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A Case Study in Large-scale Wetland Restoration at Seney National Wildlife Refuge, Upper Michigan, U.S.A.

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ABSTRACT.—A large wetland drainage project was initiated in 1912 near the town of Seney, Michigan, U.S.A. This project included the construction of a series of ditches through a large peatland to drain the land for agricultural use. The largest of these ditches was the 35 km long Walsh Ditch. Much of the drained wetland affected by the Walsh Ditch is now managed by the U.S. Fish and Wildlife Service as part of Seney National Wildlife Refuge. Between 2002–2003, nine large earthen ditch plugs were installed along a 4.5 km section of the ditch in an attempt to restore the hydrological and ecological integrity of the approximately 1400 ha of wetlands and streams. This study explores the effects that the ditch plugs had on the hydrology and vegetation structure of the adjacent landscape 8y later. Plot level measurements (707 m^2 plots) of hydrology and vegetation, combined with an analysis of land cover change using aerial imagery, indicated that the ditch plugs had been successful in altering the hydrology and vegetation over portions of the area. Mortality of upland tree species more typical of xeric conditions and colonization by typical wetland species indicated that these sites should continue to develop into wetland ecosystems. Land cover change analysis showed an increase in wetland area of 152 ha. The areas of change were concentrated near the plugged ditch and near a large anthropogenic pool.

INTRODUCTION

In North America, wetlands have experienced a drastic decline in area and function over the last century (Moreno-Mateos *et al.*, 2012). In the United States, greater than 45 million ha of wetlands, representing more than half the wetlands present prior to European settlement, have been drained (Dahl and Johnson, 1991; Dahl, 2006). Wetlands have often been seen as places of little value, and drainage was promoted by polices such as the *Swamp Land Act* of 1849 (Dahl and Allord, 1997). By the 1940s, the U.S. Federal Government was providing free technical assistance for (and cost sharing of) wetland drainage projects for agriculture. By the 1980s, there was a shift in policy, and the U.S. Government began actively discouraging the draining of wetlands.

Numerous federal wetland restoration programs have been established by the U.S. Fish and Wildlife Service and the U.S. Department of Agriculture Natural Resources Conservation Service to reverse wetland decline on a national scale, with many states following suit (*e.g.,* Moreno-Mateos *et al.*, 2012). Unfortunately, a meta-analysis of 621 wetland restoration efforts by Moreno-Mateos *et al.* (2012) reported that the biological

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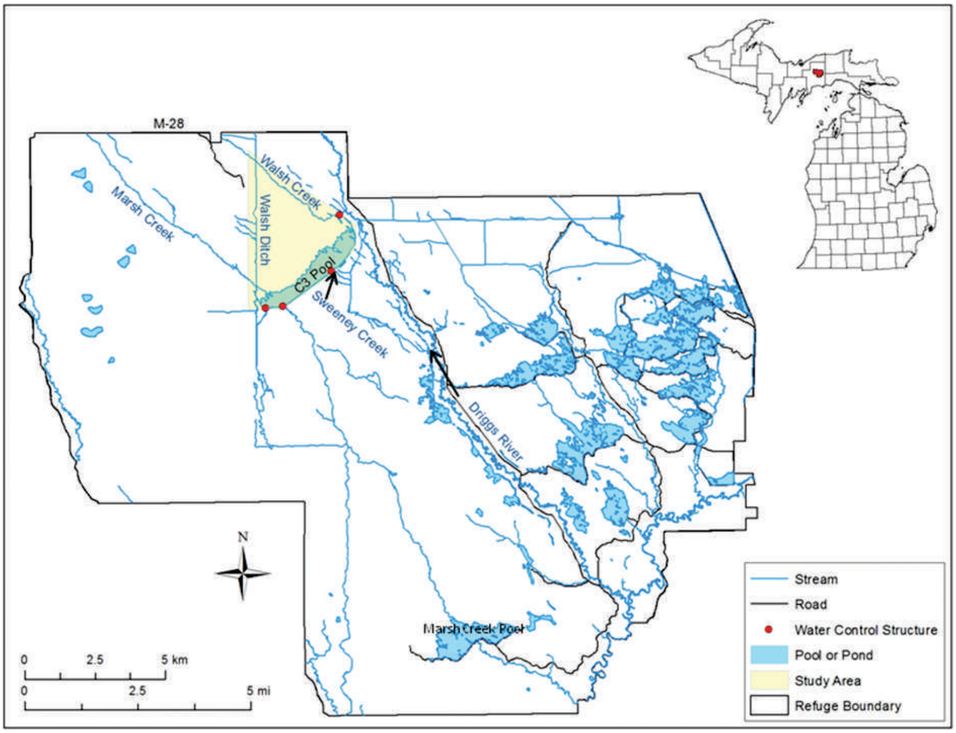

FIG. 1.—Map of Seney National Wildlife Refuge, located in Schoolcraft County in Michigan's Upper Peninsula. The study area is south of Highway M-28 and is bordered by Walsh Ditch to the west, C-3, Pool to the south and Walsh Creek to the east

structure (*e.g.,* species richness, abundance) and biochemical function (*e.g.,* carbon and nitrogen cycling, phosphorus storage, organic matter accumulation) of restored wetlands were, on average, more than 20% below reference sites. Moreover, findings from Moreno-Mateos *et al.* (2012) indicated that the size and spatial context of restoration attempts are both critical for attaining some degree of success with larger wetlands restoration attempts (>100 ha) in more natural ecoregional settings having a greater likelihood of producing desired results.

In 1912 the 35 km long approximately 6 m wide and 2 m deep Walsh Ditch was created in what was to become, in 1935, Seney National Wildlife Refuge (NWR). It diverted water away from local creeks, lowered the water table of neighboring wetlands, promoted peat subsidence, and altered the vegetation structure of the general vicinity (Wilcox *et al.*, 2006, Fig. 1). The initiation of the 1997 *Refuge Improvement Act* and the subsequent *Biological Integrity Policy* (2001) encouraged current land managers at Seney NWR and other refuges to consider restoring historical conditions, if possible (Schroeder *et al.*, 2004; Meretsky *et al.*, 2006). This change in policy allowed for an initiative to restore the hydrology within the greater than 10,000 ha affected by the Walsh Ditch. The goal of the restoration project was to restore the ''hydrology and ecological integrity of the wetlands and streams'' affected by the Walsh Ditch (USFWS, 2001), but no specific benchmarks were originally set to assess the success of the restoration project.

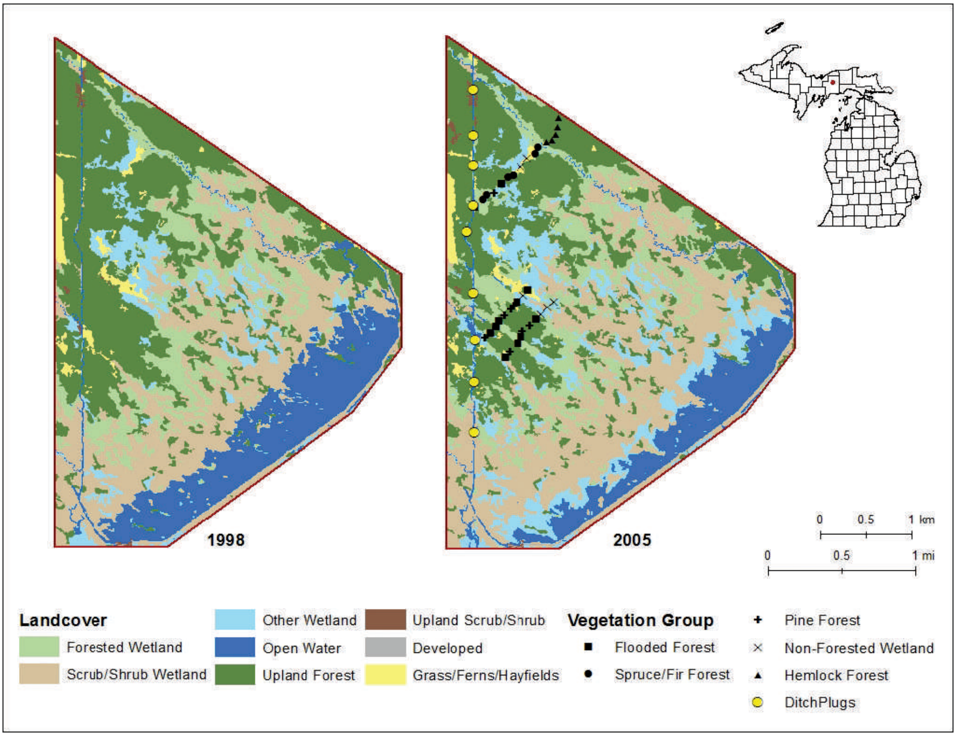

FIG. 2.—Study sites were located in Seney National Wildlife Refuge. The yellow dots show the location of the ditch plugs that divert water from the ditch to the adjacent landscape. The different shaped black markings represent the 36 vegetation plots. Each shape represents a different vegetation grouping as defined by cluster analysis. Groups were defined in PC-ORD 5.32 (McCune and Mefford, 2006) and plotted in ArcGIS version 9.3 (ERSI, 2008). The map was digitized using 1998 and 2005 orthophotos of the area. (Source of aerial photograph: USDA-FSA Aerial Photography Field Office. Datum/ Projection: NAD_1983_UTM_Zone_16N)

Between 2002–2003, nine earthen ditch plugs were installed in a portion of Walsh Ditch (Wilcox *et al.*, 2006, Figs. 1, 2). The purpose of the plugs was to prevent water from flowing down the ditch and to restore the hydrology of the land to the east of the ditch. Ideally, the hydrology would change sufficiently to increase the area covered by wetland species and reduce the upland area covered by trees more typical of xeric conditions (*e.g.*, red pine, *Pinus resinosa* Ait.). However, the goal was not to remove all upland vegetation because the refuge historically was a heterogeneous mosaic of open peatlands, lowland swamps, and upland forests (USFWS, 2009). Ideally, trees unsuitable for wetland sites would die in low-lying areas and be replaced by wetland herbaceous or woody plants. Over time this area should convert back to a functioning peatland.

Previous research in the Walsh Ditch area focused on the water channels and the area immediately surrounding those channels (Sweat, 2001; Kowalski and Wilcox, 2003; Wilcox *et al.*, 2006). This earlier work identified the impacts of altered hydrology on the surrounding vegetation. However, the ditch plugs and other control structures on the area have affected a much larger area (approximately >5000 ha) and little is known how this larger area has responded to the ditch plugs. Preliminary observations indicated that the water table in the ditch was elevated relative to preplug levels and water was flowing from the ditch to the surrounding landscape (Wilcox *et al.*, 2006), but a more complete analysis of the impacts of the ditch plugs on the local hydrology and vegetation communities was lacking.

Hydrology is one of the driving factors behind wetland structure, function, and species composition, and it is the main factor affecting restoration success (Richter *et al.*, 1996; Duranel *et al.*, 2007; Zhang and Mitsch, 2007). Different types of wetlands have different patterns of saturation and inundation throughout the year (hydroperiod). Therefore, it is important that the hydroperiod of the restored area coincides with the particular wetland type. This requires an understanding of the target hydroperiod and site conditions and hydrological monitoring.

Once the hydrology is restored, the site would ideally be recolonized by the appropriate native wetland flora. Where there is an appropriate hydrologic condition and seed bank, or nearby populations to aid in natural recruitment, wetland vegetation may partially develop on its own without active management (*e.g.,* Campbell and Bergeron, 2012; Hedberg *et al.*, 2012). However, often active management is needed to introduce native plants by transferring plant fragments, rhizomes, and seeds (*e.g.,* Ferland *et al.*, 1997; Rochefort *et al.*, 2003).

Many studies have focused on restoring harvested peatlands (*e.g.,* Rochefort *et al.*, 2003; Waddington *et al.*, 2003; Lavoie *et al.*, 2005; Graf *et al.*, 2008) or restoring peatlands that have been abandoned after being drained for agriculture or forestry (*e.g.,* Cooper *et al.*, 1998; Richert *et al.*, 2000; Mälson and Rydin, 2006; Duranel *et al.*, 2007; Thogmartin *et al.*, 2007). The drastic change in hydrology and vegetation in the land adjacent to Walsh Ditch makes restoration to historical conditions a challenge. Further affecting restoration efforts has been the impacts of the 1976 Seney Fire, which covered approximately half of the refuge and consumed an unknown amount of drained peat adjacent to the ditch (Anderson, 1980). Also, most ditch restoration projects (*e.g.,* Cooper *et al.*, 1998) have involved small ditches, whereas the Walsh Ditch is comparatively large. The goal of our work was to quantify the changes to the surrounding landscape since the installation of the ditch plugs and control structures. The objectives were to 1) determine if hydrology in the area near the ditch plugs is conducive to the creation of wetlands and 2) document how the restoration activities have changed vegetation communities throughout the study area.

METHODS

STUDY SITE AND RESTORATION WORK

Seney NWR is relatively flat with a southeast slope of 1.9 m km^{-1} and is part of the 3797 km^2 Manistique Watershed (Sinclair, 1959; USFWS, 2009). Two-thirds of Seney NWR is covered by a layer of peat up to 2 m in depth (Wilcox *et al.*, 2006). Underlying deposits include sand over Ordovician sandstone, limestone, and dolomite. The climate is strongly influenced by Lake Superior and Lake Michigan, with an annual precipitation of 810 mm. Precipitation is fairly evenly distributed throughout the year with Sep. and Oct. being the wettest months (1981–2010 averages = 94 mm and 98 mm for Sep. and Oct., respectively) and Feb. being the driest (1981–2010 average = 53 mm) (Fig. 3). Temperatures in the area range from −37 C to 36 C, with an average temperature of 5 C (Wilcox *et al.*, 2006; USFWS, 2009).

Seney NWR is covered by open peatlands, lowland swamps, and upland forests (USFWS, 2009). The peatlands are generally dominated by sedges (*Carex* L. spp.) and shrubs (*Salix* spp. L., *Alnus* spp. Mill, and *Betula pumilla* L.) interspersed with sandy knolls dominated by

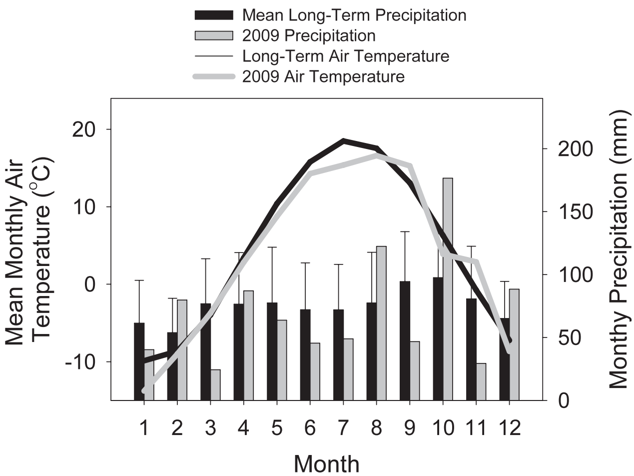

FIG. 3.—Mean daily air temperature and monthly precipitation in 2009 compared to the mean temperature and mean precipitation from 1981 to 2010. Measurements were taken at a weather station Marquette, MI located approximately 120 km from the field sites. Error bars on the precipitation represent one standard deviation

red pine and eastern white pine (*Pinus strobus* L., Heinselman, 1965). The Seney peatlands are the largest of their type in Michigan (Heinselman, 1965; Wilcox *et al.*, 2006). Within the study site, peatlands cover approximately 80% of the area. Upland areas are generally mixed hardwood forests with multiple tree species including American beech (*Fagus grandifolia* Ehrh.), sugar maple (*Acer saccharum* Marsh.), yellow birch (*Betula alleghaniensis* Britton), red pine, eastern white pine, jack pine (*Pinus banksiana* Lamb.), black spruce (*Picea mariana* (Mill.) B.S.P.), and balsam fir (*Abies balsamea* (L.) Mill.) (USFWS, 2009). Besides hydrology, fire, and human disturbances interact with surficial geology to influence the plant communities at Seney NWR (Drobyshev *et al.,* 2008a, b).

CLIMATE AND WEATHER DATA

Mean monthly air temperature and precipitation from 1981–2010 was compared with weather in 2009. We used weather data from a station located in Marquette, MI (approximately 120 km from the field site). The weather station was the closest site with a long-term data set. The data were provided by the Michigan State Climatologist office (Department of Geography, Michigan State University, Lansing, U.S.A.).

RESTORATION ACTIVITIES

Between 2002–2003, nine earthen ditch plugs were installed in a portion of Walsh Ditch (Wilcox *et al.*, 2006, Figs. 1, 2). In most instances, the soil used to construct the plugs was the same soil removed to create the Walsh Ditch. In some instances, the plugs were reinforced with metal sheet-piling. The northern-most ditch plug redirects water to the historic Walsh Creek Channel (Wilcox *et al.*, 2006, Figs. 1, 2). The remainder of the ditch plugs divert water from the ditch into the adjacent landscape where it flows primarily to the southeast (Wilcox *et al.*, 2006, Fig. 2).

Before the Walsh Ditch restoration project began and ditch plugs were installed, the majority of the discharge from C-3 Pool was sent to lower Walsh Ditch (Fig. 1). Now the majority of springtime discharge from the pool flows to Marsh Creek (3.11 m^3/s) (see Fig. 1 for place names and locations), and the remainder of the outflow goes through an old water control structure at Sweeny Creek (0.80 m^3/s) (Wilcox *et al.*, 2006). When springtime discharge exceeds the capacity of the other channels, water is diverted to the Driggs River through a new water control structure in Walsh Creek northeast of C-3 Pool. This has restored springtime flow to the Driggs River (Wilcox *et al.*, 2006). Under normal circumstances, no water is released to the portion of Walsh Ditch south of C-3 Pool, thereby significantly reducing the linear flow of water leaving this subwatershed.

VEGETATION COMPOSITION AND STRUCTURE

Vegetation composition and structure were determined at 36, 15 m-radius circular plots (707 m^2) that were installed in May 2009 (Figs. 2–4). The 36 plots were selected based on aerial photographs and past vegetation sampling. The 16 northern plots were placed according to the past work of W. Loope and A. Lucas (US, Geological Survey). The remaining 20 plots were randomly located along two transects. The transects were placed in areas that could be clearly identified as effected by flooding because of the ditch plugs. The plots were used to identify if wetland plants were colonizing flooded areas near the ditch and if the hydrology in these locations were conducive to supporting wetlands. The center of each plot was recorded using global positioning system (GPS) technology. Within each plot, the diameter at breast height (1.5 m, hereafter dbh) of all trees $\geq$2.54 cm were measured by species to estimate the basal area. Four 3 m radius sub-plots (28 m^2), and 12, 1 m^2 sub-plots, were used to inventory the shrub and herbaceous vegetation, respectively (Braun-Blanquet, 1932). The data from these plots were not used to determine changes over the entire study area. To determine changes over the entire study area we used aerial photographs (see ''Land cover change analysis,'' below).

Coarse woody debris (CWD) was measured following the methods of Waddell (2002). CWD was measured along two 30 m intercept transects running N–S and E–W through the center of the plot. All CWD at least 1 m in length, with a minimum diameter of 5 cm, was included in the sampling. For forked trees, where both branches crossed a transect, the main fork (the one with the larger diameter at the fork) was measured for the length of the log, and the smaller branch was measured starting at the fork. Each section had to meet the length and diameter criteria used for individual logs as discussed above. Where a log crossed a transect multiple times, or crossed multiple transects, it was recorded separately for each intersection. Each piece of CWD was measured for length, diameter, wood color, condition of twigs, and the presence of invading roots (Waddell, 2002). Decay class for each log was also recorded (Waddell, 2002), and logs with a decay class of five (*i.e.*, no structural integrity remained) were not included in the survey. For logs with portions in different decay classes, only sections with a decay class of $\leq$4 were measured.

To determine what types of vegetation communities are developing within the study site, all plots were classified into community groups using hierarchical cluster analysis of vegetation data with Sorensen distance measure and flexible beta ($\beta = -0.25$) using PC-ORD 5.32 (McCune and Mefford, 2006). Indicator species analysis was used to find the optimum number of groups. The optimal separation was determined by averaging the P-values obtained from Monte Carlo analysis and by totaling the number of species with significant indicator species values ($P < = 0.05$) (Chimner *et al.*, 2010). A significant indicator species value suggests there is a high probability that species is associated with a given group. Ordination of vegetation and environmental variables was carried out using Nonmetric Multidimensional Scaling (NMDS) to determine factors that best explained the separation of community groups.

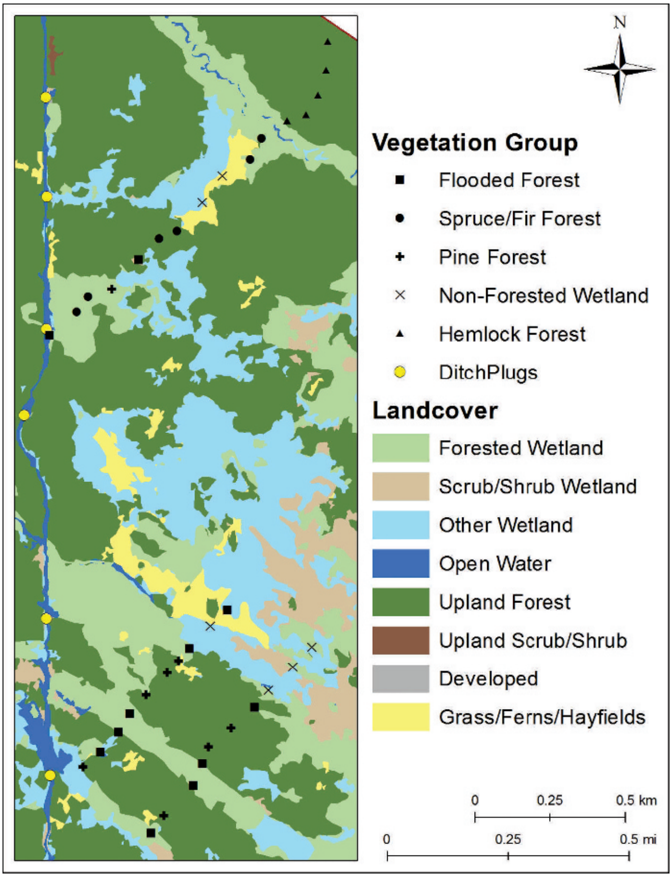

FIG. 4.—Location of 36 study plots by vegetation group as defined by cluster analysis. Groups were defined in PC-ORD 5.32 (McCune and Mefford, 2006) and plotted in ArcGIS version 9.3 (ERSI, 2008)

Lastly, data from the 16 northern-most plots were compared to 2002 pretreatment tree measurements collected by W. Loope and A. Lucas (unpublished). In contrast to our measurements, Loope and Lucas collected their data on 50 m × 30 m (1500 m^2) vegetation plots. Tree basal area and number of stems ha^{-1} were compared between the pretreatment study (*i.e.*, Loope and Lucas) and our data using two-tailed paired *t*-tests.

HYDROLOGY

At the center of each of the 36 plots, we installed a groundwater well (1 m deep) for hydrologic monitoring. Wells were constructed of slotted 3.8 cm or 5 cm polyvinyl chloride (PVC) pipe with caps at both ends and were carefully inserted into an augured hole with a slightly smaller diameter than the well pipe. The wells were slotted for their full length to the surface (1 m). Wells were installed in early May 2009 and were monitored biweekly through the first week of Sep. 2009. Water table measurements were made using a hand operated weighted measuring tape with a water sensor (ET-89, Hunter-Keck, Williamston, Michigan, U.S.A.). The purpose of the hydrologic measurements was to determine if the hydroperiod in the restored area was similar to the hydroperiod of unditched peatlands in the region. Three unditched peatlands located east of the Marsh Creek Pool (46°11′W, 86°01′N) on a poor fen peatland complex were used as a benchmark for unditched peatlands in the area. Depth to water table at three peatlands were monitored with Solinst Levelogger nonvented pressure transducers set at a recording interval of 1 h placed in slotted wells inserted to the mineral soil. A Solinst Barologger was installed to provide barometric compensation for all Leveloggers.

LAND COVER CHANGE ANALYSIS

To assess broader scale changes in the study site, a geographic information system (GIS) was used to map land cover prerestoration (1998) and postrestoration (2005, 2–3 y after ditch plug installation). Analysis was done using digital aerial imagery in ArcGIS version 9.3 (ERSI, 2008) at a scale of 1:40,000 with a pixel size of 1 m × 1 m. The system used for land cover classification was a simplified version of the National Vegetation Classification System (NVCS) (FGDC, 1997, Table 1). The minimum mapping unit was 15 m^2. Change in land cover between years was analyzed using matrix analysis in ERDAS Imagine version 10.1 (ERDAS, 2010). An x/y tolerance of 2 m was used to eliminate slivers resulting from the imperfect alignment of the different maps. The spatial patterns of change and the total area of change were analyzed.

RESULTS

WEATHER IN 2009

The regional weather center in Marquette, Michigan indicated the weather in 2009 was typical compared to the long term average for most of the year (Fig. 3), although total precipitation for 2009 was 853 mm, 52 mm less than the long term average (1981–2010). Precipitation was within two standard deviations of the mean for all months except Oct. Temperatures were also similar to the long term mean, with 8 mo falling within one standard deviation of the long term mean.

CURRENT VEGETATION AND HYDROLOGICAL CONDITIONS

We identified 119 plant species on the 36 plots, including 15 trees, 20 shrubs, and 84 herbaceous plants. No exotic species were encountered. The number of species per plot ranged from 6 to 43, with a mean of 21.4 (SE ± 1.1). The most common tree species were red pine, red maple (*Acer rubrum* L. var. *rubrum*), quaking aspen (*Populus tremuloides* Michx.), and eastern white pine. Serviceberry (*Amelanchier* spp. Medik.), spirea (*Spirea* spp. L.), and willow were the most common shrub species. Sedge species, bluejoint (*Calamagrostis canadensis* (Michx.) P. Beauv.), woolgrass (*Scirpus cyperinus* (L.) Kunth), bristly dewberry (*Rubus hispidus* L.), and starflower (*Trientalis borealis* Raf.) were the most common herbaceous plants.

Hierarchical cluster analysis and indicator species analysis resulted in an optimal solution of five vegetation community types (Fig. 5, Table 1). Group 1 consisted of flooded forested (Flooded Forests) areas with a high stable water table that ranged from 24.9 cm above the surface to 24.5 cm below the surface (Fig. 6), abundant CWD, and a high herbaceous cover. Group 1 has a relatively low living tree basal area and high dead standing (snag) basal area. Group 2 consisted of spruce/fir forests (Spruce/Fir Forests) that has a high living tree basal area, high basal area of small trees (<10 cm dbh), low snag basal area, low CWD, and a deep water table that ranged between 9 and 100 cm below the surface (Fig. 6). Group 3 consisted of pine forests (Pine Forests) that had high living tree basal area, low snag basal area, and high CWD. The water table at these sites was generally below the surface but not very deep

TABLE 1.—Defining species for each community type. Based on a Monte Carlo test of permutations, all defining species have a indicator probability value of less than 0.05 and are present in at least 70% of all plots in the group

Group	Indicator value probability
Group 1: Flooded Forested Areas	
Scirpus cyperinus (L.) Kunth	3.4×10^{-3}
Solidago altissima L.	1.4×10^{-2}
Fragaria sp.	2.0×10^{-2}
Salix	2.7×10^{-2}
Group 2: Spruce/Fir Forest	
Deschampsia flexuosa (L.) Trin.	2.0×10^{-4}
Vaccinium angustifolium Aiton	2.0×10^{-4}
Vaccinium myrtilloides Michx.	2.0×10^{-4}
Gaultheria procumbens L.	4.0×10^{-4}
Abies balsamea	2.4×10^{-3}
Corylus americana Walter	5.4×10^{-3}
Picea glauca	1.2×10^{-2}
Cladina (Nyl.) Nyl.	2.1×10^{-2}
Polytricum sp.	2.6×10^{-2}
Viburnum lentago L.	3.4×10^{-2}
Abies balsamea	4.6×10^{-2}
Group 3: Pine Forest	
Trientalis borealis Raf.	2.0×10^{-4}
Pinus strobus	8.0×10^{-4}
Amelanchier spp. Medik	1.8×10^{-3}
Rubus hispidus L.	2.0×10^{-3}
Pinus resinosa	3.0×10^{-3}
Pinus strobus (dbh > 10 cm)	1.2×10^{-2}
Populus tremuloides	2.5×10^{-2}
Prunus virginiana L.	2.8×10^{-2}
Group 4: Non-Forested Wetland	
Spirea	2.0×10^{-4}
Carex sp.	2.6×10^{-2}
Betula pumila L.	3.7×10^{-2}
Group 5: Hemlock Forest	
Cornus canadensis L.	2.0×10^{-4}
Dryopteris sp.	2.0×10^{-4}
Coptis trifolia (L.) Salisb.	2.0×10^{-4}
Oxalis montana Raf.	2.0×10^{-4}
Maianthemum canadense Desf.	8.0×10^{-4}
Clintonia borealis (Aiton) Raf.	1.4×10^{-3}
Tsuga canadensis	1.6×10^{-3}
Huperzia lucidula (Michx.) Trevis.	2.0×10^{-3}
llex verticillata (L.) A. Gray	2.4×10^{-3}
Lonicera canadensis	7.8×10^{-3}
Acer rubrum	7.8×10^{-3}
Lycopodium obscurum L.	1.5×10^{-2}
Osmunda regalis L.	1.5×10^{-2}
Ribes triste Pall.	1.5×10^{-2}
Betula alleghaniensis	1.7×10^{-2}

TABLE 1.—Continued

Group	Indicator value probability
Rubus pubescens Raf.	3.0×10^{-2}
Aralia nudicaulis L.	3.1×10^{-2}

and ranged from 6.0 cm above the surface to 29.8 cm below the surface (Fig. 6). Group 4 was a nonforested wetland (NonForested Wetland) characterized by high water tables that ranged between 16.7 cm above the surface to 23.0 cm below the surface (Fig. 6). Plots in this group had a low living tree basal area, higher snag basal area, low CWD, and high shrub cover. Group 5 consisted of an eastern hemlock (*Tsuga canadensis* L.) forest (Hemlock Forests) that had a high living tree basal area, high CWD, high herbaceous cover, and deep water table that ranged between 9.4 cm below the surface to 73.8 cm below the surface.

The NMDS resulted in a 3-dimensional solution with a cumulative r^2 of 0.88 and a final stress of 10.4 (Fig. 7). The first axis was most correlated with percent shrub cover and percent herbaceous cover ($r^2 = 0.13$). The second axis was most correlated with basal area of large trees (≥10 cm dbh), the amount of CWD and percent shrub cover ($r^2 = 0.20$). The third axis was most correlated with average water table, basal area of large trees (≥10 cm dbh), and percent shrub cover ($r^2 = 0.54$).

CHANGES IN TREE COVER

At sites with pre (2002) and post (2009) treatment data (northern 16 plots—Fig. 2), we found no change in the basal area (P = 0.92) and stem density (P = 0.74) of trees ≥10 cm: 24.0 m^2 ha^{-1} and 444 stems ha^{-1}, respectively, in 2009, vs. 23.7 m^2 ha^{-1} and 410 stems ha^{-1}, respectively, in 2002. There were also no differences in the basal area (P = 0.12) and stem density (P = 0.06) of smaller trees between pre and postrestoration. The basal and stem density of small trees <10 cm were 1.6 m^2 ha^{-1} and 694 stems ha^{-1}, respectively, in 2009, and 0.96 m^2 ha^{-1} and 397 stems ha^{-1}, respectively, in 2002.

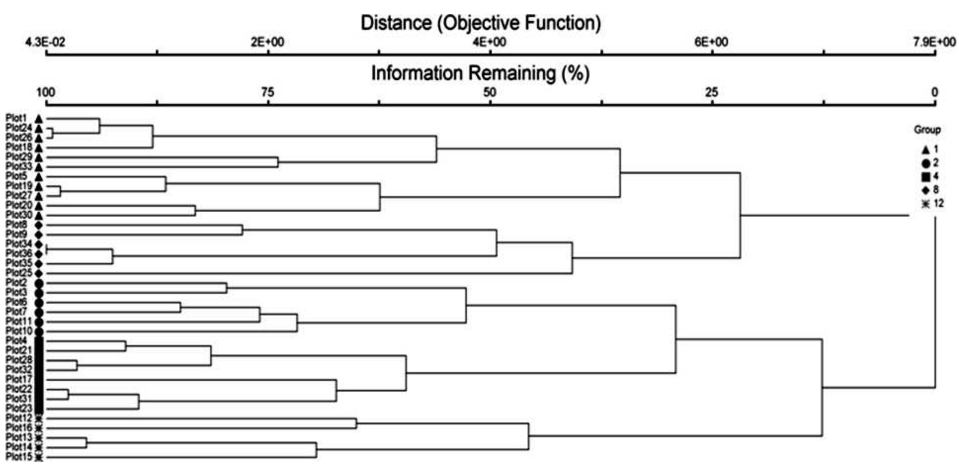

FIG. 5.—Hierarchical cluster analysis dendrogram used to sort community groups with a Sorensen distance measure and flexible beta (β = −0.25) using PC-ORD 5.32 (McCune and Mefford, 2006)

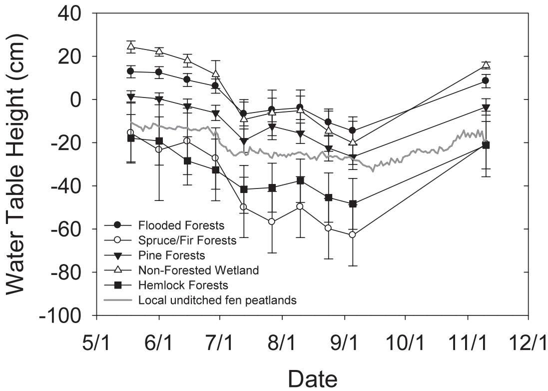

FIG. 6.—Mean water table for all plots in each group. Data are from manual water table readings. Error bars represent standard errors. The grey line represents the mean water table height from three unditched peatlands located within Seney NWR

LAND COVER CHANGE

Land cover change analysis, from imagery 7 y apart, showed increases in scrub/shrub wetland area from 380 ha to 428 ha, forested wetland area from 194 ha to 218 ha, and other wetland area from105 ha to 186 ha (Figs. 8 and 9). The total increase in wetland area was

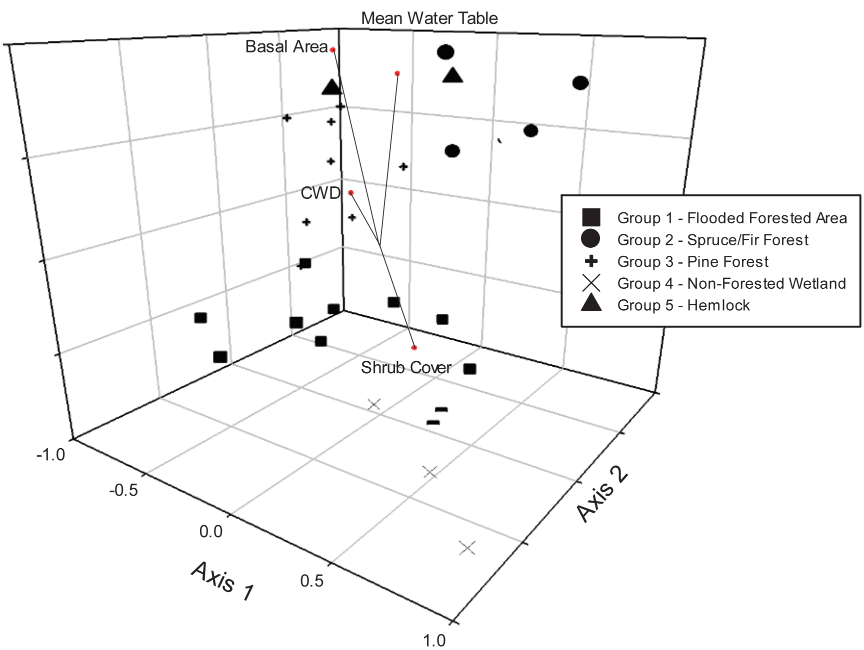

FIG. 7.—Nonmetric Multidimensional Scaling ordination of plots grouped by community type (Groups 1 to 5). Vectors represent important environmental variables: CWD—volume of coarse woody debris/ha, Basal Area—basal area of trees >10 cm DBH ha^{-1}, Shrub Cover—total shrub cover for plot

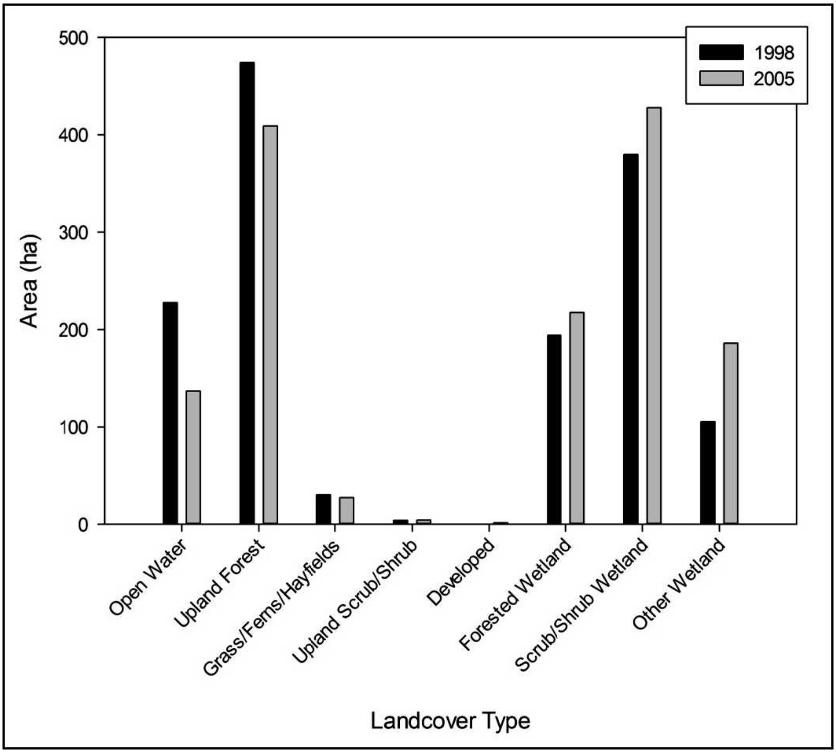

FIG. 8.—Change in total area of each cover type between 1998 and 2005. Areas were calculated from digitized land cover maps in ArcGIS version 9.3 (ERSI, 2008)

152 ha, or 11% of the study area. Most of the increase in wetland area occurred where C-3 Pool had receded (62 ha), but flooding of pine forests located near the ditch accounted for another 14 ha. There was a slight increase in developed area from 0.6 ha to 1.6 ha as a result of the earthen ditch plugs. Open water decreased from 228 ha to 137 ha, most of which was a result of the open water of C-3 Pool converting to other wetland or scrub/shrub wetland (62 ha and 36 ha, respectively). Upland forest also decreased from 474 to 409 ha. Most of this change was a result of flooding without a shift in vegetation type, leading to a classification of forested wetland (47 ha). Upland forest areas that were flooded and no longer supported woody vegetation were classified as other wetland (8.6 ha). Upland scrub/shrub increased slightly from 3.8 ha to 4.1 ha. The area of grass/ferns/hayfields decreased slightly from 30 ha to 27 ha. The total decrease of upland cover types was 68 ha, or 4.8% of the study area.

DISCUSSION

PLOT-SCALE HYDROLOGY AND VEGETATION

Wetland restoration often depends on restoring the hydrological condition and related patterns and processes (*e.g.*, Zedler, 2000; Moreno-Mateos *et al.*, 2012). The hydroperiod for all the measurement sites show that the water tables remain close to the soil surface throughout the growing season. The Flooded Forests (Group 1), NonForested Wetlands

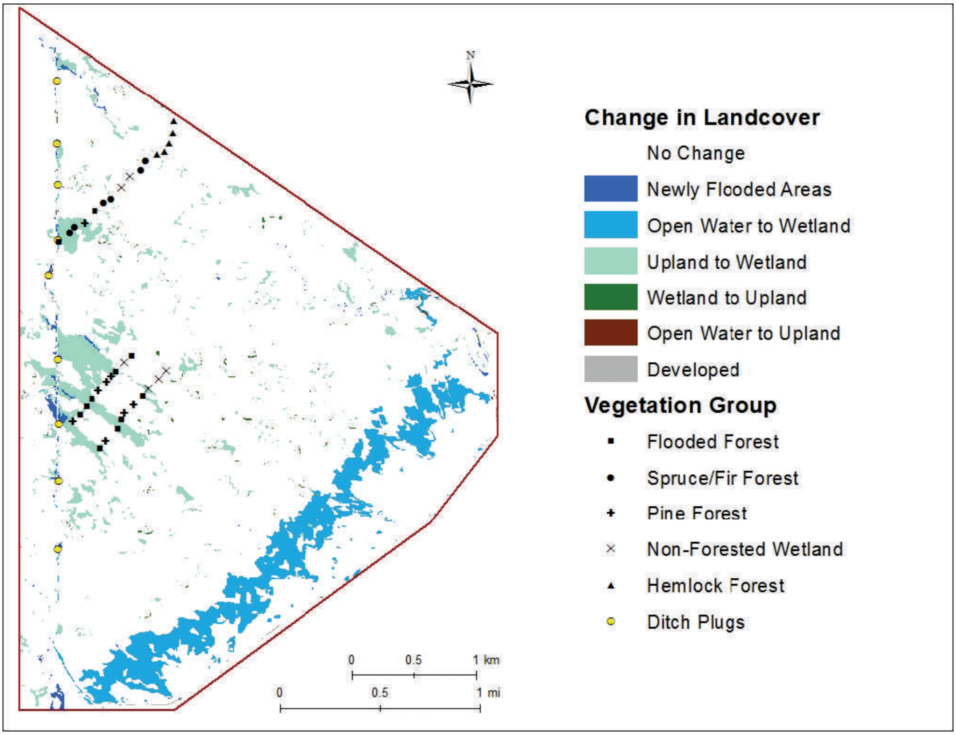

FIG. 9.—Map of land cover change from 1998 to 2005 at the study site in Seney National Wildlife Refuge. The 1998 map was digitized using a mosaic of standard digital orthophoto quadrangles (DOQ) from the National Digital Cartographic Data Base. (Datum/Projection: NAD_1983_UTM_Zone_16N). The 2005 map was digitized using color digital imagery collected by United States Department of Agriculture—Farm Service Agency (USDA-FAS) Aerial Photography Field Office. (Datum/Projection: NAD_1983_UTM_Zone_16N)

(Group 4) and the Pine Forest (Group 3) have hydroperiods that are similar to other unditched peatlands located in Seney NWR (Fig. 6). In particular the Flooded Forests and the NonForested Wetland groups have water table heights that ranged <30 cm and maintained water tables within 20 cm of the surface. Relative to the other groups the Flooded Forest group had the highest snag basal area, high quantities of CWD, and were being colonized by wetland plants, suggesting these forests were in a process of transition to wetland. This further suggests that, in this portion of Seney NWR, wetland plants should establish in forests that are flooded.

The NonForested Wetland group (Group 4) also had abundant wetland species. However, these areas were likely wetlands prior to the installation of the ditch plugs, as suggested by the landscape analysis. We do not, however, know whether or not the species composition of these wetlands changed after the installation of the ditch plugs.

Normal precipitation during our study suggested that the Flooded Forests, NonForested Wetlands and the Pine Forest groups should maintain water table heights at or near the surface in years with average or above average precipitation. Furthermore, the water table heights were similar to unditched peatlands within the larger Seney NWR peatland complex (Fig. 6). Hence, the conversion of uplands to wetlands in these areas is likely permanent.

It is possible that the Pine Forest group (dominated by red pine, a species of more xeric conditions) will die because of the high water tables and that species preferring wetland conditions will colonize the sites once more light penetrates to the ground. In contrast the upland communities of the Spruce/Fir (Group 2) and Hemlock Forests (Group 5) appear unlikely to convert to wetland communities as their water tables are lower and more variable relative to unditched peatlands (Fig. 6).

The preEuropean landscape of Seney NWR consisted of a mosaic of upland and wetlands communities (Heinselman, 1965; Comer *et al.*, 1995; Silbernagel *et al.*, 1997). Therefore, it is unlikely, nor desirable, that hydrologic restoration will cause the entire study area to revert to a single wetland community type. The vegetation group that is the most likely to develop back into a peatland is the Flooded Forest group (Group 1). The success of restoration at these sites is comparable with other studies, in which blocking ditches led to flooding (Cooper *et al.*, 1998; Richert *et al.*, 2000; Mälson and Rydin, 2008), with a slow shift in vegetation to wetland species (Richert *et al.*, 2000; Mälson and Rydin, 2008).

The ditch plugs altered the hydrologic regime and resulted in a community transition to increased wetlands in low-lying areas near the ditch. If restoration to a historic wetland cover is to be achieved over a greater portion of the study area, additional effort will likely be needed to restore the water to the areas not currently in the path of direct flooding. It is uncertain if these wetlands will convert back into peatlands or to another type of wetland community, and this study should thereby provide a baseline for future studies. The formation of alternative stable states has been found in many wetland restoration projects and could be possible at Seney NWR (Moreneo-Mateos *et al.*, 2012). The increase in wetland area near the ditch was mostly on areas immediately adjacent to ditch plugs. The localized nature of the flooding may indicate that the ditch plugs did not result in sheet flow across the entire area, but instead, followed localized flow paths.

LAND COVER CHANGE

In addition to an increase in wetland vegetation near the ditch, land cover change analysis also showed changes occurring away from the ditch plugs near C-3 Pool, where formerly open water had shifted toward a scrub/shrub wetland. The loss of open water may be due to a decline in water inputs from the ditch because of the ditch plugs. Water now must travel overland, or via meandering stream channels, to reach C-3 Pool.

A change of 11% of the land area to wetland plant species is promising. However, most of this change occurred where upland forest converted to wetlands near the ditch plugs and in the regions near C-3 Pool where open water converted to wetland. Both of these changes likely occurred because of the ditch plugs. The regions near the Walsh Ditch were flooded because water moved across the landscape instead of down the ditch. Furthermore, the ditch was supplying less water to C-3, Pool, thereby lowering its height. The loss of standing water allowed shrubs and other low lying vegetation to colonize these sites. Changes from upland forest to forested or nonforested wetland was largely absent in the areas between the two aforementioned areas. This suggests that the water table has not risen sufficiently to cause mortality between the areas immediately adjacent to the ditch and the C-3 Pool, or more time is needed for the higher water tables to affect the trees in this region.

The establishment of wetland species in areas that converted from upland to wetland (Fig. 9) is promising. As hoped the regions that did experience flooding were colonized by wetland species, and there was no presence of exotic species. It was not expected that these areas would immediately reestablish a functioning peatland. In the future, a peatland may form in this area. The restored wetland is situated within a larger wetland complex, which may facilitate the transport of sedge seeds into the area. *Sphagnum* spp. will also recolonize a restored ditch area if hydrologic conditions are right (Bess and Chimner, unpubl. data), but reestablishment may also require introduction (*e.g.,* Ferland and Rochefort, 1997; Budelsky and Galatowitsch, 2000; Richert *et al.*, 2000). Active introduction of moss diaspores, mulching, and phosphorous fertilization have been demonstrated to improve the establishment success of *Sphagnum* spp. (Rochefort, 2000; Rochefort *et al.*, 2003). Future research is required to determine the successional path these newly reestablished wetlands will take.

CONCLUSIONS

In parts of the study area, raised water tables and colonization by wetland species in previously forested areas are indicators that natural hydrologic processes have begun to be restored over portions of the study area. Because of the long term nature of wetland restoration, however, it may be some time before some sites show a response to the altered hydrologic conditions (*e.g.,* Moreno-Mateos *et al.*, 2012). Portions of the study area that were wooded have begun a change to wetlands, but other portions of the landscape will remain as uplands. Other regions, such as the Pine Forest (Group 3), may become a lowland system if the trees die because of a high water table. In time active management may be required to establish *Sphagnum* species in the newly formed wetland area. However, we believe areas furthest removed from the ditch plugs are unlikely to show much change without additional efforts to distribute the water more evenly throughout the area. The results from this study indicated restoration management activities can be successful at over large areas. However, future restorations would benefit from pretreatment monitoring of the local hydrology and the establishment of clear restoration objectives that can be monitored.

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LITERATURE CITED

ANDERSON, S. H. 1982. Effect of the 1976 Seney National Wildlife Refuge wildfire on wildlife and wildlife habitat. U.S. Fish and Wildlife Service, Resource Publication 146.

BRAUN-BLANQUET, J. 1932. Plant sociology (Transl. G. D. Fuller and H. S. Conrad). McGraw-Hill, New York. 539 p.

BUDELSKY, R. A. AND S. M. GALATOWITSCH. 2000. Effects of water regime and competition on the establishment of a native sedge in restored wetlands. *J. App. Ecol.*, **37**:971–985.

CAMPBELL, D. AND J. BERGERON. 2012. Natural revegetation of winter roads in peat lands in the Hudson Bay Lowland, Canada. *Arct Antarct Alp Res*, **44**:155–163.

CHIMNER, R. A., J. M. LEMLY, AND D. J. COOPER. 2010. Mountain fen distribution, types and restoration priorities, San Juan Mountains, Colorado, U.S.A. *Wetlands*, **30**:769–771.

COMER, P. J., D. A. ALBERT, H. A. WELLS, B. L. HART, J. B. RAAB, D. L. PRICE, D. M. KASHIAN, R. A. CORNER, AND D. W. SCHUEN. 1995. Michigan's native landscape. Michigan Natural Features Inventory, Lansing, Michigan.

COOPER, D. J., L. H. MACDONALD, S. K. WENGER, AND S. W. WOODS. 1998. Hydrologic restoration of a fen in Rocky Mountain National Park, Colorado, U.S.A. *Wetlands*, **18**:335–345.

DAHL, T. E. 2006. Status and trends of wetlands in the conterminous United States 1998 to 2004. U.S. Department of the Interior; Fish and Wildlife Service, Washington, D.C. 112 p.

——— AND G. J. ALLORD. 1997. Technical aspects of wetlands; history of wetlands in the conterminous United States. National water summary on wetland resources, United States Geological Survey Water Supply Paper 2425.

——— AND C. E. JOHNSON. 1991. Status and trends of wetlands in the conterminous United States, Mid-1970s to Mid-1980s. U.S. Department of the Interior; Fish and Wildlife Service: Washington, D.C, 28 p.

DROBYSHEV, I., P. C. GOEBEL, D. M. HIX, R. G. CORACE III, AND M. E. SEMKO-DUNCAN. 2008. Pre and postEuropean settlement fire history of red pine dominated forest ecosystems of Seney National Wildlife Refuge, Upper Michigan. *Can. J. For. Res.*, **38**:2497–2514.

———, ———, D. M. HIX, R. G. CORACE III, AND M. SEMKO-DUNCAN. 2008b. Interactions between forest composition, structure, fuel loadings and fire history: a case study of red pine dominated forests of Seney National Wildlife Refuge. *For. Ecol. Manage.*, **256**:1723–1733.

DURANEL, A. J., M. C. ACREMAN, C. J. STRATFORD, J. R. THOMPSON, AND D. J. MOULD. 2007. Assessing the hydrological suitability of floodplains for species-rich meadow restoration: a case study of the Thames floodplain, U.K. *Hydrol. Earth Syst. Sc.*, **11**:170–179.

FERLAND, C. AND L. ROCHEFORT. 1997. Restoration techniques for Sphagnum-dominated peatlands. *Can. J. Bot.*, **75**:1110–1118.

GRAF, M. D., L. ROCHEFORT, AND M. POULIN. 2008. Spontaneous revegetation of cutaway peatlands of North America. *Wetlands*, **28**:28–39.

HEDBERG, P., W. KOTOWSKI, P. SAETRE, K. MÄLSON, H. RYDIN, AND S. SUNDBERG. 2012. Vegetation recovery after multiple-site experimental fen restorations. *Biol Conserv*, **147**:60–67.

HEINSELMAN, M. L. 1965. String bogs and other pattered organic terrain near Seney, Upper Michigan. *Ecology*, **16**:185–188.

KOWALSKI, K. P. AND D. A. WILCOX. 2003. Differences in sedge fen vegetation upstream and downstream from a managed impoundment. *Am. Midl. Nat.*, **150**:199–220.

LAVOIE, C., K. MARCOUX, A. SAINT-LOUIS, AND J. S. PRICE. 2005. The dynamics of a cotton-grass (*Eriophorum vaginatum* L.) cover expansion in a vacuum-mined peatland, southern Quebec, Canada. *Wetlands*, **25**:64–75.

MÄLSON K. AND H. RYDIN, The regeneration capabilities of bryophytes for rich fen restoration. *Biol. Conserv.*, **135**:435–442.

MCCUNE, B. AND M. J. MEFFORD. 2006. PC-Ord. Multivariate analysis of ecological data. Version 5.32. MjM Software, Gleneden Beach, Oregon, U.S.A.

MERETSKY, V. J., R. L. FISCHMAN, J. R. KARR, D. A. ASHE, J. M. SCOTT, R. F. NOSS, AND R. L. SCHROEDER. 2006. New directions in conservation for the National Wildlife Refuge System. *BioScience*, **56**:135–143.

MITSCH, W. J. AND J. G. GOSSELINK. 2007. Wetlands. 4th edition. John Wiley & Sons Inc., Hoboken, New Jersey.

MORENO-MATEOS, D., M. E. POWERS, F. A. COMIN, AND R. YOCKTENG. 2012. Structural and functional loss in restored wetland ecosystems. *PloS Biol.*, **10**:1–8.

RICHERT, M., O. DIETRICH, D. KOPPISCH, AND S. ROTH. 2000. The influence of rewetting on vegetation development and decomposition in a degraded fen. *Restor. Ecol.*, **8**:186–195.

RICHTER, B. D., J. V. BAUMGARTNER, J. POWELL, AND D. P. BRAUN. 1996. A method for assessing hydrologic alteration within ecosystems. *Conserv. Biol.*, **10**:1163–1174.

ROCHEFORT, L. 2000. Sphagnum: a keystone genus in habitat restoration. *Bryologist*, **103**:503–508.

———, QUINTY, S. CAMPEAU, K. JOHNSON, AND T. MALTERER. 2003. North American approach to the restoration of Sphagnum dominated peatlands. *Wetl. Ecol. Manag.*, **11**:3–20.

SCHROEDER, R. L., J. I. HOLLER, AND J. P. TAYLOR. 2004. Managing National Wildlife Refuges for historic and nonhistoric conditions: determining the role of the refuge in the ecosystem. *Nat. Resour. J.*, **44**:1185–1210.

SILBERNAGEL, J., J. CHEN, M. R. GALE, K. S. PREGITZER, AND J. PROBST. 1997. An interpretation of landscape structure from historic and present land cover data in the eastern Upper Peninsula of Michigan. United States Forest Service, GTR NC-192, St. Paul, Minnesota.

SWEAT, M. J. 2001. Hydrology of C-3 watershed, Seney National Wildlife Refuge, Michigan. U.S. Geological Survey, Lansing, Michigan.

THOGMARTIN, W. E., M. GALLAGHER, N. YOUNG, J. J. ROHWEDER, AND M. G. KNUTSON. 2007. Factors associated with succession of abandoned agricultural lands along the lower Missouri River, U.S.A. *Restor. Ecol.*, **17**:290–296.

U.S. FISH AND WILDLIFE SERVICE (USFWS). 2001. Refuge Management; National Wildlife Refuge System (Part 601); Biological Integrity Diversity, & Environmental Health. Regional Office, Fort Snelling, Minnesota.

———. 2009. Seney National Wildlife Refuge Comprehensive Conservation Plan. Regional Office, Fort Snelling, Minnesota.

WADDELL, K. L. 2002. Sampling coarse woody debris for multiple attributes in extensive resource inventories. *Ecol. Indic.*, **1**:139–153.

WADDINGTON, J. M., M. J. GREENWOOD, R. M. PETRONE, AND J. S. PRICE. 2003. Mulch decomposition impedes recovery of a new carbon sink function in a restored peatland. *Ecol. Eng.*, **20**:199–210.

WILCOX, D. A., M. J. SWEAT, M. L. CARLSON, AND K. P. KOWALSKI. 2006. A water-budget approach to restoring a sedge fen affected by diking and ditching. *J. Hydrol.*, **320**:501–517.

ZEDLER, J. B. 2000. Progress in wetland restoration ecology. *Trends Ecol. Evol.*, **15**:402–407.

ZHANG, L. AND W. J. MITSCH. 2007. Sediment chemistry and nutrient influx in a hydrologically restored bottomland hardwood forest in Midwestern U.S.A. *River Res. Appl.*, **23**:1026–1037.

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