrows, or wetland edges. Of the wetlands not classified as ditch-drained, 87.0% of the objects did not have a drainage feature present. In the remaining 13.0% of cases, misclassification was due to ditch features being too shallow and therefore no curvature feature was detected, or a curvature feature was present but it was more than 10 m from the wetland.

Discussion

Automated wetland mapping methods that provide inventories of existing, lost and restorable wetlands are needed to provide tangible launch points for wetland management. This study quantifies the potential for wetland recovery by categorizing and quantifying permanent wetland loss, temporary wetland loss, and restorable wetland loss. Previous studies have used similar methods to identify temporary loss (Van Meter and Basu 2015) or permanent loss (Serran and Creed 2016); however, this study presents a comprehensive inventory of wetland loss directed toward wetland restoration.

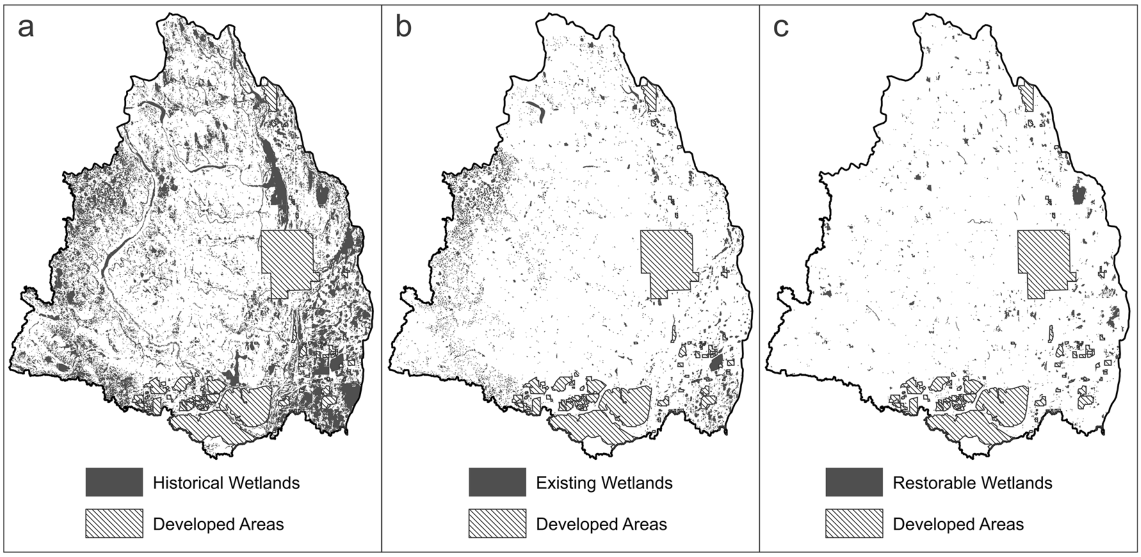
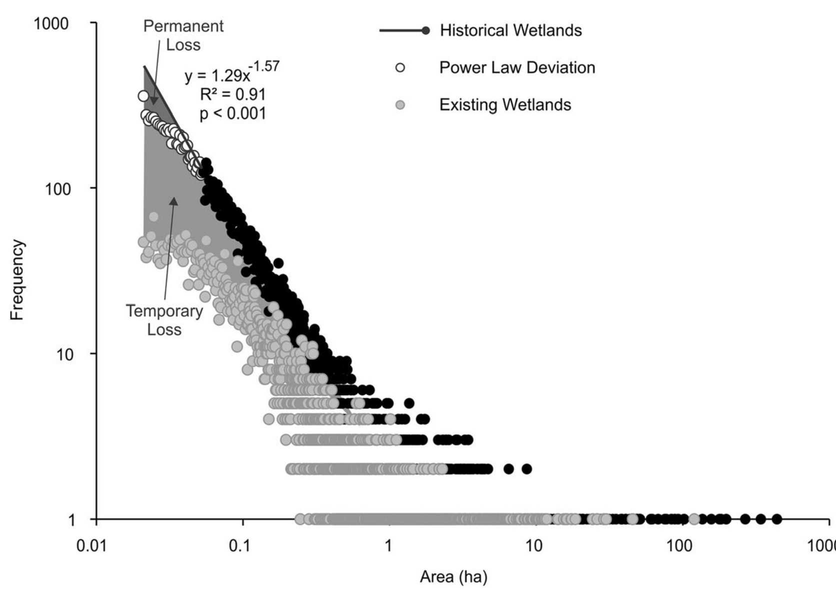


Fig. 4 Maps of (**a**) historical wetlands (not including the aspatial estimate of permanent loss), (**b**) existing wetlands, and (**c**) ditch-drained wetlands within the undeveloped areas of Nose Creek watershed

Historical Wetland Inventory

The historical distribution of wetlands can be difficult to determine given that they are often altered and therefore more difficult to detect on contemporary landscapes. A historical time series of imagery can be used to create wetland inventories through time; however, this requires a significant amount of data and analysis (Ozesmi and Bauer 2002) and assumes that the imagery is available at the temporal and spatial resolution needed. The automated method of identifying historical wetland extent presented here was chosen for the following reason – historical wetland estimates could be identified based on contemporary landscapes. This strength is particularly advantageous in areas where historical data are not available. The development of continually improving technologies to capture topography is resulting in the increased availability of finer resolution data both in space and time (Knight et al. 2015). The accessibility of fine resolution data means that the automated tool has the potential to be applied broadly to provide historical estimates of wetland extent as well as permanent and temporary loss. In addition to identifying wetlands, the use of LiDAR data also allows for wetland characterization and classification by providing information on vegetation (Rosso et al. 2006; Gilmore et al. 2008), and water flow and storage (Lindsay et al. 2004; Lane and D'Amico 2010; Huang et al. 2011; Knight et al. 2013). LiDAR data can also be combined with emerging airborne and satellite remote sensing technologies (Töyrä and Pietroniro 2005; Moffett and Gorelick 2013; Huang et al. 2014; Lang et al. 2015), further improving and enhancing wetland identification and

Fig. 5 Area vs. frequency distributions in logarithmic-logarithmic scales of the historical wetland inventory and existing wetland inventory with a bin size of 9 m^2. Analysis of the historical inventory identifies the power law which is extrapolated to identify permanent loss. A comparison of the historical and existing inventories identifies temporary loss



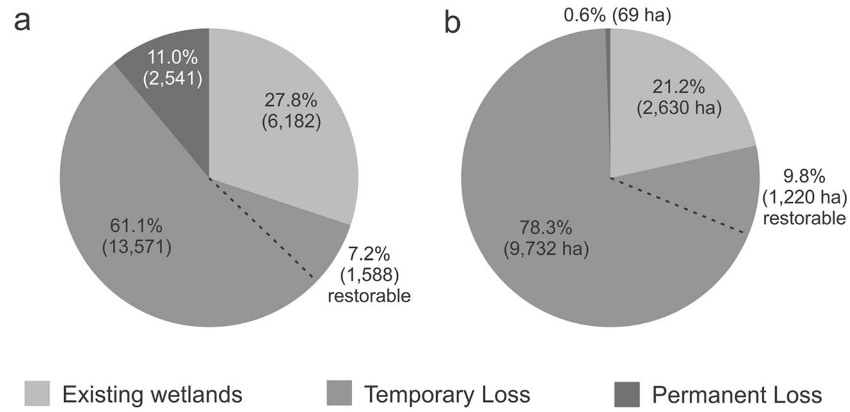


Fig. 6 Pie charts showing the percent and absolute numbers (in brackets) (**a**) by wetland number, and (**b**) by area of historical wetlands in the undeveloped areas of the watershed which are permanently lost, temporarily lost, and existing. Under temporary loss, the percent of historical wetlands which are restorable is also shown

characterization capabilities. One limitation of this method is that identifying wetland extent using depressional filling algorithms takes into account topography but not hydrology, and therefore wetland area can be overestimated. Area estimates can be improved by considering hydric soil data in establishing wetland presence (McCauley and Jenkins 2005; Miller et al. 2009; Van Meter and Basu 2015). Where culvert information is available, the wetland count can also be improved by identifying wetlands bisected by roads which are in fact one wetland connected through culverts.

Proportion of Permanently and Temporarily Lost Wetlands

The power law line in the area vs. frequency plot was used to estimate permanently lost wetland number and area; i.e., wetlands that have been filled and whose basins are no longer detectable. These non-spatial estimates of permanent wetland loss are adequate given that permanently lost wetlands are unlikely to be the focus of restoration efforts. The combination of historical wetland inventory and existing wetland inventory was used to capture temporary wetland loss; i.e., those wetlands that are not intact but whose wetland basins are still detectable. There are remarkably few detailed estimates of wetland loss for the region against which to compare these results (Dahl and Watmough 2007). Wetland loss across the Prairie Pothole Region has been reported to be between 40 and 70% since settlement (Schick 1972; Lynch-Stewart 1983; Rakowski and Chabot 1984; Environment Canada 1986; Glooschenko et al. 1993; Strong et al. 1993; Rubec 1994; Alberta Environment 1996; Dahl and Watmough 2007); however, many reports of loss lack vital details of how loss is defined, how estimates were derived, what the minimum size of wetland that is considered, and whether loss is reported by number or area. Several wetland loss estimates commonly cited for the region are also derived from unpublished reports (e.g., Schick 1972; Goodman and Pryor 1972; Rakowski and Chabot 1984; Strong et al. 1993), making this knowledge on wetland loss inaccessible and possibly not peer reviewed. Working in the Alberta aspen parklands, Schick (1972) made use of township survey plans, government drainage districts, and aerial photographs to assess wetland change between 1900 and 1970 and found a 61% loss of wetland area. Estimates of 40–70% loss are below this study's estimate of 72.2% total wetland loss by number and 78.8% total loss by area, which may be due to continued wetland loss since previous estimates were made.

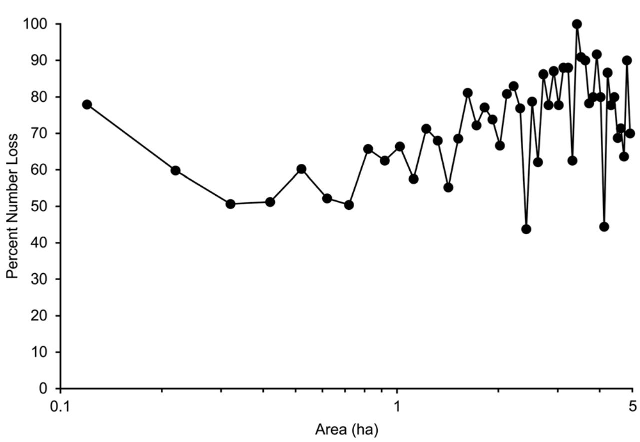


Fig. 7 Percent total wetland loss by number for different wetland sizes (includes permanent loss and temporary loss)

Some estimates of wetland loss are based on wetland inventories derived from high-resolution LiDAR data. Van Meter and Basu (2015) generated a historical wetland inventory in the Des Moines lobe portion of the Prairie Pothole Region in Iowa using a 1 m DEM and 1:15,840 scale hydric soil data (with a coarser 0.04 ha minimum mapping unit compared to the 0.02 ha minimum mapping unit in this study). Using existing wetland estimates based on the 1:24,000 scale U.S. National Wetland Inventory, the authors estimated a 90% historical loss of wetland area, which is more comparable to the loss estimates found for the Nose Creek watershed.

Other estimates of wetland loss are based on changes in wetlands between two points in time. For example, Goodman and Pryor (1972) used aerial photographs, waterfowl capability maps, agricultural capability maps, soil surveys, and field

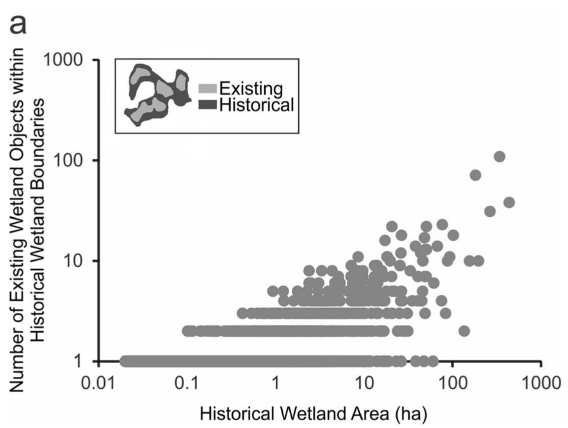


Fig. 8 (**a**) The number of existing wetland objects within historical wetland boundaries as a function of historical wetland area in logarithmic-logarithmic scales, and (**b**) the sum of existing wetland area within historical wetland boundaries as a function of historical wetland area in logarithmic-logarithmic scales. The line shows a 1:1 relationship

surveys to sample 600 random quarter sections and found a 13% net loss of wetland area between 1940 and 1970 in areas of Alberta, Saskatchewan and Manitoba. More recently, Watmough and Schmoll (2007) surveyed transects across the Canadian Prairies and estimated wetland loss at 5% over a 17 year period (between 1985 and 2001). Similarly, Clare and Creed (2014) used wetland inventories generated from aerial photographs over an 11 year period (between 1999 and 2009) and estimated 242 wetlands totalling 71 ha lost in the Beaverhill watershed of central Alberta. Loss estimates between two points in time do not reflect historical loss, and therefore it is not possible to state the relative importance of the wetland loss estimates from these studies – they only confirm that wetland loss is continuing, and emphasize the need for standardized historical reference conditions.

Total wetland loss has included the preferential loss of both small (< 0.3 ha) and large (> 0.8 ha) wetlands, leading to a homogenization of wetland sizes with historical wetlands disintegrating into smaller fragments. The selective loss of wetland sizes has also been found in other agricultural settings, including northern Iowa (Miller et al. 2012; Van Meter and Basu 2015) and Indiana (Christensen et al. 2016). This homogenization has implications for wetland ecosystem functions (Creed et al. In Press). For example, small and isolated wetlands such as those found in the Prairies support species richness and biodiversity (Semlitsch and Bodie 1998; Leibowitz 2003; Scheffer et al. 2006). Waterfowl populations are particularly vulnerable to wetland loss and size homogenization, as they depend on a variety of wetland types and sizes throughout their life cycles (Kantrud et al. 1989). Water storage (Miller and Nudds 1996; Gleason et al. 2007) and water purification (carbon sequestration, nitrogen removal, and phosphorus retention) functions also vary with wetland size and connectivity (Whigham and Jordan 2003; Marton et al. 2015; USEPA 2015).

Table 2 Accuracy assessment for ditch-drained or undrained wetlands

Class	Reference totals	Classified totals	Number correct	Producer's accuracy (%)	User's accuracy (%)
Ditch-drained wetland	78	100	65	83.3	65.0
Undrained wetland	122	100	87	71.3	87.0
Totals	200	200	152	-	-

Proportion of Easily Restorable Wetlands

The automated method for mapping restorable wetlands was based on high-resolution DEMs and targeted a specific mechanism of wetland loss. Targeting ditch-drained wetlands identifies wetlands that can be easily restored. The method developed is simple and replicable with an acceptable overall accuracy of 76.0%. However, the user's accuracy for identifying drained wetlands (65.0%) can be a barrier for restoration practitioners, as it still leaves false positives to filter through with misclassification due to confusion with other concave features, including furrows. While the method does not distinguish between natural and human made ditch features, natural drainage pathways from fill and spill can be human modified acting in the same manner as drainage ditches (Watmough and Schmoll 2007) and are therefore appropriate to include in the inventory for further investigation. In the Nose Creek watershed, 9.7% of historical wetland area, representing 1,588 wetlands and 63,217 ha, were drained in the agricultural area of the watershed. The spatial and temporal variability of the few estimates of drained wetlands available are very difficult to compare directly to the estimate found for the Nose Creek watershed. For example, within the Prairie Pothole Region, Goodman and Pryor (1972) found 19% of wetland area had been affected by drainage or partial filling between settlement and 1970, and Schick (1972) found 34% of wetland area had been lost to drainage between 1900 and 1970. These estimates are higher than those for the Nose Creek watershed. The deficiency of published estimates emphasizes the need for automated tools that standardize the method of identifying lost and restorable wetlands.

Implications for Wetland Management

The wetland loss estimates observed in this study – a total loss of 72.2% of wetlands by number and 78.8% by area – will have been accompanied by a loss of wetland ecosystem functions and associated services (Zedler and Kercher 2005). Restoration of the drained wetlands is likely to lead to a substantial recovery of wetland ecosystem services in the Nose Creek watershed. Of the 11,279 non-riparian agricultural wetlands in the watershed, approximately 14.1% were identified as drained, equivalent to 39.9% of the non-riparian agricultural wetland area. However, while restorable wetlands represent almost 9.8% of the historical inventory area, this proportion is still relatively small.

While some wetland loss was accounted for as permanent (i.e., filled wetlands) or restorable (i.e., ditch-drained), the temporary loss of 61.1% by number remains an untapped resource. Automated methods for detecting other types of restorable wetlands, such as cultivation and subsurface drainage, should be pursued. For example, Naz et al. (2009) have used high-resolution aerial imagery to map subsurface drainage using edge detection filters. Turner et al. (1987) found that an approximately 40% of wetlands between 1981 and 1985 in the Canadian Prairies were affected by cultivation, suggesting a high potential for recovery among cultivated wetlands. Cultivated wetland basins can be identified through land use classification (e.g., Fenstermacher et al. 2014), and subsurface drained wetlands can be identified using edge detection filters of on high-resolution aerial imagery (Naz et al. 2009). If the aim of wetland management strategies is to restore historical watershed scale patterns in wetland distribution, the loss of both small and large wetlands should be considered simultaneously in restoration efforts, as restoration efforts themselves can contribute to landscape homogenization when specific wetland types or sizes are favoured (Bedford 1999; Miller et al. 2012).

Conclusion

Wetland inventories are the foundation of sustainable wetland management. The automated wetland inventory methods presented here are simple, transparent and reproducible.

These automated methods coupled with statistical power law techniques for estimating historical, permanently and temporarily lost wetlands do not require historical data or laborious aerial photograph or satellite image interpretation. The digital terrain analysis method for estimating restorable ditch-drained wetlands could successfully target a mechanism of wetland loss, creating tangible launch points for restoration efforts. The generated inventories facilitate a multi-faceted view of the fate of wetlands, estimating the extent of historical, permanently and temporarily lost, restorable, and existing wetlands. An understanding of this wetland change on a landscape scale, together with insights on changes in their distribution, can guide both protection and restoration efforts, and shape wetland and watershed management goals. Further work will focus on extending the automated methods to quantify other types of restorable wetlands.

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