Received: 3 January 2017 | Revised: 11 March 2017 | Accepted: 21 March 2017
DOI: 10.1002/eco.1862

WILEY

RESEARCH ARTICLE

Source water contributions and hydrologic responses to simulated emerald ash borer infestations in depressional black ash wetlands

Matthew J. Van Grinsven[1,2] | Joseph P. Shannon[2] | Joshua C. Davis[2] | Nicholas W. Bolton[2] | Joseph W. Wagenbrenner[2,3] | Randall K. Kolka[4] | Thomas G. Pypker[2,5]

[1] Department of Earth, Environmental and Geographical Sciences, Northern Michigan University, Marquette, MI, USA

[2] School of Forest Resources and Environmental Science, Michigan Technological University, Houghton, MI, USA

[3] USDA Forest Service Pacific Southwest Research Station, Arcata, CA, USA

[4] Forestry Sciences Lab, USFS Northern Research Station, Grand Rapids, MN, USA

[5] Natural Resource Sciences, Thompson Rivers University, Kamloops, BC, Canada

Correspondence
Matthew J. Van Grinsven, Department of Earth, Environmental and Geographical Sciences, Northern Michigan University, Marquette, MI, USA.
Email: mvangrin@nmu.edu

Funding Information
USDA Forest Service; US-EPA Great Lakes Restoration Initiative, Grant/Award Number: DW-12-92429101-0; Michigan Tech's Center for Water and Society; Michigan Tech's Ecosystem Science Center

Abstract

Forested wetlands dominated by black ash (*Fraxinus nigra*) are currently threatened by the rapid expansion of the exotic emerald ash borer (EAB; *Agrilus planipennis*, Coleoptera: Buprestidae) in North America, and very little is known about the hydrology and ecology of black ash wetlands. The ecohydrological response of forested wetlands following a canopy disturbance has the potential to affect critical ecosystem services, and the degree of this effect may largely depend on the wetland's hydrogeological setting. The main objectives of this study were to characterize the hydrologic connectivity of uninfested black ash wetlands and evaluate the water table response to a simulated EAB disturbance. We hypothesized that black ash wetlands in northern Michigan were (a) seasonally connected to, and derived the majority of their water from groundwater, and (b) wetland water tables would be elevated following a simulated EAB infestation due to decreased transpiration with the loss of black ash. The results indicate that the black ash wetland sites received most of their water from groundwater discharge. Significantly smaller site transpiration fluxes and significantly slower rates of drawdown were detected during the growing season in the girdled and ash-cut treatment sites, and these responses collectively produced significantly elevated wetland water tables when compared to control sites in the latter portions of the growing season. However, the wetlands' strong connection with groundwater sources likely buffered the magnitude of hydrological responses associated with the loss of black ash from the landscape.

KEYWORDS

black ash, emerald ash borer, forested wetlands, groundwater, invasive pest disturbance, natural tracers, wetland hydrology

1 | INTRODUCTION

Invasive species, such as disease organisms and exotic insects, have caused extensive tree mortality throughout North America's forested landscapes (Ellison et al., 2005; Kenis et al., 2008). Understanding ecohydrological responses to non-native species invasions has been targeted as a critical research need for the 21st century (Vose et al., 2011). Within the limited existing body of research linking invasive species and ecohydrological responses (Brantley, Miniat, Elliott, Laseter, & Vose, 2015; Ford & Vose, 2007; Nagler et al., 2009), even less research has focused on exotic insect disturbances in forested wetland landscapes (Davis, Shannon, Bolton, Kolka, & Pypker, 2016; Slesak, Lenhart, Brooks, D'amato, & Palik, 2014). The rapid and extensive spread of emerald ash borer (EAB; Buprestidae: *Agrilus planipennis*) populations is considered an important ecological and economic disturbance as it has the potential to decimate existing ash stands throughout North America (Gandhi & Herms, 2010; Kovacs et al., 2010).

Most research on EAB and ash trees has focused on dispersal (Prasad et al., 2009), stand mortality (Smitley, Davis, & Rebek, 2008), and biological control (Liu & Bauer, 2008) with less attention given to the ecosystem impacts. Recent research has highlighted the impact of EAB on regional ash-forest carbon cycling and stand diversity (Flower & Gonzalez-Meler, 2015; Flower, Knight, & Gonzalez-Meler, 2013), examined the impact of simulated EAB infestations on water tables in mineral-soil black ash wetlands (Slesak et al., 2014), and evaluated vegetation responses to real and simulated EAB infestations (Costilow, Knight, & Flower, 2017; Davis et al., 2016; Looney, D'amato, Palik, Slesak, & Slater, 2017). To our knowledge, no studies have coexamined source water contributions and water table response to an EAB infestation in ash (genus: *Fraxinus*)-dominated forests. Specifically, it is unknown how EAB-induced mortality or suppression of the major overstory component in depressional black ash (*Fraxinus nigra*) wetlands will affect wetland water tables.

The hydrological setting is the most important environmental factor associated with the formation and function of wetlands (Fretwell, Williams, & Redman, 1996), and water plays an integral role in physical, chemical, and biological processes. Disturbance to the dominant living component of a forested wetland may affect the wetland's water table position due to the decreased amount of transpiration and interception losses (Sun et al., 2001). Specific impacts cannot be attributed to an EAB-induced disturbance until a comprehensive assessment of site hydrology is conducted.

Black ash wetlands in the Upper Peninsula of Michigan are located in landform depressions within headwater catchments and are commonly inundated following spring snowmelt. Both groundwater and throughfall, defined here as the gross rainfall that drips from foliage or branches of canopy or passes directly through the canopy (Bahmani, Attarod, Bayramzadeh, Ahmadi, & Radmehr, 2012), likely contribute water to these wetlands, but the relative contribution of each source and how the proportions change throughout the year are not known. Stable isotopes and hydrometric monitoring methods have been successfully used to partition groundwater and precipitation contributions to headwater streams, riparian areas, and wetlands (Devito, Hill, & Roulet, 1996; Fitzgerald, Price, & Gibson, 2003; Kolka, Giardina, Mcclure, Mayer, & Jurgensen, 2010). Furthermore, Clay, Bradley, Gerrard, and Leng (2004) and Ferone and Devito (2004) examined lateral connectivity between groundwater and surface water in headwater watersheds, and Hunt, Bullen, Krabbenhoft, and Kendall (1998) identified seasonal patterns of groundwater and precipitation contributions to wetland mixtures using hydrometric and natural tracer monitoring methods. Moreover, the type of groundwater connection, defined as either a local, intermediate, or regional flow system by Tóth (1963) and Winter (1998), can have a substantial effect on the chemistry and biological communities within ecosystems located at the intersection of groundwater and surface water (Winter, 2001) as well as influence run-off production, degree of water table fluctuations, chemical processes, and sensitivity to disturbance (Devito & Hill, 1997; Roulet, 1991; Sebestyen, Verry, & Brooks, 2011).

Numerous studies have focused on disturbance events in forested wetland landscapes associated with timber harvesting activities (Kolka, Sebestyen et al., 2011; Sun et al., 2001; Trettin et al., 1995). Because timber harvesting in northern forested wetland landscapes often occurs when snow cover is present, the soil compaction and erosion impacts typically associated with logging operations can be reduced (Grigal & Brooks, 1997). Because very few studies have investigated invasive pest disturbances in forested wetland landscapes, timber harvest impact studies provide valuable insight.

Timber harvesting activities have been shown to increase water yields and cause elevated water tables from undrained black spruce (*Picea mariana*) lowlands and peatlands in Ontario, Canada, (Haavisto, Fleming, & Skeates, 1988; Pothier, Prevost, & Auger, 2003) and other forested wetlands in Quebec, Canada (Dubé, Plamondon, & Rothwell, 1995). In contrast, black spruce peatlands in northern Minnesota were shown to be hydrologically resilient to disturbance, as effects on water yield and water table fluctuations were not detected following timber harvest (Sebestyen et al., 2011; Verry, 1986). Vegetative response, micrometeorology, and the degree of hydrological connectivity with regional groundwater sources have been shown to mitigate expected water table responses and the duration of hydrologic recovery following disturbance events (Pothier et al., 2003; Sebestyen et al., 2011; Verry, 1986).

Slesak et al. (2014) showed that water tables were significantly higher in black ash wetland complexes following harvest treatment when compared to control sites. In contrast to the mineral soil wetland complexes in northern Minnesota, northern Michigan black ash wetlands used for this study have Histosol soils, are considerably smaller (0.25–1.25 ha), and exist as isolated depressions on the landscape. For these reasons, we expect that water tables may respond differently in Michigan black ash wetlands and believe that these results will augment the currently limited body of research surrounding invasive pest disturbances to forested wetlands.

In this study, we combined catchment hydrology and disturbance response monitoring components to better understand the processes and mechanisms that regulate water table position and hydrologic response in black ash wetlands. Specifically, our objectives were to (a) partition source water contributions and investigate the timing and direction of groundwater-wetland water interaction using hydrometric monitoring and stable isotope tracers and (b) evaluate the response of water tables to a simulated EAB disturbance by monitoring wetland water levels, precipitation, and overstory transpiration. Specifically, it is unknown whether black ash wetlands are primarily recharged by groundwater or precipitation, and if black ash wetland water tables are sensitive to disturbance. We hypothesize that (a) groundwater is the predominant source of water contributing to depressional black ash wetlands and (b) wetland water tables will be elevated as a result of the decreased amount of transpiration in girdled and ash-cut treatments sites. These objectives and hypotheses will augment the currently limited understanding surrounding the hydrological implications of a looming EAB infestation.

2 | METHODS

2.1 | Study area

Black ash wetlands in the western Upper Peninsula of Michigan commonly occur in landform depressions surrounded by mixed hardwood upland species such as sugar maple (*Acer saccharum)* and white pine (*Pinus strobus)*. Other species, such as red maple (*Acer rubrum)*, yellow birch (*Betula alleghaniensis*), balsam fir (*Abies balsamea)*, and white cedar (*Thuja occidentalis)* commonly exist as lesser overstory components within black ash wetlands (Davis et al., 2016). Shrub, fern, and graminoid cover in the understory is significant but unevenly distributed. Black ash wetlands in Michigan's Ottawa National Forest are also characterized by seasonally inundated conditions and commonly have discernible surface drainage outlet channels.

Typical annual hydrographs consist of large spring snowmelt events, summertime baseflow conditions, and autumn rainfall driven run-off (Devito et al., 1996; Verry, 1997). Snowfall is a significant component of annual precipitation in the western Upper Peninsula of Michigan, which averages 478 cm annually (Arguez et al., 2012). Black ash wetland surface soils are characterized by a woody peat Histosol layer. Soils in our study sites were comprised of 75% organic matter and 47% organic carbon on average. Organic soil profile thicknesses in the study sites ranged between 5 cm and greater than 690 cm, with an average depth of 140 cm, and a clay lens or a poorly sorted clay-loam was commonly detected at the bottom of the organic layer.

Microtopography within three affiliated black ash wetland study sites was determined by comparing the absolute elevation difference between paired hummock and hollow locations within 1 m. The average absolute vertical difference between paired locations (n = 900) was 13.5 cm with a standard error of 0.2 cm, and the range was between 2.0 and 63.5 cm. The transition between wetlands and adjacent uplands was generally abrupt, and laggs were not visible near the perimeter.

The Gogebic soil series surrounded sites in treatment Groups 2 and 3, and the Graveraet and Stutts soil series surrounded treatment Group 1 sites (Soil Survey Staff). Gogebic soil family texture is classified as coarse-loamy, and horizon textures range from fine sandy loam to gravelly sand. Permeability of the unaltered parent material horizon is characterized as moderate to rapid in Gogebic soils. Graveraet and Stutts soil family textures are classified as coarse-loamy and sandy, respectively, and horizon textures range from loam to sand. Permeability of the unaltered parent material horizons ranged from moderately slow to moderate in Graveraet and high to very high in Stutts.

All study sites are underlain and surrounded by Quaternary sediments, and the majority of sites are underlain and surrounded by 30–60 m of glacial till or coarse-grained stratified sediments (Soller, Packard, & Garrity, 2012). The three sites in treatment Group 3 (see Figure 1) are underlain and surrounded by 0–15 m of patchy quaternary sediments. Excluding Group 3 sites, the average extent of deep glacial till or coarse-grained stratified sediments extended 6.1 km away from study sites before encountering shallower patchy Quaternary sediments. Beneath the Quaternary deposits, the Group 1 sites are underlain by Jacobsville sandstone, the Group 2 sites are underlain by clastic-sedimentary and metamorphic rocks of the Michigamme formation, and the Group 3 sites are underlain by Archean granite and gneissic-metamorphic bedrock (MGDL, 1987).

2.2 | Experimental design

Nine sites within the Ottawa National Forest ranging in size from 0.25 to 1.25 ha were established in 2011 in isolated landform depressions within first-order watersheds (Figure 1). Sites were grouped into blocks constrained by geographic proximity using the blockTools package (Moore, 2012; R Core Team, 2015) and also using an algorithm that minimized among-treatment variability in percent *F. nigra* basal area, depth of organic soil, and site area. Control, girdle, and ash-cut treatments were then randomly assigned resulting in one site of each treatment type in each of three geographically constrained blocks. A 1-year baseline data collection period (2012) preceded the 2-year post-treatment monitoring period (2013–2014) of the study. Treatments were applied between November 2012 and February 2013.

The girdle treatment was used to mimic an end-stage EAB infestation, wherein complete stand mortality may take 3–4 years (Cappaert, Mccullough, Poland, & Siegert, 2005; McCullough & Katovich, 2008). In the girdled sites, all black ash trees greater than 2.5-cm diameter at breast height (DBH) had the bark, cambium, and phloem removed

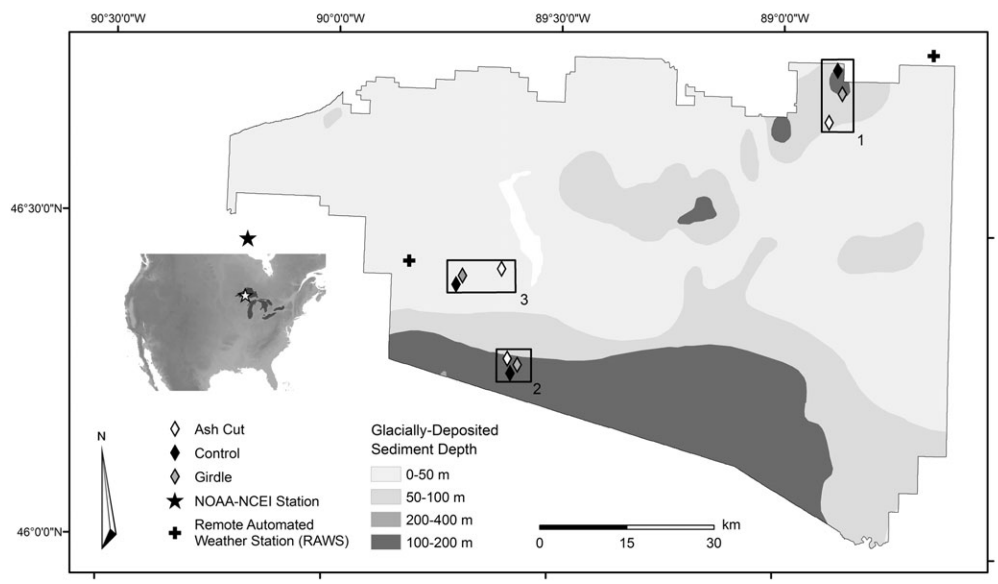

FIGURE 1 Map of Ottawa National Forest and black ash wetland study sites. Treatment sites (shaded diamonds) were combined into three treatment blocks (rectangles). The large white polygon east of Lake Gogebic is private land. Regional precipitation data were recorded at the National Weather Service station in Ironwood, Michigan, USA (NOAA-NCEI; star) and localized remote automated weather stations in Pelkie and Wakefield, Michigan, USA (RAWS; plus)

in a 60-cm tall circumferential band between November 2012 and January 2013. Based upon mean litter deposition measurements, approximately 25% of the ash canopy leafed out during the first post-treatment year, and approximately 10% ash canopy leafed out during the second post-treatment year (Davis et al., 2016). Most girdled ash trees exhibited epicormic branching beneath the girdle wound during 2013 and 2014.

The ash-cut treatment was used to mimic longer term post-disturbance changes in ecosystem conditions. In the ash-cut treated sites, all black ash trees greater than 2.5-cm DBH were hand-felled with chainsaws where stumps were up to 1.2 m tall because of the deep snow during the winter of 2013. Most ash stumps exhibited sprouting during 2013 and 2014 that grew to heights of more than 6 m by 2016 (Davis et al., 2016).

2.3 | Instrumentation and monitoring

A combination of automated measurement and manual sampling methods were used to examine hydrological connectivity, source water contributions, and water table responses in the nine sites. Routine monthly grab samples of accumulated throughfall, wetland soil-pore water, and upland groundwater were collected from June–October of each year (2012–2014).

Bulk density samples were collected during a 2012 peat carbon soil survey (Chimner, Ott, Perry, & Kolka, 2014). Saturated hydraulic conductivity (K_{sat}) estimates were derived from a relationship established between soil bulk density and K_{sat} for wetland soils in the Great Lakes region (Boelter, 1969). The K_{sat} estimates were used to determine equivalent types of mineral soils following guidelines established by the Natural Resource Conservation Service (Soil Survey Staff).

Two co-located monitoring wells were installed in hollows within each wetland to a depth of approximately 1.5 m. Three-instrument cluster locations were randomly established in 2011, and the wetland monitoring wells were installed within the instrument cluster nearest the outflow channel. Monitoring well locations consisted of one 5 cm inner-diameter (ID) metal monitoring well was used for water table measurements, and one co-located 5-cm ID polyvinyl chloride monitoring well used exclusively for soil water sampling. Two 3.2 m ID metal monitoring wells were also installed in the adjacent upland mineral soils to a depth of approximately 2.5 m. The surface elevations for upland wells were located approximately 1 m above the wetland soil surface, and the upland wells were installed along the largest hills adjacent to two of the randomly established wetland instrument clusters. All wells were surveyed with a total station (Leica TC600, Leica Geosystms Inc. Norcross, Georgia, USA). Surface elevations were computed relative to a local benchmark (Fraser, Roulet, & Lafleur, 2001), and calculated elevations were used to compute hydraulic gradients.

Wetland water tables were monitored with a pressure transducer (Levelogger Junior M5, Solinst Canada Ltd., Georgetown, Ontario, Canada) and recorded at 15 minute intervals during snow-free periods. Water level conversions were adjusted for atmospheric pressure using a barometer located in each treatment block (HOBO Barometric Pressure Smart Sensor, Onset Computer Corporation). Water table levels were manually confirmed during monthly site visits with a water level meter (102M Water Level Meter, Solinst Canada Ltd.).

Upland groundwater wells were purged one to three times prior to collection of water samples. Three full casing volumes were extracted prior to sampling (USGS, 2006) when water tables intersected the highly conductive E, B and 2E horizons commonly detected closer to the surface in fine sandy loam soils found in till landforms (Soil Survey Staff). When water tables were located in the poorly sorted glacial till commonly located near the bottom of the monitoring well, all water was completely purged from the well at least once and allowed to recharge for approximately 90 min prior to sampling.

A 25 cm long dual inflatable packer assembly (EPA, 2006) and low flow sampling technique (Puls & Barcelona, 1996) was used to sample near surface wetland soil water. The packer assembly was rinsed with deionized water and then with sample water prior to being lowered into the monitoring well. When the water table was greater than 5 cm beneath the surface, the upper packer was located at 5 cm depth. In all other cases, the upper packer was located at the water table surface. A water sample was retrieved after the dissolved oxygen observations (ProODO, YSI Inc., Yellow Springs, OH, USA) within the isolated segment of well screen stabilized.

Six funnel-type throughfall collectors (Glaubig & Gomez, 1994), constructed from 72.4 cm tall by 10.2-cm ID polyvinyl chloride pipe and a 10.2-cm ID high density polyethylene funnel, were placed in pairs spaced 6 m apart in each of the three randomized instrument cluster locations. The top of each collector was located 1.25 m above the ground surface. Water was removed from the bottom of each collector through a stopcock drain plug during monthly visits. Water samples were collected from each collector and composited into one bulk sample for each site. To prevent evaporation, 25 ml of food-grade mineral oil was inserted into each collector prior to each sample period.

Regional weather was summarized using 30-year monthly average (1981–2012) precipitation data (Arguez et al., 2012) for Ironwood, MI, and daily data for 2012, 2013, and 2014 water years (NOAA-NCEI). Recording tipping bucket rain gages (HOBO Rain Gage, Onset Computer Corporation, Bourne, MA) were co-located in open areas within 1.3 km of each site and were used to calculate daily gross precipitation during snow-free seasons.

2.4 | Hydrometric and isotopic analysis

Hydraulic gradients were computed between the upland and wetland water tables for each site for the snow-free seasons (June–October) for 2012–2014 according to Equation 1:

$$i = \frac{h_2 - h_1}{l}, \tag{1}$$

where i was the hydraulic gradient, h_2 was the hydraulic head (m) within the upland groundwater monitoring well, h_1 was the hydraulic head (m) within the wetland groundwater monitoring well, and l was the horizontal length (m) between the upland and wetland wells. Positive hydraulic gradients indicate that upland groundwater table elevations were higher than wetland groundwater table elevations. Because distances between wetland monitoring wells and upland monitoring wells were variable and there were no monitoring wells located near the perimeter of the wetland, relative magnitudes were not specifically evaluated and hydraulic gradient data were summarized using a binary condition as either gaining or losing.

Monthly water samples were collected from wetland soil pore water (n = 1), throughfall (n = 6), and groundwater (n = 2) during the snow-free season (June–October) over the study period. Both throughfall and groundwater samples were composited during monthly sampling events to minimize influence of space and time invariance, stored in 6-ml high density polyethylene containers, and immediately wrapped with Parafilm (Bemis Company Inc., Oshkosh, WI, USA) to prevent evaporation-induced fractionation prior to analysis. Samples were kept in the dark in a temperature-controlled laboratory prior to analysis for deuterium (^{2}H) and oxygen-18 (^{18}O) isotopic composition on a liquid water isotope analyzer (Los Gatos Research Inc., Mountain View, CA, USA). The stable isotopic composition of the sample was reported as delta (δ) per mill (‰) relative to the Vienna Standard Mean Ocean Water standard and the stable isotopic composition (Kendall & Caldwell, 1998).

EPA's IsoError (Phillips & Gregg, 2001) program, a linear mixing model using δ^{2}H, was used to partition source water contributions and calculate confidence intervals for these estimates. One mixture with two end member mixing models were estimated in IsoError. The mass balance equations (Equation 2) defined by Phillips and Gregg (2001) were used to estimate the contribution of source waters to wetland groundwater mixtures according to Equation 3:

$$\begin{aligned}\bar{\delta}_{WS} &= f_{TF}\bar{\delta}_{TF} + f_{GW}\bar{\delta}_{GW} \\ 1 &= f_{TF} + f_{GW},\end{aligned} \tag{2}$$

$$f_{TF} = \frac{\bar{\delta}_{WS} - \bar{\delta}_{GW}}{\bar{\delta}_{TF} - \bar{\delta}_{GW}}, \tag{3}$$

where $\bar{\delta}$ represents the mean isotopic composition (δ^{2}H) for the wetland soil water mixture (*WS*) and throughfall (*TF*) and groundwater (*GW*) sources, respectively; and f represents proportions of sources (*TF* and *GW*) and a mixture (*WS*). Proportional contributions of *TF* and *GW* were estimated for the *WS* mixture for each sampling event to ensure mixing process linearity assumptions of end-member mixing analysis were met.

Monthly source water contributions were estimated for the wetland soil-pore water mixture for all nine sites during 2012 and for six sites during 2013. Mixing models were not computed if the upland monitoring well was dry during routine monthly site visits and were discarded if end-member contributions were greater than 100% or less than 0% of the mixture. Thirty-eight models (of 93) were discarded.

2.5 | Transpiration

Transpiration estimates for each study site were generated using models based on sap-flux measurements of the three most common overstory species (*F. nigra*, *B alleghaniensis*, and *A. rubrum*), which together comprised an average of 88% of the overstory basal area (Davis et al., 2016). Measurements were recorded in all of the control and girdle sites, and models were used to estimate transpiration in the ash-cut sites. Measurements of sap flux were made following the techniques described in Lu, Urban, and Zhao (2004) using two Granier-style thermal dissipation probes per stem in 11 *F. nigra*, eight *B. alleghaniensis*, and seven *A. rubrum* across the six control and girdle sites. For *B. alleghaniensis* and *A. rubrum*, raw voltage differentials were converted to sap flux following Granier's empirical equation (Granier, 1987; Lu, 1997) modified from Granier (1985). Conversion of the signal from *F. nigra* utilized an empirical equation derived for *Fraxinus excelsior* (Herbst, Rosier, Roberts, Taylor, & Gowing, 2007). For all species, corrected voltages were calculated when sapwood depth was less than the 2 cm probe length (Clearwater, Meinzer, Andrade, Goldstein, & Holbrook, 1999). Daily sap-flux estimates ($m^3 \cdot m^{-2} \cdot d^{-1}$) were generated using a mixed-effects model (Bates, Mächler, Bolker, & Walker, 2015; R Core Team, 2015) with the interaction of daylight normalized vapor pressure deficit (D_z; Oren, Zimmermann, & Terbough, 1996) and DBH as fixed effects and with uncorrelated random intercepts and slopes for each random effect of probe nested within species. Data to calculate D_z were taken from the remote automated weather stations in Pelkie, MI, (PIEM4) and Wakefield, MI, (WKFM4) at both the daily (Western Regional Climate Center, 2015) and hourly (MesoWest, 2015) time steps. Model coefficients adjusted for random effects were extracted at the species level and used to generate daily sap-flux rates for overstory trees located within the fixed-radius (11.3 m) vegetation plots.

Modeled sap-flux values were scaled to sap flow ($m^3 d^{-1}$) for each inventoried tree using equations from studies within the region for *A. rubrum* (Bovard, Curtis, Vogel, Su, & Schmid, 2005) and *B. alleghaniensis* (Tang et al., 2006), and sapwood area (A_s) estimated for each overstory tree using an empirical equation for *F. nigra* according to Equation 4:

$$A_s = 4.183 + 1.125^{*}DBH + 0.076^{*}DBH^2. \tag{4}$$

The radial decline in sap flux with increasing distance from the bark was modeled according to Pataki, Mccarthy, Litvak, and Pincetl (2011), where sapwood area estimates resulted in a sapwood depth of greater than the length of the probe. Sap flow estimates of inventoried trees were summed by species within a site and divided by the total area of the inventory plots by site to produce transpiration estimates (m^3 d^{-1}).

2.6 | Statistical analysis

All statistical analysis were performed in R (R Core Team, 2015). Linear mixed-effects models (LMM) were fit with the restricted maximum likelihood procedure in the lme4 package (Bates et al., 2015). The lsmeans package with a Tukey–Kramer adjustment (Lenth, 2013) was used for all pairwise comparisons, and degrees of freedom were calculated according to the Kenward–Roger approximation (Halekoh, Højsgaard, Højsgaard, & Matrix, 2014). The significance level was $p < .05$ for all comparisons. The MuMIn package (Barton, 2010) was used to calculate the conditional (r_c^2) and marginal (r_m^2) coefficients of determination. LMM terminology (Gałecki & Burzykowski, 2013) was used to describe LMM structure, and construction of LMMs for analysis of repeated measures data followed established guidelines (Baayen, Davidson, & Bates, 2008; Bates et al., 2015; Gałecki & Burzykowski, 2013). The predicted response surfaces for all LMMs were generated with the effects package (Fox, 2003).

Monthly isotopic signatures were used for calculation of summary statistics, and to test the response of isotopic signatures to grouping factors of sample type (*WS*, *GW*, and *TF*) and year (2012 and 2013) with a two-level LMM. The grouping factors were used as interacting fixed effects terms, and sampling event and site were used as crossed random effects. Logistical factors contributed to exclusion of 2014 for isotopic sampling and analyses.

Monthly manual water level measurements were used for calculation of summary statistics, and to test the response of wetland water levels to grouping factors of treatment and study period (pre-treatment or post-treatment) with a two-level LMM. The grouping factors were used as interacting fixed effects terms, and sampling event and site were used as crossed random effects.

Daily site transpiration estimates were calculated for each site from modeled transpiration data. The transpiration response to treatment (fixed effect) during pre-treatment and post-treatment study periods (fixed effect) was tested with a two-level LMM where a random intercept and fixed mean was generated for each year nested within site (nested random effects).

Wetland water level measurements recorded by loggers were used for the growing season drawdown and 24-hr precipitation response LMMs. The pressure sensors in two control study sites were affected by biofouling during 2012 and 2013, and these data were not used for analysis. Wetland water level data were available from one control site each in 2012 and 2013 and from three control sites in 2014. Inclusion of year within site as a nested random effect guided the structure of variance–covariance matrix and calculation of the degrees of freedom and therefore accounted for these missing data for both the growing season drawdown and 24-hr precipitation response LMMs. Zero-mean, normality, and homogeneity of variance assumptions were determined to be acceptable following visual examination of residual plots for all LMMs. Statistical comparisons were made for pre-treatment versus post-treatment data, where the similar post-treatment years were combined for explanatory purposes.

Daily wetland water level data recorded at midnight (WL_{day}) were normalized (WL_{Norm}) using a correction factor equal to the difference between the wetland soil surface and WL_{day} on June 15 for every study site, each year. Normalized values were used to compare both the mean water table position and rate of drawdown during the growing season (defined here as June 15–September 1) within and between treatments using a two-level LMM. The response of WL_{day} to date (day of year) was predicted for each treatment and study period (interacting fixed effects) with a random intercept and fixed mean for each year nested within site (nested random effects). An additional two-level LMM was constructed, where month (fixed effect) was substituted for day of year, and used to compare mean WL_{day} for each month.

Precipitation data from the co-located tipping buckets, transpiration estimates, and wetland water level data were summarized for each day in each study site. The difference between logged water level measurements (WL) recorded at midnight ($WL_{0:00}$) and 23:45 ($WL_{23:45}$) were used to calculate the 24-hr change in water level ($\Delta WL_{24\text{-hr}}$). Net water gain was calculated as the difference between the sums of precipitation (P) and the transpiration (T) for each day ($P–T_{24\text{-hr}}$) where $P–T_{24\text{-hr}}$ was greater than zero. The response of $\Delta WL_{24\text{-hr}}$ to $P–T_{24\text{-hr}}$ was predicted for each treatment and study period (interacting fixed effects) during the growing season with uncorrelated random intercepts and slopes for each year nested within site (nested random effects).

3 | RESULTS

3.1 | Wetland soils and regional climate

A distinct vertical gradation of physical organic soil properties existed within the study sites (Table 1). The lowest bulk densities and largest hydraulic conductivity estimates occurred near the surface. Bulk densities increased, and hydraulic conductivity estimates decreased in the deeper portions of the wetland soil profiles.

The study sites received slightly below average precipitation during 2012 and above average precipitation during 2013 and 2014, respectively (Figure 2). The Ottawa National Forest (ONF) received approximately two-and-one-half times more snow-water equivalent (SWE) precipitation during 2013 and two times more SWE during 2014 when compared to 2012. When compared to 30-year average snowfall data, the ONF received 36% less snow during 2012, 19% more snow during 2013, and 2% more snow during 2014. Moreover, the ONF received approximately 25% more rain during 2013 and 15% more rain during 2014 when compared to 2012. The timing of

TABLE 1 Physical and hydrological properties of the Histosol layer in study sites

Soil depth (cm)	Sample size	Average bulk density (g/cm^{-3})	95% Conf. interval	Est. K_{sat} (cm/s)	Mineral soil (K_{sat}) equivalent
0–15	10	0.09	0.03	1×10^{-3}	Sand
15–30	12	0.16	0.02	1×10^{-4}	Loam
30–50	15	0.23	0.02	1×10^{-5}	Clay

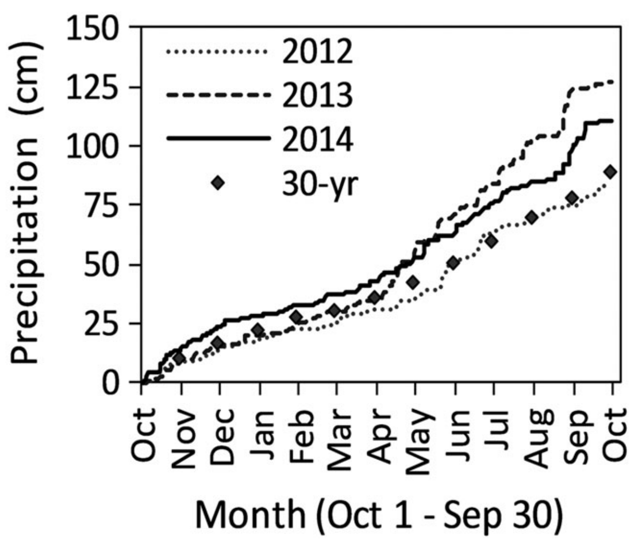

FIGURE 2 Cumulative precipitation for the 2012–2014 water years and the 30-year monthly average (1981–2010) precipitation at Ironwood, MI (NOAA-NCEI)

snowmelt was also considerably different among the 3 years. All sites were snow free by mid-March during 2012, whereas snowpack persisted in all sites beyond May 1 during 2013 and 2014. Furthermore, the total amount of precipitation received by mid-May was below average during 2012 and above average during 2013 and 2014, respectively (Figure 2).

3.2 | Source water contributions

Positive hydraulic gradients indicating gaining conditions (see Equation 1), were detected in eight of nine study sites during the spring and early summer each year (Figure 3). Positive hydraulic gradients were rarely detected in one study site (Group 2 girdle) throughout the entire study period. Positive hydraulic gradients were detected in at least four study sites during the entire 2012 snow-free season and in seven sites during the 2013 and 2014 snow-free seasons (Figure 3).

Average monthly wetland soil-pore water (*WS*) deuterium signatures closely resembled groundwater (*GW*) deuterium signatures, and both were much less variable during 2012 and 2013 when compared to throughfall (*TF*) deuterium signatures (Figure 4). The *WS* and *GW* standard errors (δ^2H ‰) were similar and less than *TF* standard errors during both 2012 and 2013 (Table 2). No significant differences were detected between mean *WS* and *GW* deuterium signatures during 2012 or 2013, but significant differences were detected when *TF* signatures were compared to *WS* and *GW* signatures during 2012 and 2013 (Table 2).

Isotopic mixing models indicate that *GW* contributed more source water than *TF* to black ash wetlands during 2012–2013 snow-free periods (Figure 5). The relative contribution of *TF* to *WS* increased during late summer and fall, but *GW* contributions were still greater than *TF*. On average, *GW* contributed 77% and 75% to *WS* during 2012 and 2013, respectively. The average modeled standard error for 2012 was 2.3% (n = 34), and for 2013 was 1.9% (n = 19).

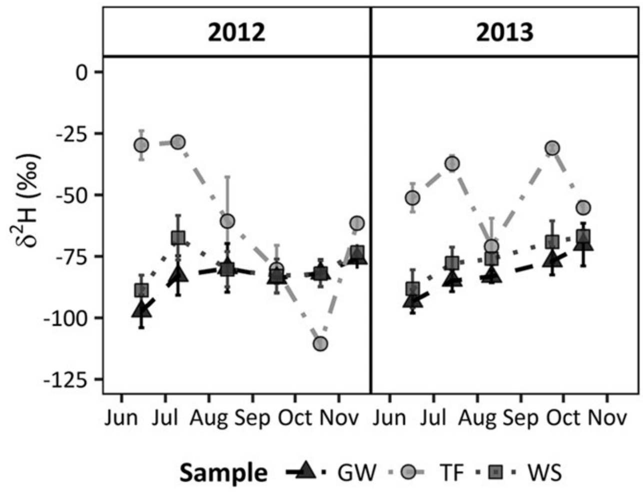

FIGURE 4 Monthly average deuterium (δ^2H) signatures with 95% confidence intervals for all source and mixture water samples collected during 2012 and 2013 where *WS* is the wetland soil water mixture, *TF* is throughfall, and *GW* is groundwater

TABLE 2 Monthly average (n = 7 for 2012 and n = 5 for 2013) deuterium signatures for each sample type (n = 3) and year (n = 2)

Study period	Sample type	n	Mean deuterium (δ^2H ‰)	Standard error (δ^2H ‰)	Mean test
2012	*WS*	60	−79.4	1.55	a
	GW	60	−85.7	1.57	a
	TF	60	−52.5	4.18	b
2013	*WS*	28	−76.1	1.85	a
	GW	28	−82.3	1.93	a
	TF	28	−49.6	3.05	b

Note. *WS* is the wetland soil water mixture, *TF* is throughfall, and *GW* is groundwater. Different letters "a" and "b" denote significant differences among treatment means.

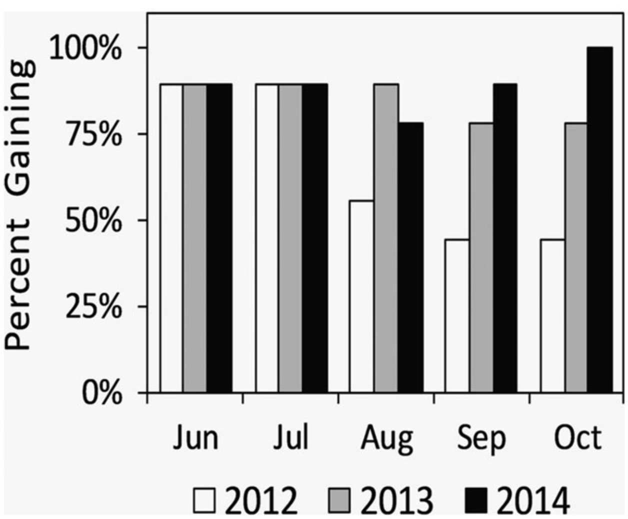

FIGURE 3 Percent of study sites where a positive hydraulic gradient (gaining conditions) were detected for each snow-free monthly site visit during the study period (2012–2014). Nine study sites (n = 9) were used for percent calculation for each sampling month

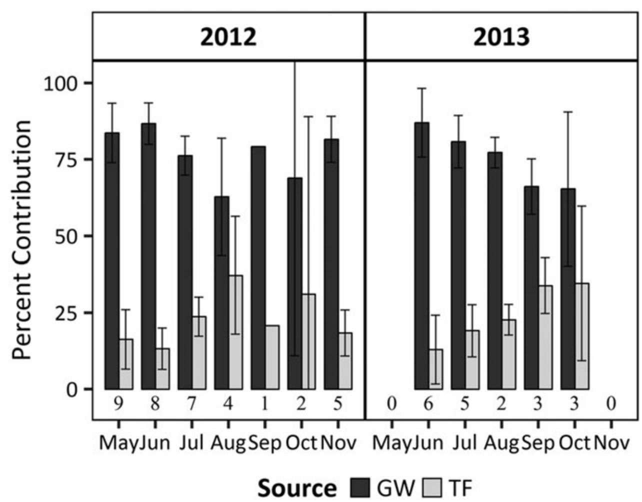

FIGURE 5 Source water contribution estimates, with 95% confidence intervals (whiskers), for black ash wetlands during 2012 and 2013. The number of study sites (n) used for each monthly estimate is shown above x-axis

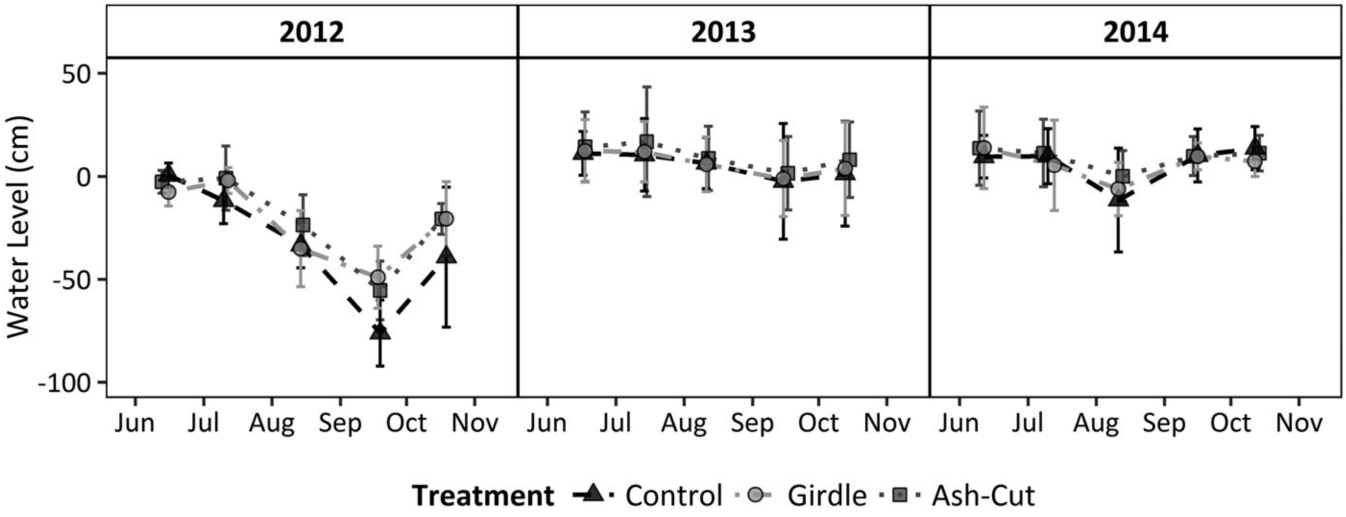

FIGURE 6 Average wetland water levels (points) with 95% confidence intervals (whiskers), relative to wetland soil surface (zero) recorded during monthly monitoring visits during snow-free periods (n = 3 for each treatment)

TABLE 3 Monthly-recorded (n = 5/year) wetland water level statistics for each study period and treatment

Study period	Treatment	n	Mean water level (cm)	Standard error (cm)
Pre	Control	15	−31.9	7.9
	Girdle	15	−22.8	5.4
	Ash-cut	15	−20.6	5.8
Post	Control	30	5.8*	2.8
	Girdle	30	6.3*	2.4
	Ash cut	30	9.6*	2.4

*Significant differences for within-treatment comparisons (between study periods) were detected.

3.3 | Hydrological responses

Wetland water table positions relative to the soil surface were lower, on average, during pre-treatment (2012) when compared to post-treatment (2013 and 2014) in all study sites (Figure 6 and Table 3). There were no significant differences detected between treatments during pre-treatment or post-treatment study periods for manual monthly recorded water level positions (Table 3).

Mean daily transpiration was 0.15, 0.13, and 0.13 cm d^{-1} for girdle, ash-cut, and control sites, respectively, during pre-treatment, and 0.05, 0.04, and 0.13 cm d^{-1} for girdle, ash-cut, and control sites, respectively, during post-treatment (Figure 7). No significant differences were detected among treatments during the pre-treatment study period. Transpiration was significantly lower in treated sites (girdle and ash cut) when compared to the control during the post-treatment study period due to the elimination of black ash transpiration, which accounted for 70% of site total on average.

Growing season drawdown rates and mean normalized water level positions (WL_{Norm}) were calculated for each treatment during pre-treatment and post-treatment study periods (Figure 8), and this LMM explained 64% of the r_m^2, 75% of the r_c^2, and had a residual standard error of 8.3 cm. No significant differences were detected among treatment drawdown rates during the pre-treatment period, but the −0.56 cm d^{-1} drawdown rate in the control was significantly greater than the −0.21 and −0.17 cm d^{-1} drawdown rates in girdle and ash-cut sites, respectively, during post-treatment (Figure 8). Post-treatment growing season drawdown rates were significantly lower in all sites when compared to the pre-treatment rates. No significant differences in WL_{Norm} were detected among treatments when the entire growing season was considered, but the girdle and ash-cut WL_{Norm} were significantly higher than the control in August during post-treatment. Mean August WL_{Norm} in the girdle and ash-cut sites were 20.4 cm and 21.9 cm higher during post-treatment than the control, respectively.

The 4.8 cm cm^{-1} slope of the $\Delta WL_{24\text{-}hr}$ response to $P–T_{24\text{-}hr}$ measured in the control sites was significantly larger than the 2.7 and 1.8 cm cm^{-1} slopes in the girdle and ash-cut sites, respectively, during

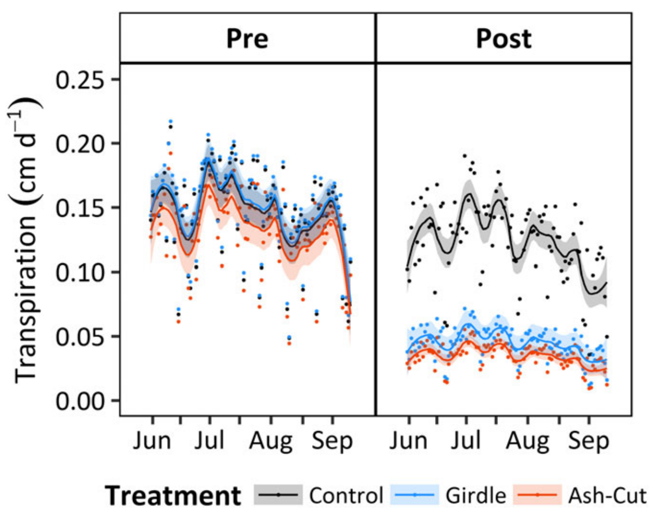

FIGURE 7 Mean daily transpiration estimates (points) for each treatment with Loess smoothed lines and shaded 95% confidence intervals during pre-treatment and post-treatment study periods. The four dominant canopy species were used to calculate daily transpiration for each site

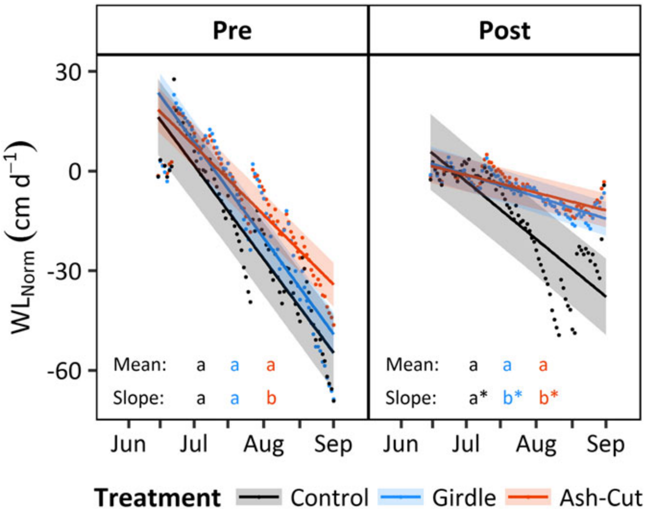

FIGURE 8 Mean daily normalized water levels (points; WL_{Norm}) and estimated rates of drawdown (lines) with 95% confidence intervals (shaded) during pre-treatment and post-treatment growing seasons (June 15–September 1). Different letters "a" and "b" denote significant differences among treatment means or slopes were detected. The "*" denotes significant differences for within-treatment comparisons (between study periods) were detected

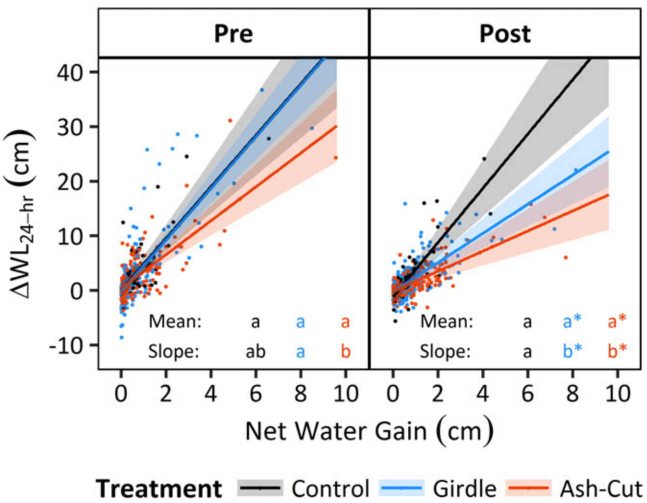

FIGURE 9 The response of 24-hr water level change ($\Delta WL_{24\text{-hr}}$) and shaded 95% confidence intervals predicted by the 24-hr net water gain for each treatment during pre- and post-treatment study periods. Different letters "a" and "b" denote significant differences among treatment means or slopes were detected. The "*" denotes significant differences for within-treatment comparisons (between study periods) were detected

post-treatment (Figure 9). The girdle and ash-cut slopes and means were also significantly lower during the post-treatment period when compared to the pre-treatment period, but significant within-treatment slope differences were not detected between periods for the controls. No significant slope differences were detected among treatments during pre-treatment. In the 24-hr change in water level LMM, the r_m^2 was 58%, the r_c^2 was 61%, and the residual standard error was 2.96 cm.

4 | DISCUSSION

4.1 | Source water contributions

Eight of the nine sites were well supplied by upland groundwater during spring and early summer during each of three study years (Figure 3 and 4). Gaining conditions were rarely observed in the one study site, which was the smallest (0.3 ha) and had the shallowest mean organic soil layer (15 cm) of all sites. The below average precipitation and snowfall received during 2012 in combination with the considerably earlier timing of snowmelt likely contributed to the less frequent occurrence of gaining groundwater conditions during late summer and fall. Conversely, the above average precipitation during 2013 and 2014 likely caused the greater occurrence of gaining conditions in those years (Figures 2 and 3). The importance of snowmelt-derived source water to headwater watersheds in the Great Lakes region has been well documented (Kolka et al., 2010; Stottlemyer & Toczydlowski, 1996; Stottlemyer & Toczydlowski, 2006), and the larger relative amounts of SWE during 2013 and 2014 likely had a strong influence on wetland water table levels in the snow-free periods for those years.

Mean-monthly *WS* and *GW* deuterium signatures tended to be very similar to one another and dissimilar to *TF* during the snow-free periods (Figure 4). The smaller standard errors and similar mean-annual *WS* and *GW* deuterium signatures (Figure 4; Table 2) indicate that black ash wetlands were well supplied by groundwater. Source water contribution estimates from isotope composition mixing models demonstrated that upland groundwater contributed at least 75% of the source water to the sites during the first three sampling months for both 2012 and 2013 (Figure 5). The consistent dominance of upland groundwater contributions to the wetland-soil water mixture during the spring and early summer was associated with the prevalence of gaining conditions that existed among sites (Figure 3). Whereas the greater variability of source water contribution estimates during late summer and fall in 2012 was likely associated with the smaller occurrence of gaining conditions among sites (Figure 3). Furthermore, the greater occurrence of gaining conditions during 2013 (Figure 3) is corroborated by the lower variability of monthly source water contribution estimates during 2013 when compared to 2012 (Figure 5). Overall, strong agreement existed between the hydraulic gradient, isotopic mixing model, and mean deuterium signature results, and indicate depressional black ash wetlands were well supplied by groundwater.

The lower density fibric peat layer detected near the surface of the sites exhibited larger hydraulic conductivity values (≈1×10^{-3} cm s^{-1}) when compared to the denser sapric peat detected below a depth of 30 cm (Table 1) and are commonly considered the hydrologically active layers of peatlands (Guertin, Barten, & Brooks, 1987). The combined presence of clay lenses at the organic-mineral soil interface and the low hydraulic conductivity values (<1×10^{-5} cm s^{-1}) exhibited by well-decomposed sapric peat found in deeper Histosol layers (Boelter, 1969) likely confine and significantly retard vertical movement of water within the sites (Ferone & Devito, 2004). Because vertical groundwater seepage is highly restricted or non-existent, surface water run off, lateral groundwater flow, and evapotranspiration are the three major hydrologic sinks in depressional black ash wetlands.

Other studies have shown that seasonal connections between groundwater and similar depressional wetlands provide major inputs of water through shallow lateral seepage pathways (Devito et al., 1996; Hill & Devito, 1997). Because of the strength of the connectivity between our sites and groundwater flow systems, this lateral seepage at shallow depths provides a substantial input of water to our sites. As a result, the water table positions in depressional black ash wetlands will likely be less sensitive to drawdown by evapotranspiration during wetter years due to the more persistent connection with groundwater flow systems.

Water table fluctuations within depressional black ash wetlands were less variable during wetter years due to the more consistent wetland–upland groundwater linkage, as was observed during the considerably wetter 2013 and 2014 water years (Figures 2 and 6). Water table elevations in wetlands that are perennially connected to regional groundwater systems tend to have little variability (Sebestyen et al., 2011), whereas water table fluctuations in wetlands that are primarily connected to local groundwater and perched systems tend to be larger and highly variable (Devito et al., 1996; Winter, 1999). A range of wetland–upland water table connection regimes have been shown to exist (Devito et al., 1996; Siegel, 1987), where transient or local groundwater flow systems can intermix with intermediate and regional scale groundwater flow systems (Tóth, 1963; Verry & Boelter, 1978; Winter, 1998). Because black ash wetlands were persistently connected to groundwater flow systems during 2013 and 2014 and maintained connection with groundwater flow systems during spring and early summer of 2012, it is likely that intermediate scale groundwater is a major contributor to these sites. The relative influence of groundwater on wetland water table variability will consequently be influenced by annual precipitation patterns where stronger connections to groundwater flow systems and smaller water table variability are expected during wetter years.

The depressional position of our black ash wetland sites on the glacially deposited landscape and our results, including the persistent gaining conditions (Figure 3), the large groundwater contribution estimates from the isotopic mixing models (Figure 5), and the similarity between the *WS* and *GW* deuterium signatures (Figure 4; Table 2) indicate that these wetlands are well connected to intermediate scale groundwater flow systems that intermix with local scale groundwater flow systems (Tóth, 1963; Winter, 1999). These results are further corroborated by the fact (a) the majority of sites are located within 30–60 m of glacial till or coarse-grained stratified sediments deposits (Soller et al., 2012) that extended 6.1 km away, on average, from each site before encountering the shallower (0–15 m) patchy Quaternary sediment deposits that are less likely to contain intermediate scale groundwater flow systems, and (b) these parent materials have predominantly sandy textures and high rates of permeability within the surrounding Gogebic, Graveraet and Stutts soil series profiles (Soil Survey Staff). Specifically, the majority of our sites were connected to intermediate groundwater flow systems during spring, early summer, and fall 2012, and for the entire 2013 and 2014 snow-free periods. During drier years, depressional black ash wetland water tables will be more variable due to lessened influence of intermediate groundwater flow systems, as was observed during the 2012 water year (Figure 2 and 6). Therefore, transpiration-induced water table drawdown will likely be exacerbated, and precipitation-driven groundwater recharge and water table variability will be larger during drier years due to the reduced influence of the wetland–upland groundwater linkage.

4.2 | Hydrologic responses

Drawdown rates were significantly lower during the growing season in girdle and ash-cut treatments when compared to the controls (Figure 7), which ultimately generated the significantly higher mean normalized water levels observed in August in the girdle and ash-cut sites during the post-treatment study period (Figure 8). The significantly greater rate of drawdown detected in the controls during the post-treatment period (Figure 7) can be attributed to the significantly lower amounts of transpiration in girdle and ash-cut sites (Figure 8). Disturbance-induced alteration to stand transpiration ultimately produced the significantly higher August wetland water levels in the ash-cut and girdle sites as compared to the controls (Figure 7).

The persistent connection to intermediate groundwater flow systems observed in depressional black ash wetlands (Figures 3–5) likely influenced the duration and magnitude of detectable water level responses to the girdle and ash-cut treatments. Similar to the reduced water table variability observed in northern Minnesota black spruce peatlands connected to regional groundwater flow systems (Sebestyen et al., 2011), the persistent connection to intermediate groundwater flow systems observed in depressional black ash wetlands (Figures 3–5) likely diminished the observed treatment response despite the significantly lower transpiration rates detected in the treated sites when compared to the controls (Figure 7). The intensified connection with intermediate groundwater flow systems during the wetter post-treatment study period (Figures 3–5) likely buffered the relative magnitude of transpiration-induced drawdown during the growing season and ultimately reduced the duration and magnitude of detectable treatment-induced drawdown responses in depressional black ash wetlands.

Specifically, wetter weather and decreased transpiration during late August and early September of the post-treatment years (Figures 2 and 7) caused wetland water tables to rise in all study sites in September (Figures 6 and 8). Post-treatment water table fluctuations were also less variable in September in all study sites, and mirrored springtime water table conditions. We attribute the higher and less variable water tables to a seasonal reconnection with the intermediate groundwater flow systems. Conversely, the sensitivity of wetland water tables to evapotranspiration was greater during drier periods when the connection to groundwater flow systems was weaker and resulted in greater proportion of locally derived water in the wetland water samples (Figures 4 and 5). These results suggest that during drier periods there was either a weaker connection with intermediate groundwater flow systems or the connection with groundwater flow systems was dominated by local scale source water (Winter, 1999). Therefore, the duration of detectable transpiration-induced water table drawdown in depressional black ash wetlands (Figures 7 and 8) was limited to drier periods when the connection to groundwater was weaker or dominated by local flow systems. Ultimately, water tables in girdled and ash-cut sites were significantly higher than controls by the end of the growing season, but treatment-induced alteration to wetland water tables quickly disappeared once connection with intermediate groundwater flow systems was re-established.

These results support the findings of an earlier study in Minnesota where water tables within black ash wetland treatment blocks were significantly higher than those in unmanipulated control blocks (Slesak et al., 2014), but also highlights the unique responses to treatment associated with the hydrogeological setting of depressional wetlands. In Slesak et al. (2014) study, 1.6 ha treatment blocks were used to simulate an ash borer disturbance and potential timber management scenarios within two 100–150 ha mineral soil black ash wetland complexes. Differences between the timing, duration, and magnitude of responses detected in our depressional study sites and the large mineral soil wetlands found in northern Minnesota can likely be attributed to the considerably thicker organic soil deposits (mean depth = 1.4 m), smaller size (0.25–1.25 ha), and distinct hydrogeological position on the landscape found in our study sites. For these reasons, resource managers should consider soil type and hydrogeological setting of black ash before implementing forest management practices in black ash wetlands.

The physical properties of organic soils likely contributed to the significantly smaller water table responses to episodic precipitation events observed in treated sites when compared to controls (Figure 9). Soil bulk density measurements were considerably larger, and hydraulic conductivity estimates were considerably lower for soils located between 30 and 50 cm beneath the surface when compared to shallower soil layers in depressional black ash wetlands (Table 1) and these data suggest that the magnitude of water table response to episodic water inputs in deeper soil layers would be larger when compared to water table responses in shallow layers. Therefore, the significantly lower water tables observed in control sites during the post-treatment growing seasons occurred where larger soil bulk density and smaller hydraulic conductivity values were observed, and together, these conditions generated the larger episodic water table response to precipitation magnitudes in control sites when compared to treated sites (Figure 9).

The highest water table positions occurred during the spring, early summer, and fall, when black ash wetlands were well supplied by intermediate groundwater flow systems and water tables were less sensitive to treatment related impacts. In contrast, mean normalized water levels were significantly higher in treated sites during the latter portion of the post-treatment growing seasons when connections to ground water flow systems were weaker or derived from local sources. Therefore, and as highlighted by Costilow et al. (2017), we do not expect woody regeneration to be adversely affected by the decreased drawdown rates and elevated water table positions, and new vegetative growth should eventually compensate for the sudden losses in transpiration detected immediately following EAB disturbance. Continued study of black ash wetlands is needed to better understand effects of EAB on wetland water tables during water years with average or below-average precipitation.

5 | CONCLUSIONS

Nine forested wetlands with black ash dominant canopies in northern Michigan were monitored for three years. Treatments were applied to simulate EAB infestations in six of the sites after the first year. A girdle treatment was used to simulate an active EAB infestation, and an ash-cut treatment was used to simulate post-infestation ecosystem conditions. All sites were located in landscape depressions within headwater watersheds. Water tables, precipitation, isotopic signatures, and canopy transpiration were monitored to determine wetland–upland hydrologic connectivity and hydrologic responses to treatments.

Our data demonstrate that depressional black ash wetlands were persistently connected to and well supplied by groundwater. Furthermore, isotopic source water analysis, water level variability, and the physical properties of the surrounding surficial sedimentary deposits indicate our sites were recharged by intermediate scale groundwater flow systems. This strong wetland–upland groundwater connection was most pronounced during spring, early summer, and fall and was especially pronounced during water years with above average precipitation. Our data also demonstrate that the magnitude and variability of wetland water levels will be less influenced by episodic precipitation and transpiration-induced water exchange mechanisms when strong wetland–upland groundwater linkages exist due to the controlling influence of intermediate groundwater flow systems and the physical properties of peat.

Our results indicate that EAB infestations have the potential to significantly affect black ash wetland hydrology, though these changes are likely to be moderated by connections between depressional black ash wetlands and groundwater flow systems. The rate of drawdown was significantly lower in girdled and ash-cut treatment sites when compared to unmanipulated control sites due to the decreased amount of transpiration caused by a simulated EAB infestation. Consequently, mean water table positions were significantly higher in treated sites, but the duration of significant treatment-induced water level differences was limited to the end of the growing season when the relative influence of groundwater connection was reduced. As a result, the above average precipitation received during the post-treatment water years effectively diminished the magnitude of treatment-induced alteration to black ash wetland water tables.

ACKNOWLEDGEMENTS

We appreciate the support of the USDA Forest Service, in particular Mark Fedora and the Ottawa National Forest. Funding for this project was provided by the USDA Forest Service, US-EPA Great Lakes Restoration Initiative (DW-12-92429101-0), Michigan Tech's Center for Water and Society, and Michigan Tech's Ecosystem Science Center. Special thanks to Stephen Sebestyen, Alex Mayer, Robert Froese, Jarrod Nelson, and Nicholas Schreiner for their technical and logistical support. We appreciate the constructive critique from the reviewers, especially Kevin Devito, which led to great improvements in the manuscript.

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How to cite this article: Van Grinsven MJ, Shannon JP, Davis JC, et al. Source water contributions and hydrologic responses to simulated emerald ash borer infestations in depressional black ash wetlands. *Ecohydrology*. 2017;e1862. https://doi.org/10.1002/eco.1862