FINAL REPORT
South Carolina SWG Project T-36-HM
South Carolina Department of Natural Resources
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A GIS-based Model to Guide Landscape-scale Restoration at the Woodberry Tract and Hamilton Ridge properties.

Job 1. Create GIS-based model of pre-fire excluded landscape patch dynamics and hydrologic change for the Woodberry Tract and Hamilton Ridge properties.

JUSTIFICATION AND NEED

A landscape mosaic of open- and closed- canopied habitats historically characterized the Southeastern Coastal Plain. Longleaf pine (*Pinus palustris*) savannas and flatwoods, with a diverse groundcover and rich faunal component, were typical of open canopy habitats. Mixed pine-hardwood forests were characteristic of closed canopy habitats. Mixed pine woodlands with moderately open canopies occurred within and transitionally among other landscape components. This landscape dynamic was maintained by high frequency, low intensity fire caused by lightning and humans. By the 1950's, fire exclusion became a dominant silvicultural practice in the Southeast Coastal Plain. Fire exclusion and land use changes resulted in a 97 % decline in longleaf habitats, and consequently, many associated species are now imperiled.

The South Carolina DNR recently acquired two large tracts of land. The Woodberry Tract, Marion County, is 25,668 acres, and the Hamilton Ridge property, Hampton County, is13,281 acres. Land use histories vary for both properties but currently include industrial forestry. The DNR is in the process of developing Conservation Plans for these two tracts that may include ecological restoration. A controversial issue of ecological restoration is the selection of reference conditions and the way in which restoration success is measured. Attributes such as community structure or species composition of unaltered, pristine sites are often used to identify reference conditions and gauge management success. However, the reliance on such reference sites can bias restoration efforts because few examples of pristine habitats exist, and thus are unlikely to encompass the range of variation that historically existed across landscapes. To avoid this problem, some restoration ecologists have proposed a landscape approach to ecological restoration. The landscape approach is based on the reestablishment and continuation of key-stone processes. The broad scale of the landscape approach can initially place more emphasis on coarse structural components of landscape patches (e.g., open and closed canopy habitats) and less emphasis on species composition. This approach to restoration assumes that once the key-stone processes are reestablished, successional pathways will resume, selection filters will determine species composition, ecological engineers will reestablish, and the site being restored will return to and continue on a 'natural' trajectory. This approach to restoration is ideally suited to the Woodberry Tract and Hamilton Ridge properties because of their size. Further, the method will restore important ecological processes to the landscape (e.g., fire) and provide habitat structures needed to maximize conservation benefits while using the species composition of the current industrial forest.

Objective 1. Identify reference conditions and model spatial correlations between the pre-fire excluded landscape patches and geomorphic features.

Anthropogenic landscape change resulting from logging, fire exclusion, and soil drainage practices (e.g., ditches), altered and destroyed the majority of remaining old growth pine woodlands and savannas by the mid 20th century. Efforts to restore pine woodlands and savannas have been impeded by a lack of remnant sites to serve as a reference for guiding restoration. However, the rate, timing, and scale of anthropogenic landscape change are dependent on dynamic cultural drivers (e.g., access to market) that respond to economic incentives and technologic advances. Cultural drivers affect landscapes variably, both spatially and temporally, which contributes to overall heterogeneity. Although dynamic, these cultural drivers add predictability to landscape change, and can be applied to historical data to identify likely locations of remnant landscapes at different periods in time.

We used historical data and GIS software to examine landscape change in the South Carolina Coastal Plain. We used a landscape-ecology approach to conduct searches for historical documents relevant to fire-maintained habitats at large spatial scales. We examined historical reports, travel accounts, various historical maps, and historical aerial photographs to identify remnant pine woodlands and savannas from the early and mid 20th century. We used interviews and oral histories to corroborate documentation of remnant landscapes. Once identified, historical aerial photographs and hard-copy maps were used to create GIS data to document the distribution and spatial characteristics of pine woodlands and savannas.

Historical aerials were used with GIS software to model landscape dynamics of remnant woodland - savanna landscapes and derive landscape scale reference conditions. Our specific goals were to:

1) Compare shape complexity of landscape patches within remnant fire-maintained landscapes to random landscapes
2) Characterize remnant landscapes based on open-canopy, fire-maintained habitat patches and examine spatial correlations between the fire-maintained landscape patches and geomorphology
3) Examine the distribution of remnant landscapes in relation to drivers of anthropogenic landscape change within a historical context that considers the time scales of pine woodlands –savanna dynamics

Approach

We conducted searches for aerial photographs and historical documents relevant to longleaf ecosystems in South Carolina at the National Archives and Records Administration, USGS Earth Resources, University of South Carolina Map Library, South Caroliniana Library, and the South Carolina Archives. Multiple sources, including 1938 USDA aerial photographs and verification by two living witnesses were used to delineate the spatial extents of remnant woodland – savanna landscapes. Remnant landscapes were defined as fire-maintained tracts > 1,000 ha that contained mature longleaf pine (*Pinus palustris*). Additionally, remnant tracts had experienced only limited cutting (i.e., no large-scale clear cuts) and turpentine extraction. Because longleaf pine is a long lived, fire dependant species with poor dispersal abilities, the presence of mature longleaf pine was used as an indication that the landscapes had been under a high frequent fire regime for several hundred years. Thus, the term remnant was used to identify large tracts of fire-maintained landscapes with mosaics of open- and closed-canopy patches created and maintained by the interactions among vegetation, fire, and geomorphology.

Hardcopies of aerial photographs were scanned using a Canon Canoscan 8800F© desktop scanner and were digitally stored as TIFF files. Aerials were scanned at 600 – 1200 dpi resolutions, depending on the scale and the resolution of the data to be derived from the aerial photograph. Digital copies of aerial photographs were geo-rectified using ArcGIS and ERDAS Imagine GIS-software platforms with a target RMSE < 9. Remnant landscapes were identified and delineated based on spatial congruence between eyewitness accounts and the visual evidence of historical aerial photos (Fig. 1). Corroborating evidence included the spatial imprints of temporary railroad track spurs, which were historically used during large clear-cuts, and the likely presence of mature pine trees within uncut areas adjacent to clearcuts. The likely presence of mature trees was based on shadow-based estimates of tree heights > 30 m.

Figure 1. Portion of remnant landscape in Jasper County 1938 with evidence of temporary railroad track spurs used for clearcutting mature pine woodlands. Eyewitness accounts of uncut pine woodlands and savannas were corroborated with historic aerial photographs.

1) Comparing shape complexity of remnant fire-maintained landscapes to random landscapes

Because landscape patch complexity is predicted to be higher in un-humanized landscapes, as compared to humanized landscapes, landscape-patch complexity may serve as a spatial signal for remnant landscapes. This is important because spatial signatures could be used to assist with selecting properties with high conservation potential. Thus, we compared remnant landscapes to random landscapes to determine if shape complexity could be used to distinguish remnant pine woodlands and savannas in the South Carolina Coastal Plain.

We conducted supervised classifications to identify landscape-scale habitat patches based on percent canopy cover (Table 1). Tree canopy spectral signatures were identified and reclassified to generate binary grids identifying the presence and absence of canopy cover (i.e., grid values 1 and 0 respectively). The binary canopy cover grids were visually compared to the digital imagery and edited, when necessary, using ArcMap 9.3 Spatial Analysis. A 15 x 15 m roving neighborhood operator was used to sum grid values, which were then divided by the total neighborhood area (i.e., 15 x 15 = 225) and multiplied by 100 to generate continuous percent canopy cover grids. Percent canopy cover grid values were reclassified to characterize areas as one of the three landscape habitat patch types (Table 1).

Table 1. Landscape habitat patch types classified by tree canopy spectral signatures of historical aerial photographs taken in Jasper County, 1938.

Habitat patch	Percent canopy	Estimated trees/ha (trees/acre)
Forest	> 80	1 – 10 (1 – 4)
Woodland	31 - 70	10 – 90 (4 – 40)
Savanna	0 – 30	> 90 (> 40)

Two random landscape samples from each remnant tract were compared to random samples (n=16) selected from the entire study area (i.e., 1938 Jasper County, South Carolina) to test for differences in patch shape complexity in humanized and non-humanized fire-maintained landscapes (Fig. 2). A range of circular buffers (200, 400, 600, 800, and 1000 m radii) were used at each sample location to examine scale-dependant responses. Each scale was analyzed separately and sample plot distance criteria were used during random location selection to avoid plot overlap and to maintain sample independence. Two metrics of shape complexity were used independently as landscape response variables. Specifically, area weighted mean patch fractal dimension (MPFD) and area weighted mean shape index (MSI) values were calculated using ArcGIS and FRAGSTATS. Both metrics were used to compare the shape complexity of habitat patch types. Analysis of variance (ANOVA), with the two samples from each remnant tract treated as subsamples, were performed for each sample scale and response variable using SAS 9.1.

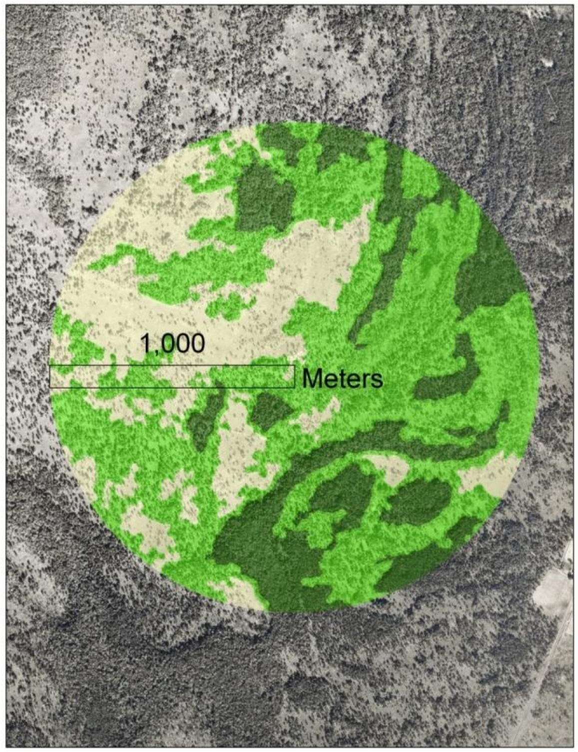

Figure 2. Landscape habitat patches were derived from historical aerial photographs based on tree canopy spectral signatures. Landscape shape complexity was compared between remnant landscapes and random locations within Jasper County 1938.

Results

Analysis of variance indicated that remnant landscapes had greater shape complexities than random landscapes (Figs. 3 & 4). Greater complexity was indicated by both landscape metric and greater shape complexity was detectable at multiple scales. When these results were applied to historic aerial photographs, they indicated that shape complexity can be used as a spatial signal to identify likely locations where remnant fire-maintained landscapes most recently occurred in the South Carolina Coastal Plain.

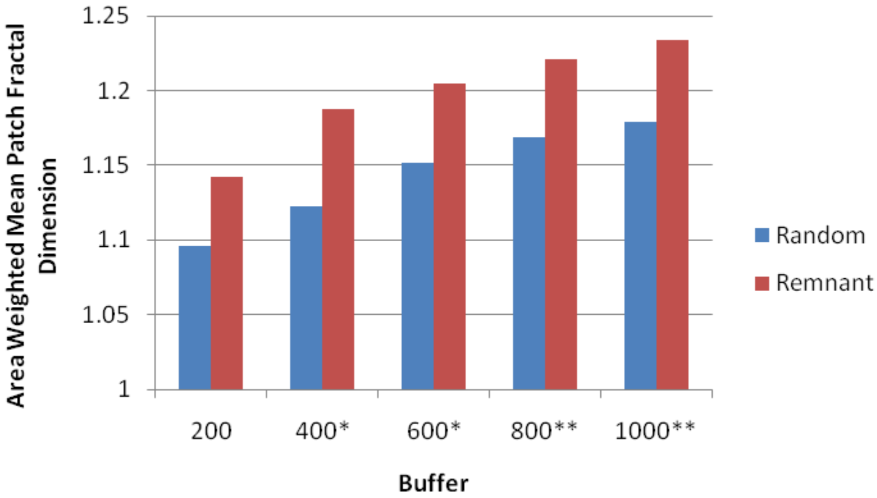

Figure 3. Area-weighted mean patch fractal dimension of remnant landscapes versus randomly located landscapes in Jasper County, SC 1938. Analysis was conducted at multiple scales derived from buffers (m). Higher response values indicate greater shape complexity. A single asterisk indicated significance at $P < 0.05$. A double asterisk indicated significance at $P < 0.01$.

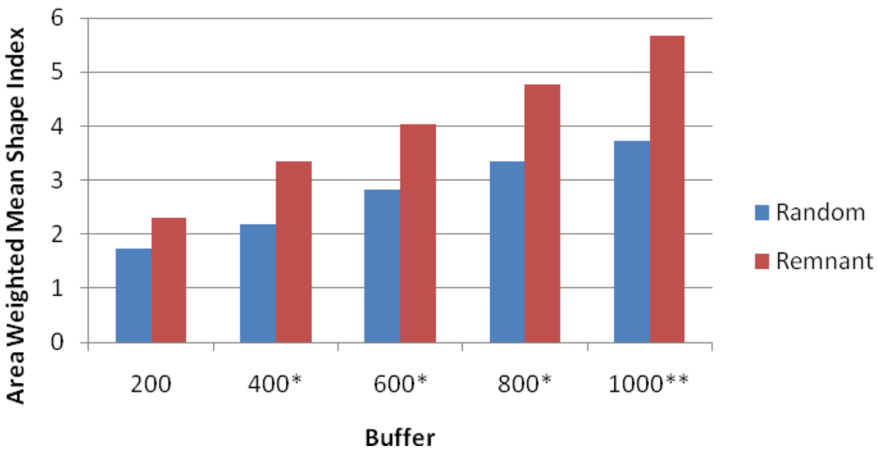

Figure 4. Area-weighted mean shape index of remnant landscapes versus randomly located landscapes in Jasper County, SC 1938. Analysis was conducted at multiple scales derived from buffers (m). Higher response values indicate greater shape complexity. A single asterisk indicated significance at $P < 0.05$. A double asterisk indicated significance at $P < 0.01$.

2) Characterize remnant landscapes based on open-canopy, fire-maintained habitat patches and examine spatial correlations between the fire-maintained landscape patches and geomorphology

Geo-referenced aerial photographs of remnant pine woodland savannas contain spatially-explicit information not available from extant reference sites. Specifically, historical remnants provide insight into habitat patch size, shape, distribution, and overall structure of non-extant landscapes. Thus, we used georeferenced historical aerials of remnant pine woodland savannas in the South Carolina Coastal Plain to derive landscape- and patch-scale GIS data, which can serve as reference conditions for restoration on WT and HR properties.

For these analyses, woodland and forest habitat patches were combined to reduce spatial resolution disparities among GIS-based geomorphology data, elevation date, and our landscape habitat patches. Specifically, we chose to pool woodland and forest patches while maintaining savanna patches, as defined in this study, for three reasons. First, we observed savannas with large spatial extents within remnant landscapes. Secondly, the current rarity of large southeastern savannas with canopy closures < 30% gives more conservation weight to reference conditions derived from the remnant savannas that we observed. Third, because savannas require high-frequency fires and were historically referenced to geomorphic features (e.g., seasonally inundated soils), we expected that the spatial-extent of savannas in historical aerials would provide the strongest signal for modeling the spatial relationship between open-canopy, fire-maintained landscape patches and geomorphic features. Thus, by combining woodlands and forest patches, we increased our probability of successfully modeling the spatial relationship between open, fire-maintained landscape patches and geomorphic features. Because savannas were a dominant landscape feature and many of the savannas we observed exceeded the spatial extents of our remnant landscapes, the landscape metrics (e.g., area) calculated in this study likely underestimated the true spatial dimensions of historic savannas (e.g., Fig. 5).

Figure 5. Fire-maintained savannas occurred at large spatial extents within remnant landscapes, which is portrayed in this 1938 image of a Jasper County landscape. Fire-maintained savannas occurred as both landscape patches and the landscape matrix.

To model the spatial relationships between fire-maintained landscape dynamics relative to geomorphic features, we used randomly generated points as independent samples to characterize landscape features. Random points were stratified by open- and closed-canopied habitats and were spatially joined to geomorphology data. Open versus closed canopy landscape patches were used as binary response variables and geomorphology data were used as predictors in logistic regression analysis.

Results

Fire-maintained open canopy savannas were significant components of remnant landscapes and occurred at both the patch-scale and as a landscape matrix (Table 2). Modeling attempts failed to identify spatial correlations between fire-maintained landscape dynamics and geomorphic features. Spatial disparities in data resolution are the most likely explanation for our inability to model these relationships, and thus limit our ability to predict landscape patch dynamics.

Table 2. Savanna patch and landscape component structures within remnant landscapes, Jasper County, 1938. Average savanna patch size (ha) represents savannas that occurred completely within a remnant landscape sample. Total savanna matrix refers to the area (ha) encompassed by savannas with extents that exceeded the boundaries of remnant landscape samples (i.e., these savannas were treated as a landscape matrix rather than a patch). Percent depicts the portion of the remnant landscape characterized as savanna, based on less < 30% canopy closure.

Remnant Landscape	Average Savanna Patch	Total Savanna Matrix	Percent
1	2.94	1585.44	0.26
2	12.89	213.98	0.24
3	2.82	721.83	0.40
4	9.18	374.41	0.33

3) Examine the distribution of remnant landscapes in relation to drivers of anthropogenic landscape change within a historical context that considers the time scales of pine woodlands – savanna dynamics

Within the historic range of southeastern pine woodlands, it is assumed that market access to timber resources was responsible for the loss of remnant landscapes. Specifically, technological advancements, such as the use of steam locomotives for large-scale clear-cuts, increased market access to these natural resources and facilitated there exploitation. We examined the distribution of remnant landscapes at a broad based on an access-to-market model within the historical context of our data. We assumed market-access was a significant driver of natural resource extraction and landscape change. We used timber as the natural resource of economic interest and assumed that transporting mature timber was historically a significant cost that limited access to market. Specifically, we assumed that transporting mature timber from its felling to a major transportation route (e.g., navigable river) had an economic cost that historically drove timber extraction at our study scale. Therefore, we used the distance to three different transportation methods (navigable water-ways, maintained roads, and main-line railroads) as a measure of market access. We selected these methods of timber transportation because when combined, they spanned a range of historical harvest technologies that encompassed pre-industrial (i.e, navigable water-ways) to mid-20th century (i.e., maintained roads) technologies. Anthropogenic landscape change, market access, and technologies were linked when newer technologies reduced distance-based costs and incentivized resource extraction. Because the temporal extents of remnant landscapes spanned several centuries, we assumed that remnant distributions would reflect the cumulative effects of older technologies that defined market access for longer periods. Thus, we hypothesized that the distribution of remnant landscapes in 1938 was largely determined by the distribution of navigable water-ways.

Historical documents (e.g., nautical charts) and geo-rectified 1938 aerial photos were used to identify and digitized transportation routes. We used ArcGIS to generate two sets of random point locations. One set of random points was limited to, and stratified by, remnant landscapes. The second set of random points was applied across the entire study area (i.e., Jasper County 1938). We calculated minimum Euclidian distances from random points to each of the three transportation methods. Remnant point locations were stratified by landscape (N = four landscapes) and distance values (*D*) were averaged for each transportation method (*TM*). Distance values were used to generate four distance ratios using the following equation:

$$TM = \frac{D_{remnant}}{D_{random}}$$

Averaged point locations for the entire study area were used as the denominator in the *TM* equation. Individual quotients for each of the four remnants were combined within each transportation method and used to perform t-tests with the null hypotheses that $\mu = 1$. Ratios that did not differ from 1 indicated no spatial relationship between transportation method and remnant landscapes, while ratios significantly greater than 1 and significantly less than 1 indicated that remnants occurred further from or closer to transportation methods, respectively.

Results

Distance ratios indicated that 1938 remnant landscapes occurred further from navigable waterways than expected (Tables 3 and 4).

Table 3. Raw distance scores calculated from randomly generated point locations within the entire study area (Jasper County 1938). Site includes random, which refers to the randomly generated points covering the entire study area, and remnant sites, which refer to random locations limited to the spatial extents of remnant landscapes.

Site	Navigable Waterway	Maintained Road	Mainline Railroad
Random	3512.91	1287.48	3313.52
Remnant 1	4457.42	1605.74	2427.26
Remnant 2	7842.38	914.14	1879.52
Remnant 3	5602.24	1550.26	1556.54
Remnant 4	6006.26	1692.48	2179.60

Table 4. Distance ratios and average distance ratio for remnant landscapes relative to three transportation modes used for large-scale timber harvests. Significance was based on T-tests with a null hypothesis that average ratio per transportation mode = 1.

Site	Navigable Waterway	Maintained Road	Mainline Railroad
Remnant 1	1.27	1.25	0.73
Remnant 2	2.23	0.71	0.57
Remnant 3	1.59	1.20	0.47
Remnant 4	1.71	1.31	0.65
Average Ratio	1.70*	1.12	0.61**

* $P < 0.05$
** $P < 0.01$

<u>Significant Deviations</u>: None

Objective 2. To create a GIS coverage identifying former isolated wetlands and a GIS coverage identifying changes in hydrology related to road construction and ditching at the Woodberry Tract and Hamilton Ridge properties.

Historical aerial photographs from 1938, through 1979, and DOQQs from 1994 and 2006 were used to provide an approximate 68-year temporal extent at 10-15 year intervals within the study areas. Hardcopies of aerial photos were scanned using a Canon Canoscan 8800F© desktop scanner and were digitally stored as TIFF files. Aerials with 1:20,000 scale were scanned at 900 dpi and 1:40,000 scale aerials were scanned at 1200 dpi, which resulted in 0.45 and 0.8 m ground pixel resolutions, respectively, post georectification. Although camera calibration reports were not available, coarse camera focal lengths and photo scales were used in conjunction with digital photogrammetry, 2006 DOQQs, and USGS National Elevation Dataset (NED) 1/3 second raster elevation data to orthorectify the historical aerial photos. Horizontal and vertical ground control point coordinates were collected from the 2006 DOQQs and 1/3 second NED data. Block aerial triangulation adjustments (bundle block adjustment) based on a least squares approach and a non-metric camera models were used to estimate unknown photogrammetric variables (e.g., camera interior and exterior orientations in relation to the Earth's surface). These estimated photogrammetric variables were then used to orthorectify the aerial photography and minimize spatial error.

Orhtorectifided aerial photos were used with soils and elevation data to identify and delineate isolated wetlands within the study areas (Fig. 6). We used the historical aerial photographs to quantify the effects of ditching, logging, and road development (Table 5) on isolated wetlands. Further, we visually estimated isolated wetland canopy coverage. We quantified these variables for each temporal sequence of aerial photographs (hereby referred to as individual years) within 50-m buffers of isolated wetlands on HR, Webb, and the Woodbury tract property. A subsample of ponds was selected for statistical modeling (i.e., HR, N = 57; Webb, N = 27). Thus, we recorded six scores for ditches, logging activity, roads, and canopy cover (i.e., one score for each year) for each isolated wetland. We scored logging activity, ditches, and roads as categorical variables (Table 5). Logging activity included four categories, ranging from 0 through 3, with 0 representing undetectable logging activity and 3 representing large-scale logging activity that included clear-cuts, site-preparation, and planting. Ditches were represented by 3 categories (i.e., 0, 1, and 2) in which 0 indicated that ditches were not visible, 1 indicated that one ditch was present, and 2 indicated that either two ditches were present or that older ditches were improved (i.e., widened). Similarly, we categorized roads based on their extent within the pond buffer such that 0 indicated that no roads were present within the buffer, 1 indicated that one road was present within the buffer, and 2 indicated that > 1 road crossed through the pond buffer or that at least one road was positioned along the edge of the wetland (within the inner boundary of the buffer) or bisected the wetland, likely affecting wetland hydrology. We used these variables (i.e., canopy cover and scores of logging, ditches, and roads) to develop three indices of isolated wetland degradation related to hydrologic changes:

1) Mechanical Degradation Index (MDI): measures the extent to which isolated wetlands suffered mechanistic degradation (i.e., logging, ditches, and roads) for each year represented in the study. We used yearly MDI estimates to calculate a cumulative estimate (MDI_c) for each isolated wetland.

2) Integrity Index (INI): provides a measure ecological integrity for each isolated wetland (per year) that can be used to assess restoration value/potential. Yearly INI estimates were used to calculate a cumulative INI estimate (INI_c) for each pond.
3) Legacy Index (LGI): measures restoration potential of each wetland, based on time since degradation and current habitat structure, relative to other isolated wetlands in the study area.

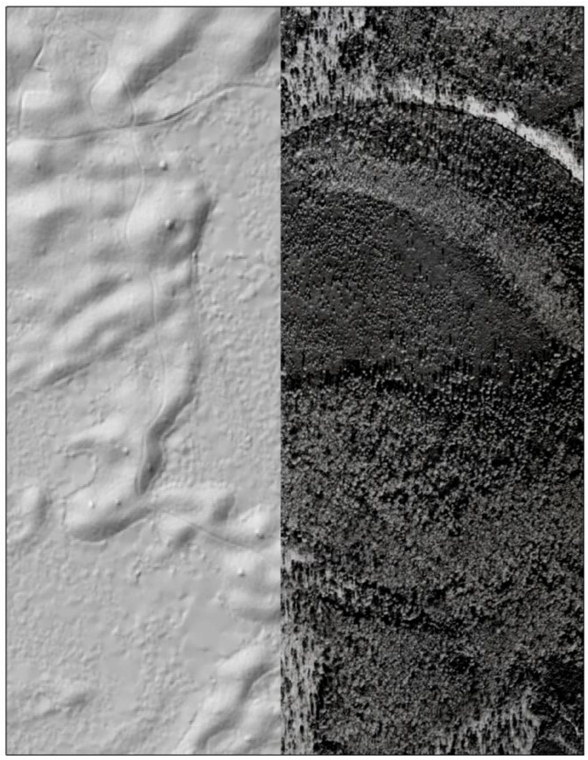

Figure 6. Orthorectified historical aerials (right side of image) were used in conjunction with elevation data (left side of image) to delineate isolated wetlands within a 75-year time frame at the Woodbury tract and Hamilton Ridge properties.

Table 5. Variables included in isolated wetland degradation indices for isolated wetlands on Hamilton Ridge WMA, Webb WMA, and the Woodbury Tract. All variables were visually estimated from aerial photographs taken between 1938 and 2006.

Variable	Type	Description
Roads	Categorical	0 = none 1 = one road present 2 = more than one road, or large road that likely affected wetland hydrology
Ditches	Categorical	0 = none 1 = one ditch present 2 = more than one ditch present or an improvement of an existing ditch (e.g., widening or lengthening)
Logging	Categorical	0 = no evidence of logging activity 1 = evidence of timber harvests (thinning or clear-cutting without supplemental site preparation) 2 = large-scale timber harvest with limited subsequent site preparation (e.g., wind rows, bedding), but no evidence of planting 3 = large-scale timber harvest with extensive site preparation and re-planting
Canopy Cover	Continuous	Canopy coverage to the nearest 10 %

Mechanical Degradation Index

We used three variables (i.e., roads, ditches, and logging; Table 5) to calculate MDI values, which measured the extent of mechanical degradation experienced by isolated wetlands for each year represented in the study. The MDI was intended to provide insight into the time-frame in which isolated wetlands were degraded and allowed for comparisons through time and across landscapes subjected to different land-use practices. We assumed that mechanical disturbances disproportionately degraded isolated wetlands, and thus, we weighted degradation scores to reflect this relationship. Specifically, logging was weighted six times greater than roads and twice that of ditching. Degradation from ditches were weighted three times that of roads. As such, these weights were incorporated into computing the raw mechanical degradation score (M) using the following equation,

$$M = \left(\left(\frac{R}{R_{max}}\right) + \left(3 \times \left(\frac{D}{D_{max}}\right)\right) + \left(6 \times \left(\frac{L}{L_{max}}\right)\right)\right)$$

where R = road score, D = ditch score, L = logging score. We used the quotient of M divided by its maximum possible value (M_{max} = 10) to derive a yearly MDI (MDI_y) for each pond:

$$MDI_y = \frac{M}{M_{max}}$$

We summed MDI_y values across years for each pond (m) and divided by the maximum observed value of m (i.e., m_{max} = 33) to estimate the cumulative MDI (MDI_c):

$$MDI_c = \frac{m}{m_{max}}$$

The MDI is bound between 0 and 1 such that 1 = extreme degradation and 0 = no degradation.

Integrity Index

Because fire is a keystone process in southeastern pine savannas and woodlands, and thus effects isolated wetlands, fire exclusion is a form of ecological degradation with negative impacts on imperiled flora and fauna. Highly imperiled ecosystems that depend on keystone processes (e.g., fire) for self-organizing feedbacks among ecological structures and functions require active management to maintain ecological integrity. In this study, we computed an integrity index as a tool for measuring the restoration potential of isolated wetlands within in landscape context that assumed high-frequency fires were necessary to maintain ecological integrity. We used canopy closure as an indicator of fire frequencies, and thus ecological integrity. Specifically, we assumed that closed canopies were indicative of fire-excluded habitats. Thus, we developed an integrity index (INI) that incorporated fire as a keystone process necessary to maintain wetland integrity.

The INI differs from the MDI in its incorporation of canopy closure as a measure of wetland degradation resulting from sources that are not directly detectable in aerial photographs (e.g., fire exclusion). Isolated wetlands occurring in functional southeastern pine savannas and woodlands experienced frequent fires, resulting in an open canopy structure. Within this landscape context, isolated wetlands with closed canopies were considered degraded. Thus, we incorporated canopy cover in the INI so that non-mechanical degradation (e.g., fire exclusion) that affected habitat structure was incorporated in the final integrity ranking. For example, closed canopy isolated wetlands that did not receive extensive mechanical degradation (i.e., they ranked low on the MDI), were penalized by the INI due to poor habitat structure.

We incorporated canopy cover into the calculation of the disproportionate effects of ditches (d), roads (r), and logging (l) using the following equations,

$$r = C\left(\frac{R}{R_{max}}\right)$$

$$d = C\left(\frac{D}{D_{max}}\right) \times 3$$

$$l = C_{max}\left(\frac{L}{L_{max}}\right) \times 6$$

where C refers to the inverse natural log of canopy cover and C_{max} refers to the maximum possible canopy value (i.e., C_{max} = 2.72). We multiplied the sum of r, d, and l by C to derive the raw integrity score (I), which was standardized by the maximum possible integrity score (i.e., I_{max} = 73.89) to derive yearly INI (INI_y) for each wetland:

$$I = C(r + d + l)$$

$$INI_y = \frac{I}{I_{max}}$$

We summed INI_y values across years for each pond (i) and divided by the maximum observed value of i (i.e., i_{max} = 1.7) to estimate the cumulative INI (INI_c):

$$INI_c = \frac{i}{i_{max}}$$

The INI is bound between 0 and 1 such that 1 = limited or no integrity and 0 = high integrity.

Legacy Index

Successful restoration ecology is often gauged by the presence and recolonization of imperiled species. Restoration efforts benefit from the endemic species with temporal persistence, i.e., those species that tend to persist for extended periods following habitat degradation, even when populations are not viable. Such species are considered biological legacies and are often important for successful restoration. Southeastern savanna endemics tend to have slow life histories, characterized by longevity and poor dispersal ability, and thus many persist as decades as biological legacies in degraded habitats. We calculated a legacy index (LGI) that incorporated biological legacies into our measure of wetland integrity.

Similar to INI, the LGI incorporates mechanistic degradation, canopy structure, as well as a penalty for time since degradation that is based on the temporal persistence of biological legacies, which are more likely to be present when disturbance or degradation is recent. We assumed that pine savanna remnants could function as biological legacies for approximately twenty years post- wetland degradation (based on maximum longevities and habitat specificity of pine savanna endemics), although any biological legacy time-scale can be used in the LGI. Based on this assumption, the LGI weighted degradation relative to biological legacy persistence such that degradation that occurred outside the time-frame in which legacies persisted (i.e., 20 years prior to present day, or 2010) were penalized. Specifically, the LGI penalized the wetland integrity score (I) for degradation (i.e., logging, ditches, roads, and closed canopy) that occurred 20 years prior to 2010; thus, scores recorded from aerials ranging between 1938 and 1979 were penalized to account for the loss of biological legacies. Because it is based on past degradation, time since degradation, and habitat structure, the LGI provides insight into restoration potential for isolated wetlands relative to other wetlands within the same study area. Thus, the final LGI can be used to rank wetlands according to their restoration potential.

Penalties for time since degradation were applied to the integrity score (I), in which r, d, and l values from 1938 – 1978 were scored 30 % higher than scores recorded in 2006:

$$r_p = r \times 1.3 \qquad r_{2006} = r \times 1.0$$
$$d_p = d \times 1.3 \qquad d_{2006} = d \times 1.0$$
$$l_p = l \times 1.3 \qquad l_{2006} = l \times 1.0$$

The equations used to calculate INI_y and INI_c were modified and to calculate LGI_y and LGI_c. For LGI_y, $I_{max} = 103.45$ was as used as the divisor of I. We summed LGI_y values across years for each pond (i) and divided by the maximum observed value of i (i.e., $i_{max} = 1.43$) to estimate the cumulative LNI (LNI_c). LNI is bound between 0 and 1 such that 1 = limited or no integrity and 0 = high integrity.

Statistical analyses
We used a subset of ponds from HR (N = 57) and Webb (N = 27) to assess performance of the indices with respect to different land use histories. We used repeated measures logistic regression (Proc Genmod) to examine the relationships between indices (i.e., MDI and INI), location (HR versus Webb), and year. The models included year, location, and their interaction as predictor variables, yearly index values as the response, and individual ponds as the repeated factor. Because land-use practices on HR centered on industrial forestry for > 40 ybp, we expected that isolated wetlands on HR would be more degraded and lave less integrity than those on Webb. Further, we expected that degradation would increase in association with the industrialization of forestry, i.e., technological advancements that allowed for large-scale harvests, site preparation, bedding, and planting. We compared all cumulative values for all three indices between study sites using T-tests.

Results

Isolated wetlands on Webb WMA tended to score better with respect to yearly MDI and INI, particularly after 1968 when HR wetlands were degraded by industrial forestry practices (Fig. 7). Between 1938 and 1948, yearly MDI and INI scores for the two sites were similar (Fig. 7), but increased mechanical degradation beginning in 1958 drastically increased yearly INI values for HR wetlands (Fig. 8). Repeated measures logistic regression models indicated that there were significant site, year, and site*year interactions for yearly MDI and INI scores (Table 6). Specifically, both study sites suffered from increased degradation and decreased integrity over time, but HR wetlands were subjected to more extensive degradation that those at Webb WMA.

Cumulative measures of our indices indicated that HR wetlands suffered more degradation, and thus exhibited less integrity, than Webb WMA wetlands. MDI_c scores differed between sites (Pooled T-test, df = 81, t = 10.27, $P < 0.001$), averaging 0.64 (SD = 0.17) and 0.24 (SD = 0.18) for Hamilton Ridge and Webb, respectively. INI_c values were significantly lower (Satterthwaite T-test, df = 72.3, t = 11.12, $P < 0.0001$) for Webb WMA (mean = 0.18; SD = 0.13; Range = 0.00-0.49) than for Hamilton Ridge WMA (mean = 0.59; SD = 0.20; Range = 0.23-1.00). Similarly, isolated wetlands at Webb WMA (mean = 0.18, SD 0.14, range = 0.00-049) had more integrity than Hamilton Ridge wetlands (mean = 0.58, SD = 0.19, range = 0.24-0.99; Pooled T-test, df = 81, t = 9.77, $P < 0.0001$).

Final LGI scores indicated which ponds had the highest integrity, and thus the best restoration potential. Isolated wetlands on Webb WMA had higher relative integrity than those on Hamilton Ridge, but LGI scores indicated which Hamilton Ridge wetlands were best suited for restoration efforts (Table 7). When all 84 ponds were ranked according to their final LGI score (Appendix 1), only one HR wetland ranked within the top 20 (Pond ID = 22; LGI = 0.24) and only two ranked with the top 25 ponds across both study areas.

Index values were included as attribute data for the GIS coverages of isolated wetlands generated from for HR and the Woodbury Tract. The index values calculated in this study can help managers prioritize restoration as well as identify locations likely to contain rare or imperiled species (Fig. 9). We intend to validate index performance using biological data. Specifically, we will use the results of auditory frog call surveys conducted during winter and spring 2010-2011 assess index performance. Frog call data will allow us to assess index performance for predicting breeding locations with high species diversity, particularly for southeastern pine savanna endemics (e.g., ornate chorus frog, *Psuedacris ornata*).

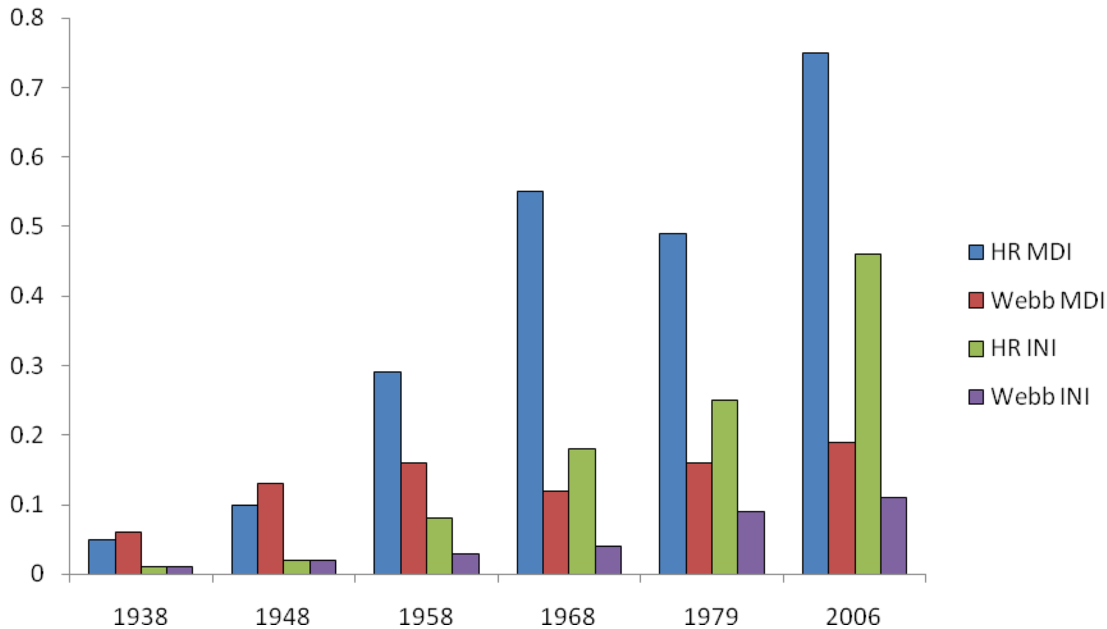

Figure 7. Average yearly mechanical degradation indices (MDI) and integrity indices (INI) for isolated wetlands at Hamilton Ridge WMA (HR; N = 56) and Webb WMA (Webb; N = 27), derived using aerial photographs taken in 1938, 1948, 1958, 1968, 1978, and 2006. Values of both indices are bound between 0 and1. MDI = 1 refers to extensive mechanical degradation, MDI = 0 refers to limited or no visible degradation. INI = 1 refers to limited or no integrity, INI = 0 refers to greater integrity.

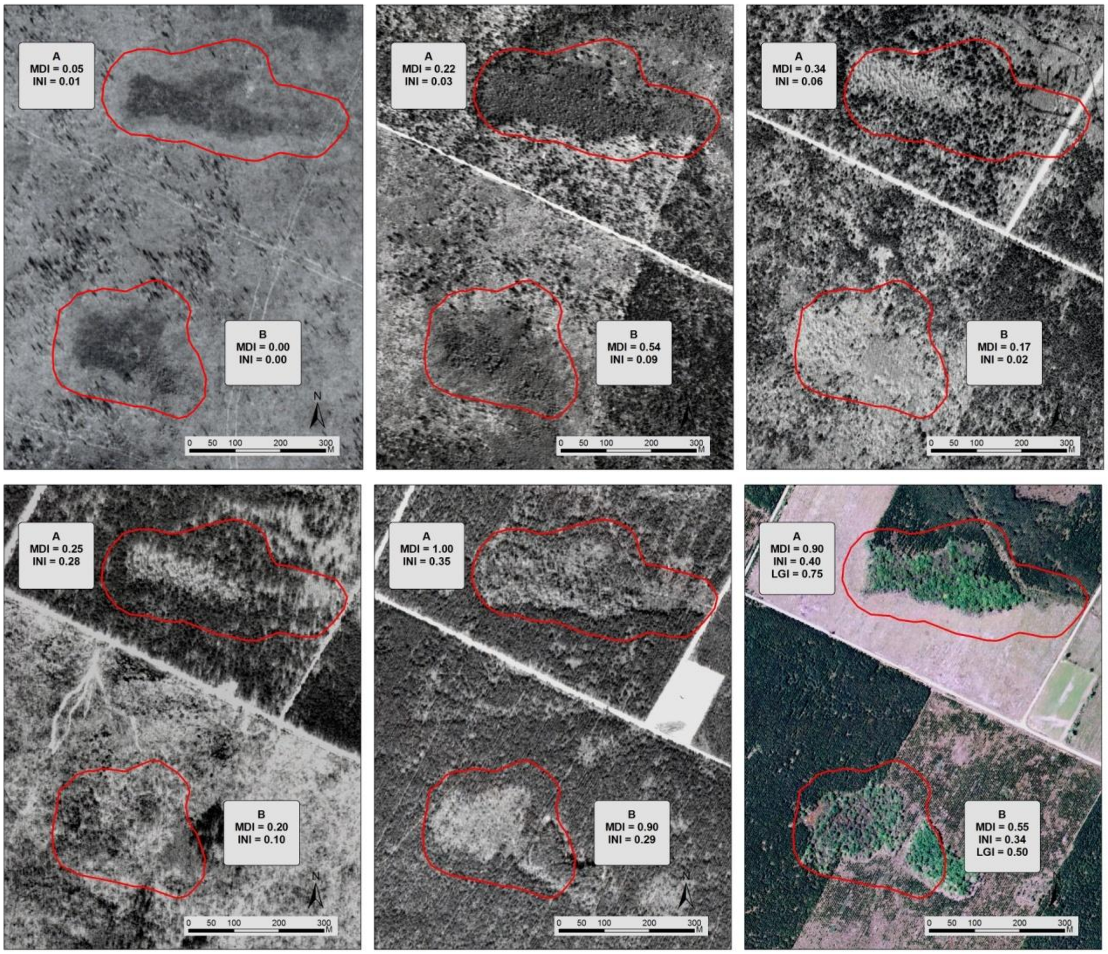

Figure 8. Historical aerial photographs were used to assess the effects of roads, ditches, and logging on isolated wetlands at the Hamilton Ridge, Woodbury Tract, and Webb WMA. These images depict landscape alterations over time (i.e., 1938, 1948, 1958, 1968, 1978, and 2006, sequentially ordered left to right) at two isolated wetlands on Hamilton Ridge. Historical aerials allowed isolated wetlands to be assessed in a dynamic manner that captured temporal patterns of degradation. This series captures the effects of industrial forestry on isolated wetland integrity. Three indices were used to assess degradation and loss of integrity, i.e., the mechanical degradation index (MDI), integrity index (INI), and the legacy index (LGI); scores from these indices are depicted on images.

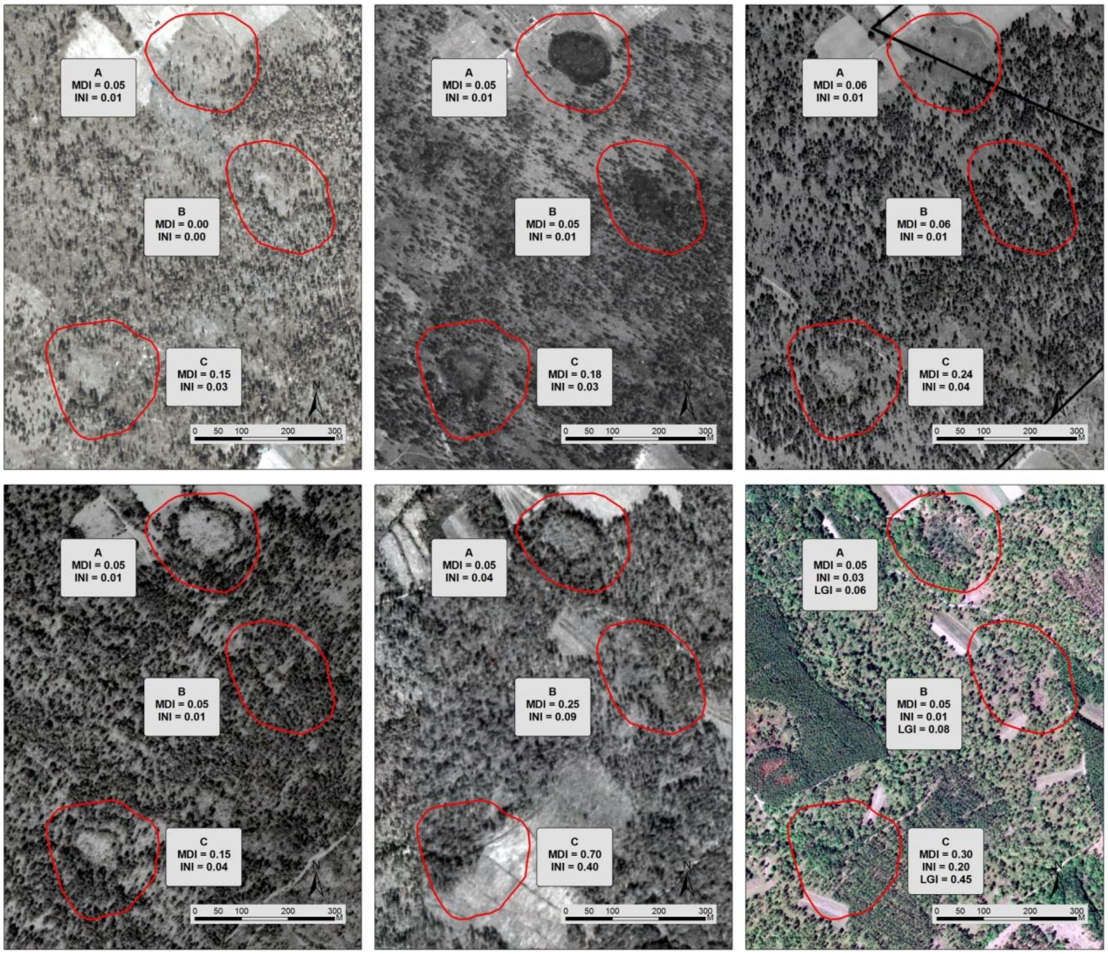

Figure 9. A long history of prescribed fire maintained ecological integrity of isolated wetlands on the Webb WMA, despite some degree of mechanical degradation. These images depict three isolated wetlands over time (i.e., 1938, 1948, 1958, 1968, 1978, and 2006, sequentially ordered left to right). Although pond C suffered extreme degradation is currently no visible in photographs, wetlands A and B maintained high integrity through time, and are known breeding locations of rare anurans. Assessing integrity over time allows managers to identify wetlands of conservation interest. Three indices were used to assess degradation and loss of integrity, i.e., the mechanical degradation index (MDI), integrity index (INI), and the legacy index (LGI); scores from these indices are depicted on images.

Table 6. Model outputs from two repeated measures logistic regression models examining the influence of year, study site, and their interaction on mechanical degradation index (MDI) and integrity index (INI) scores for isolated wetlands at Hamilton Ridge WMA (HR; N = 56) and Webb WMA (Webb; N = 27).

Index	Parameter	Estimate ± SE	Z	*P*
MDI	Intercept	−1.44 ± 0.20	−7.30	< 0.0001
	Year (1938)	−1.22 ± 0.32	−3.88	0.0001
	Year (1948)	−0.49 ± 0.27	−1.80	> 0.05
	Year (1958)	−0.24 ± 0.24	−1.02	> 0.05
	Year (1968)	−0.54 ± 0.24	−2.23	< 0.05
	Year (1979)	−0.18 ± 0.30	−0.58	> 0.05
	Study Site (HR)	2.52 ± 0.27	9.25	< 0.0001
	Year*Site (1938_HR)	−2.81 ± 0.39	−7.16	< 0.0001
	Year*Site (1948_HR)	−2.79 ± 0.40	−6.85	< 0.0001
	Year*Site (1958_HR)	−1.79 ± 0.32	−5.38	< 0.0001
	Year*Site (1968_HR)	−0.32 ± 0.35	−0.92	> 0.05
	Year*Site (1978_HR)	−0.94 ± 0.39	−2.43	> 0.01
INI	Intercept	−2.08 ± 0.19	−10.69	< 0.0001
	Year (1938)	−2.40 ± 0.31	−7.77	< 0.0001
	Year (1948)	−1.74 ± 0.26	−6.62	< 0.0001
	Year (1958)	−1.35 ± 0.29	−4.65	< 0.0001
	Year (1968)	−1.17 ± 0.32	−3.68	< 0.001
	Year (1979)	−0.19 ± 0.31	−0.63	> 0.05
	Study Site (HR)	1.90 ± 0.24	8.07	< 0.0001
	Year*Site (1938_HR)	−2.05 ± 0.38	−5.36	< 0.0001
	Year*Site (1948_HR)	−2.0 ± 0.3	−5.56	< 0.0001
	Year*Site (1958_HR)	−0.87 ± 0.33	−2.68	< 0.01
	Year*Site (1968_HR)	−0.16 ± 0.35	−0.44	> 0.05
	Year*Site (1978_HR)	−0.71 ± 0.36	−1.94	> 0.05

Table 7. The five top-ranking isolated wetlands at Webb WMA (Webb) and Hamilton Ridge WMA (HR), respectively, according to the legacy index (LGI), in which 0 = highest integrity and 1 = lowest integrity, relative to ponds within the two study areas.

Pond ID	Study Site	LGI
33	Webb	0.00
36	Webb	0.00
43	Webb	0.00
47	Webb	0.03
34	Webb	0.05
22	HR	0.24
2	HR	0.24
61	HR	0.31
19	HR	0.35
20	HR	0.35

Appendix 1. Isolated wetlands on Hamilton Ridge WMA (Ham) and Webb WMA (Webb) ranked according to the legacy index (LGI), in which 0 = highest integrity, and thus best restoration potential, 1 = lowest integrity and worst restoration potential.

Id	Location	LGI
33	Webb	0.00
36	Webb	0.00
43	Webb	0.00
47	Webb	0.03
34	Webb	0.05
28	Webb	0.06
29	Webb	0.07
79	Webb	0.08
39	Webb	0.10
49	Webb	0.11
27	Webb	0.13
35	Webb	0.13
44	Webb	0.14
32	Webb	0.16
48	Webb	0.16
37	Webb	0.17
81	Webb	0.19
31	Webb	0.23
40	Webb	0.23
22	Ham	0.24
2	Ham	0.24
30	Webb	0.25
80	Webb	0.27
61	Ham	0.31
46	Webb	0.31
26	Webb	0.33
19	Ham	0.35
20	Ham	0.35
70	Ham	0.36
62	Ham	0.37
45	Webb	0.37
9	Ham	0.38
41	Webb	0.38
1	Ham	0.40
64	Ham	0.40
17	Ham	0.42
63	Ham	0.42

82	Ham	0.43
6	Ham	0.44
38	Webb	0.45
77	Ham	0.46
73	Ham	0.46
53	Ham	0.47
23	Ham	0.48
4	Ham	0.48
16	Ham	0.49
25	Webb	0.49
8	Ham	0.50
7	Ham	0.50
72	Ham	0.51
18	Ham	0.52
15	Ham	0.53
54	Ham	0.53
12	Ham	0.53
13	Ham	0.54
50	Ham	0.55
59	Ham	0.55
76	Ham	0.55
74	Ham	0.56
51	Ham	0.57
67	Ham	0.57
66	Ham	0.59
3	Ham	0.60
65	Ham	0.61
10	Ham	0.63
55	Ham	0.67
78	Ham	0.68
14	Ham	0.70
24	Ham	0.75
57	Ham	0.75
52	Ham	0.76
69	Ham	0.79
56	Ham	0.80
60	Ham	0.81
21	Ham	0.81
68	Ham	0.84
58	Ham	0.85
84	Ham	0.88

71	Ham	0.89
75	Ham	0.91
11	Ham	0.93
5	Ham	0.95
83	Ham	1.00

<u>Significant Deviations</u>: None

Total Federal Cost : $70,500

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