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ARTICLE

Water table response to harvesting and simulated emerald ash borer mortality in black ash wetlands in Minnesota, USA

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Abstract: Black ash wetlands are seriously threatened because of the invasive emerald ash borer (EAB). Wetland hydrology is likely to be modified following ash mortality, but the magnitude of hydrological impact following loss via EAB and alternative mitigation harvests is not clear. Our objective was to assess the water table response to simulated EAB and harvesting to determine if management actions will be needed to maintain ecosystem functions following EAB infestation. We applied four replicated treatments to 1.6 ha plots as follows: (1) control, (2) girdling of all black ash trees to simulate loss via EAB mortality, (3) group selection harvests (20% of stand in 0.04 ha gaps), and (4) clear-cut harvest. Water table (WT) elevations were monitored for 1 year pre-treatment and two years post-treatment. Clear-cutting delayed WT drawdown in both years of the study, and the WT was significantly higher than the control treatment, predominantly when WT depth was below 30 cm. The effect of the group selection treatment on WT response was muted compared to clear-cutting and also limited to periods when the WT depth was below 30 cm. These responses were attributed to establishment of shallow-rooted vegetation in cut areas, which would have limited influence on WT dynamics as depth increased. There was little effect of girdling on the WT in the first year post-treatment, but effects on the WT were very similar to clear-cutting in the second year and more pronounced when the WT was within 30 cm of the soil surface. These effects were attributed to reduced transpiration coupled with the presence of a partial canopy following girdling, which would have reduced vegetation establishment and evaporation compared to clear-cutting. Given the large influence of WT depth on vegetation dynamics and associated feedbacks to altered hydrology, these early results indicate a greater risk of ecosystem alteration following EAB mortality compared to clear-cut harvesting. Depending on local hydrologic regime, variation in precipitation patterns, and time for complete canopy loss, it may be necessary for managers to implement active mitigation strategies (e.g., group selection coupled with planting of alternative species) prior to EAB infestation to maintain ecosystem processes in these forested wetland systems.

Key words: water table rise, forested wetlands, invasive species, *Fraxinus nigra* Marshall, forest management, *Agrilus planipennis* Fairmaire.

Résumé : Les milieux humides occupés par le frêne noir (*Fraxinus nigra* Marshall) sont sérieusement menacés à cause de l'agrile du frêne (AF) (*Agrilus planipennis* Fairmaire). Les caractéristiques hydrologiques des milieux humides seront probablement modifiées à la suite de la mortalité du frêne mais l'ampleur de l'impact hydrologique causé par les pertes dues à l'AF et les coupes alternatives visant à réduire cet impact n'est pas claire. Notre objectif consistait à évaluer la réaction de la nappe phréatique (NP) à l'action simulée de l'AF et à la coupe pour déterminer si des interventions d'aménagement sont nécessaires pour maintenir les fonctions de l'écosystème à la suite d'une infestation de l'AF. Nous avons appliqué les quatre traitements suivants que nous avons répétés dans des parcelles de 1,6 ha : (1) témoin, (2) annélation de tous les frênes noirs pour simuler la perte via la mortalité causée par l'AF, (3) coupes de jardinage par groupe (20 % du peuplement par trouées de 0,04 ha) et (4) coupe à blanc. Le niveau de la NP a été suivi pendant un an avant et deux ans après les traitements. La coupe à blanc a retardé le rabattement de la NP durant les deux années de l'étude et elle était significativement plus haute que dans le traitement témoin surtout lorsque la profondeur de la NP était inférieure à 30 cm. L'effet de la coupe de jardinage par groupe sur la réaction de la NP était faible comparativement à l'effet de la coupe à blanc et l'effet était limité aux périodes durant lesquelles la profondeur de la NP était inférieure à 30 cm. Ces réactions ont été attribuées à l'établissement de végétation superficiellement enracinée dans les zones coupées, laquelle aurait eu une influence limitée sur la dynamique de la NP à mesure que la profondeur augmentait. L'annélation a eu peu d'effet sur la NP durant la première année après le traitement mais les effets étaient très semblables à ceux de la coupe à blanc durant la deuxième année et plus prononcés lorsque la NP était à moins de 30 cm de la surface. Ces effets ont été attribués à la réduction de la transpiration couplée à la présence d'un couvert forestier partiel à la suite de l'annélation, laquelle aurait réduit l'établissement de la régénération et l'évaporation comparativement à la coupe à blanc. Étant donné la forte influence de la profondeur de la NP sur la dynamique de la végétation et les rétroactions associées à la modification des caractéristiques hydrologiques, ces résultats préliminaires indiquent qu'il y a un plus grand risque que l'écosystème soit modifié à la suite de la mortalité due à l'AF qu'après une coupe à blanc. Selon le régime hydrologique local, la variation dans le patron des précipitations et le temps nécessaire à la disparition complète du couvert forestier, il est possible que les

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aménagistes doivent mettre en œuvre des stratégies proactives d'atténuation (p. ex. coupe de jardinage par groupe couplée à la plantation d'autres espèces) avant une infestation afin de maintenir les processus de l'écosystème dans ces milieux humides boisés. [Traduit par la Rédaction]

Mots-clés : élévation de la nappe phréatique, milieux humides boisés, espèce invasive, *Fraxinus nigra* Marshall, aménagement forestier, *Agrilus planipennis* Fairmaire.

Introduction

The ash genus (*Fraxinus*) in North America is under threat of extirpation due to the introduction of the invasive emerald ash borer (EAB; *Agrilus planipennis* Fairmaire), which has decimated ash populations in parts of the Great Lakes region. Larvae of the insect feed on phloem and cambium of the host tree effectively girdling it and resulting in almost 100% mortality within 3–5 years after infestation (Gandhi et al. 2008). A number of studies have documented extensive economic and ecological damage from EAB (Kovacs et al. 2010; Gandhi and Herms 2010), but relatively little is known about whole ecosystem response to EAB mortality, particularly in stands where ash is a dominant component with few alternative replacement species. In Minnesota and other areas of the Great Lakes region, there is particular concern that EAB will have large impacts on the ecological functions of black ash (*Fraxinus nigra* Marshall,) wetlands, which commonly occur in almost pure stands on poorly drained sites (MN DNR 2003). These forested wetland systems cover approximately 404 000 ha located primarily in the north central to north eastern portion of the state.

Because black ash occupies a unique wet niche where few other tree species grow, it serves as a foundation tree species that strongly regulates ecosystem processes and community structure (sensu Ellison et al. 2005). Notably, it is likely that the species has large control on site hydrology via its influence on evapotranspiration (ET), which in turn has strong control on site water balance. Anecdotal evidence and past reports (MN DNR 2008; Erdmann et al. 1987) suggest that removal of black ash via clear-cutting can reduce ET, resulting in a rise in the water table (WT) and subsequent conversion to a brush-dominated wetland, especially on very wet sites. However, it is unclear if EAB-induced mortality will result in similar outcomes, as there are a number of interacting drivers that will control the post-ash vegetation dynamics, including more gradual mortality, understory vegetation dynamics, and other environmental stressors (Palik et al. 2011).

Site and watershed scale impacts of EAB to hydrology are likely, but the magnitude and duration of these impacts are unclear as relatively little research has been conducted in this wetland type. Hydroperiod is known to vary greatly amongst wetland types and landscape positions, where wetlands fed by consistent groundwater sources have very little water level variability, while alluvial bottomland floodplain forests can have variations of several meters. Generally, forested wetlands in northern North America that occur on mineral soils and are dominated by precipitation inputs have intermediate water level variability characterized by frequent water level fluctuation and a large drop during the peak of ET that occurs in mid to late summer (Mitcsh and Gosselink 2007). If the duration of time when water tables are near (<30 cm) the soil surface is extended following EAB mortality or other disturbances that alter ET, such as forest harvest, it is likely to cause a shift in the vegetation community from facultative wetland species to more obligate wetland species (Lichvar and Kartesz 2009), or favor establishment of herbaceous vegetation rather than trees (Toner and Keddy 1997). Such shifts may cause a positive feedback to hydrologic change and ecosystem conversion (Ridolfi et al. 2006, Asbjornsen et al. 2011).

Land managers are currently considering a variety of options to address the EAB threat, most of which involve conversion of stand composition to suitable species other than ash through various silvicultural techniques including selection harvesting, shelterwood systems, and clear-cutting (Gupta and Miedtke 2011). Although these efforts are intended to preemptively mitigate the impact of EAB in black ash wetlands, it is possible that some strategies may cause more harm than good. For example, Erdmann et al. (1987) reported that clear-cutting black ash on wet sites resulted in poor regeneration and eventual conversion to brush and grass, but successful stand establishment when sufficient advance regeneration was present. Similarly, the water level rise that occurs following forest harvest in boreal lowlands of eastern Canada is one of the primary mechanisms contributing to paludification on those sites (Lavoie et al. 2005). In that region, Pothier et al. (2003) found that the WT can remain elevated for at least 5 years following harvesting, with the magnitude of rise dependent on the amount of basal area removed. Similar results have been observed in the southeastern United States (Xu et al. 2002; Bliss and Comerford 2002), but the relevance of these results and others to black ash wetlands is not clear as some studies have shown a limited effect of cutting because of an increase in evaporation when the WT is near the surface (Verry 1986; Dube et al. 1995). There is an urgent need to quantify the hydrologic and vegetative response of black ash wetlands to forest harvesting and EAB mortality, so that managers can make informed decisions to maintain ecosystem functions of this extensive wetland type.

Here we report on the WT response of black ash wetlands to replicated experimental treatments that either mimic EAB mortality or represent alternative management actions to preemptively mitigate EAB impacts via manipulation of stand composition. The overall project objectives are to assess EAB impacts on plant communities, regeneration, and hydrology in northern black ash wetlands, and provide information for development of management recommendations aimed at mitigating the impacts of EAB by maintaining those wetlands in a forested condition. Specific objectives of this paper are to (1) report on the baseline hydrology of black ash wetlands prior to treatment application, and (2) assess the effect of experimental treatments on WT response for two years post-treatment. The underlying emphasis of this report focuses on localized impacts to WT dynamics caused by EAB mortality.

Methods

Site characteristics

The study area is located in north-central Minnesota USA on the Chippewa National Forest, just North of Lake Winnibigosh in Itasca County (N47°31′57″, W94°12′39″) at an approximate elevation of 400 m. The climate is continental, with mean annual precipitation of 700 mm and a mean growing season (May–Oct.) temperature of 14.3 °C. Annual precipitation consists of about 1/3 snow and 2/3 rainfall. Stands in the study area were classified as either wet (type WFn55) or very wet (type WFn64) ash swamps occurring on mineral soils, based on Minnesota's native plant community classification (MN DNR 2003). Soils are predominantly poorly to very poorly drained, with either a loam to sand texture formed from glacio-fluvial or lacustrine parent materials or a clay to silty clay texture formed from glacio-lacustrine material overlain by approximately 30 cm of muck (Soil Survey Staff, USDA-NRCS). Stand composition consists almost entirely of black ash (75%–100% of pre-harvest basal area), with minor components of

American elm (*Ulmus americana*), quaking aspen (*Populus tremuloides*), American basswood (*Tilia Americana*), and bur oak (*Quercus macrocarpa*). Pre-harvest basal areas ranged from 23.0–39.2 $m^2 \cdot ha^{-1}$ and densities ranged from 563–917 $stems \cdot ha^{-1}$. Analysis of dendroecological data collected from some of the stands and nearby areas indicated that they were strongly uneven-aged with canopy tree ages ranging from 130–232 years and radial increment patterns suggestive of gap-scale disturbance processes (A.W. D'Amato, unpublished data).

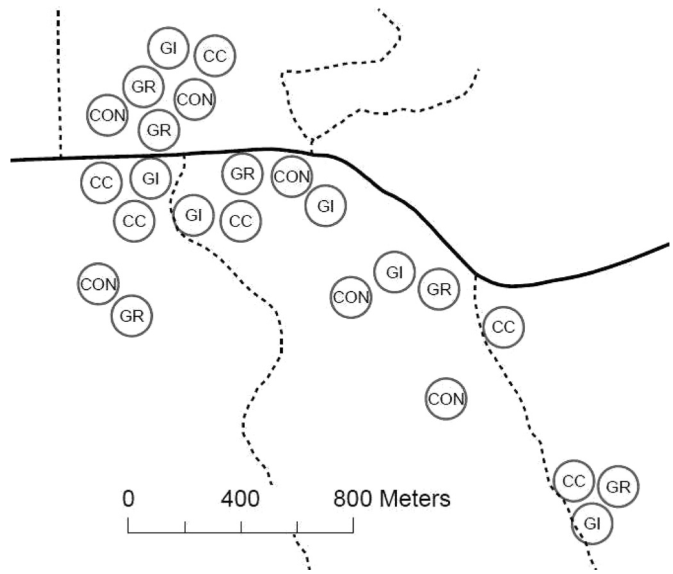
The sites are located within a complex glacial landscape that is flat to gently rolling, with the black ash wetlands found in flats in the lowest landscape positions that grade into aspen or pine-dominated upland forests. Most of the black ash wetlands are underlain by lacustrine clay at a depth of 10–150 cm, which acts as a hydrologic restraining layer. The lacustrine clay soils are interspersed with sandier upland soils that serve as groundwater recharge zones for local watersheds. Potential ET (PET) has been estimated to be about 600–650 mm per year (Sebestyen et al. 2011), while annual runoff or streamflow is approximately 160 mm per year in this region (Lorenz et al. 2009), with peak stream flows normally occurring in May to June. Hydrologic monitoring and water geochemistry investigations during the study period indicated that most of the water supply to the black ash wetlands comes from snowmelt in the spring and periodic inputs via precipitation throughout the growing season (Lenhart et al. 2012).

Experimental design and measures

Four treatments were applied in a completely randomized block design with six replications (Fig. 1). Experimental units were circular 1.6 ha plots, and block assignment was based on plot proximity, pre-treatment anecdotal impressions of hydrologic regime (e.g., wet vs. relatively drier), and native plant communities. The four treatments were (1) control, (2) girdling of all black ash trees down to 10 cm diameter at 1.3 m height, (3) group selection harvests (20% of stand in 0.04 ha gaps), and (4) clear-cut harvest (complete removal of all stems). The girdling treatment is intended to mimic the effect of EAB mortality, and the two forest harvesting treatments represent alternative management strategies that could be used to modify stand composition through natural regeneration or planting of alternative tree species to maintain a forest ecosystem following EAB infestation. All treatments were applied in the winter of 2011–2012 under frozen ground conditions, and trees were re-girdled in the winter of 2012–2013 to ensure 100% mortality in that treatment. Girdling treatments were applied manually with the use of draw knives, and the harvesting treatments were conducted with cut-to-length mechanized harvest systems. Examination of crown conditions in the girdling treatments indicated significantly declining tree vigor in these areas after two years. Non-merchantable tops and limbs were retained at the stump.

Groundwater monitoring wells constructed of 2 inch slotted PVC pipe were installed in the approximate center of each experimental replication in the fall of 2010. Wells were installed with the use of 2 inch hand augers to an approximate depth of 1.5 meters or until a confining layer was reached. Water levels were monitored in each well from May to October of each measurement year with the use of pressure transducers (Levelogger Gold Model 3001, Solinst Canada Ltd., Ontario, Canada) that were suspended ~10 cm from well bottom. Data were logged at 5 min (in 2011) or 10 min (in 2012 and 2013) intervals and corrected for atmospheric pressure, which was recorded at the same time interval by a logger (Barologger, Solinst Canada Ltd., Ontario, Canada) suspended in the headspace of one of the monitoring wells. Ground and pressure transducer elevation were determined relative to the top of each well, and absolute well elevation was measured with a survey grade GPS unit (Trimble Navigation Ltd., Sunnyvale, California, USA). A HOBO U30-NRC weather station (Onset Computer Corp., Bourne, Massachusetts, USA) was deployed at the project site during the growing season of each year to record climate data including air temperature, relative humidity, and precipitation.

Fig. 1. Site map showing the locations of treatment replications. Improved forest road is shown as a solid line, temporary haul roads are shown as dashed lines. CC = clear-cut, CON = control, GI = girdle, and GR = group selection.



Data analysis

Water table elevations were expressed relative to the ground surface for all analyses. Values from each well were averaged by day and week during the growing season of each year (May–Sept.). Treatment effects on pre- and post-treatment mean weekly relative WT depth were assessed with a mixed model analysis of variance (ANOVA) with repeated measures analysis using an auto regressive level 1 covariance matrix. We conducted a similar analysis on the post-harvest WT response of the three manipulative treatments relative to the control treatments, using the pre-treatment ratio of manipulative to control treatments as a covariate to account for inherent differences in hydrology among the experimental units. In both models, the block factor was modeled as a random effect, and treatment, week, and their interaction were modeled as fixed effects. When *F* statistics indicated significant interaction between the fixed effects, the SLICE option in SAS was used to determine weeks where the means differed among treatments followed by means comparison with the Tukey–Kramer method.

We also determined the number of growing-season days when water was ponded above the ground surface, the number of days water was within 30 cm of the surface, and the number of days when water was below 30 cm of the surface in both pre-treatment and post-treatment years to characterize growing conditions in each replication. We used a 30 cm depth to distinguish between growing condition categories given that it is commonly used to differentiate between wetlands and uplands, and most roots occurred within 30 cm of the surface at these sites (R. Slesak, personal observation). Using these data, we calculated the change in number of days within each water level category between years by subtracting pre-treatment values from those post-treatment (i.e., positive values indicate increase in number of days within a water level category following treatment). Treatment effects on the change in days for each category were assessed with a mixed effects model, where random and fixed effects were modeled as described above and the pre-treatment value was used as a covariate. Visual examination of residuals for all models indicated that

Fig. 2. Total monthly precipitation measured during the growing season of each year at the project site compared to the 50 year mean (estimated with the PRISM model, PRISM Climate Group, Oregon State University, Corvallis, Oregon, USA).

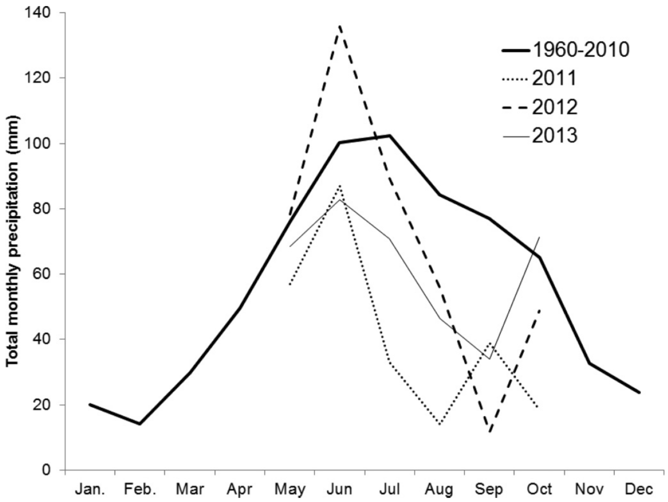
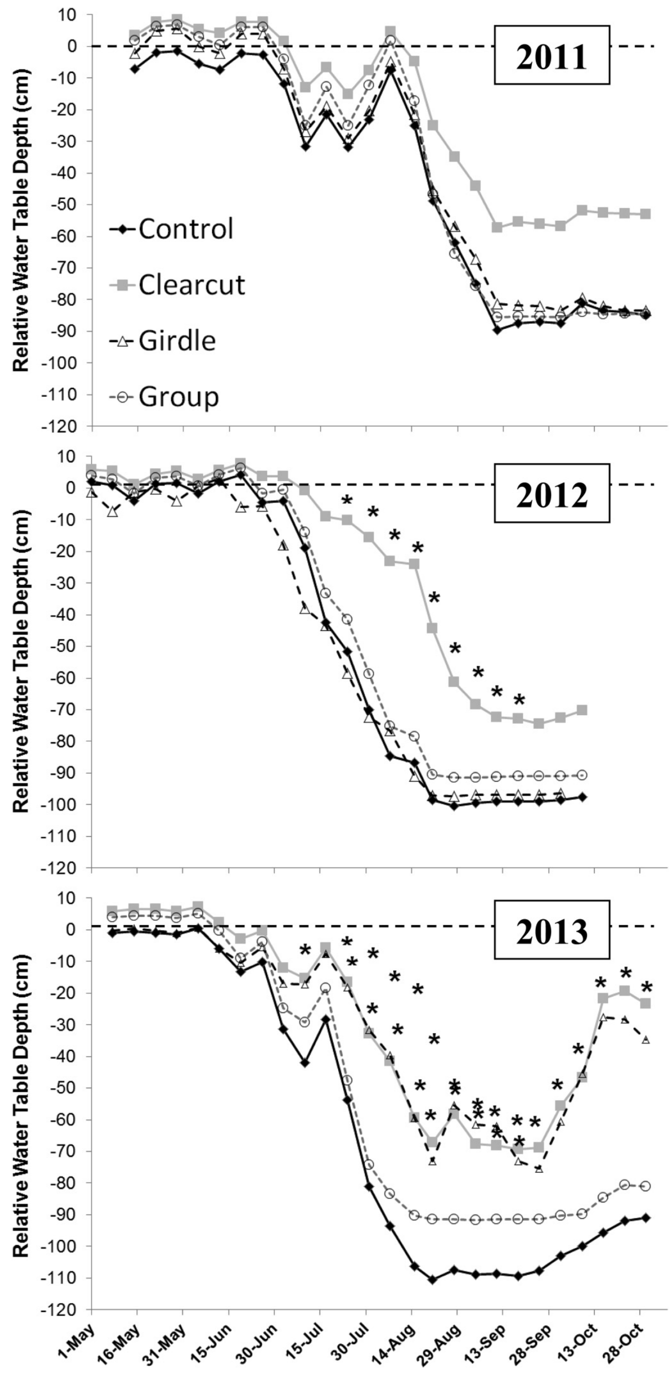


Fig. 3. Weekly mean (n = 6) water table depth relative to ground surface by treatment. Pre-treatment (2011) and post-treatment (2012 and 2013) growing seasons analyzed separately; weeks with a * indicate significant difference among treatments. There were no differences among treatments prior to treatment application (p > 0.2).



assumptions of normality and homogeneity were valid. An α level of 0.05 was used in all statistical tests. All analyses were performed in SAS V9.3 (SAS Institute, Cary, North Carolina, USA).

Results

Total growing season precipitation was 50%, 20%, and 30% lower in 2011, 2012, and 2013, respectively, compared to the 50 year mean estimated with the PRISM model (PRISM Climate Group, Corvallis, Oregon, USA). The distribution of precipitation was similar among years of the study period, with monthly precipitation in the first half of the growing season being similar to the 50 year mean, followed by lower precipitation in the latter half of the growing season (Fig. 2).

Regardless of treatment, WT elevations demonstrated a similar seasonal pattern in all years (Fig. 3). Water tables were at or near the ground surface until approximately late June, followed by a general drawdown (with periodic recharge in 2011 and 2013) during July and August, and achieved a stable minimum depth in the early fall. Minimum WT depths were lower and occurred approximately one month earlier in both 2012 and 2013, compared to 2011. Prior to treatment application, the mean number of days where the WT was above the surface, within 30 cm of the surface, and below 30 cm of the surface was 53.0 (SE = 5.8), 39.3 (SE = 3.2), and 45.7 (SE = 6.1), respectively, for the period of May 16 to September 30.

Prior to treatment, there were no significant differences among treatments in weekly WT depths (Table 1), but the mean depth to the WT in the clear-cut treatment was consistently lower compared to the other treatments (Fig. 3). Following treatment application, there was a significant effect of treatment on mean relative WT depth that varied by week in both 2012 and 2013 (Table 1, Fig. 3). In 2012, from mid-July to mid-September, depth to WT was significantly lower in the clear-cut treatment compared to all other treatments. There were no significant differences among any of the other treatments, although depth to WT in the group selection treatment was consistently lower than either the control or girdle treatment for much of the growing season. When expressed relative to the control treatment and adjusting for pre-treatment differences, changes in relative WT depth in the clear-cut treatment, compared to the other manipulative treatments, were only significant from early July to mid-August (Table 2, Fig. 4). There was no difference between the group selection and girdle treatments when expressed relative to the control, but both treatments had a significantly lower depth to WT than the control in some months of the latter part of the growing season.

In 2013, mean WT depth in the girdling treatment was similar to that of the clear-cut treatment, with both treatments having a significantly lower mean depth to the WT than either the control or group selection treatments for much of the growing season (Fig. 3). The difference in WT depth between the girdling and clear-cut treatment and the control treatment increased throughout the growing season (Fig. 4), with late-season increases due to a

Table 1. *F*-statistics and associated *P*-values for fixed treatment effects on water table heights during pre-treatment and post-treatment growing seasons (May–Sept.).

	Pre-treatment (2011)		Post-treatment (2012)		Post-treatment (2013)	
Effect	*F*-stat	*P*-value	*F*-stat	*P*-value	*F*-stat	*P*-value
Treatment	1.76	0.154	5.18	0.002	7.59	<0.001
Week	53.35	<0.001	64.28	<0.001	50.47	<0.001
Treat.*week	0.54	0.999	2.32	<0.001	3.58	<0.001

Table 2. *F*-statistics and associated *P*-values for fixed effects on the change in water table height following treatment during the growing season of 2012 and 2013 when treatment response was expressed relative to control.

	2012		2013	
Effect	*F*-stat	*P*-value	*F*-stat	*P*-value
Covariate	5.11	0.025	6.29	0.013
Treatment	4.13	0.049	3.43	0.074
Week	4.11	<0.001	6.93	<0.001
Treat.*week	1.80	0.003	2.60	<0.001

Note: Pre-treatment weekly relationship between treatments and controls was used as a covariate.

more pronounced rise in the WT in the girdling and clear-cut treatments following higher than average fall precipitation (Fig. 2). The group selection treatment also had significantly lower mean depth to the WT depth than the control treatment in 2013, with the differences largely limited to the latter part of the growing season, similar to 2012 (Table 2, Fig. 4).

There was no effect of treatment on the change in the number of days during the period May 16 to September 30, where the water was above the ground surface in either year ($p > 0.76$). In 2012, the clear-cut treatment had a significantly smaller increase in the number of days when the WT was below 30 cm of the ground surface compared to the other treatments. In 2013, the girdling treatment had a significantly higher number of days when the WT was within 30 cm of the ground surface, and a significantly lower number of days when water was below 30 cm of the surface, compared to the other treatments (except the clear-cut treatment for >30 cm depth; Table 3). Regardless of treatment, the change in number of days when water was above the surface, within 30 cm of the surface, and below 30 cm of the surface generally indicated deeper WT in the post-treatment period compared to pre-treatment.

Discussion

Evapotranspiration is a major pathway of water loss in terrestrial ecosystems, comprising losses associated with transpiration, interception, and evaporation. In the black ash wetlands we studied, growing season water balance is largely controlled by precipitation and ET, as there is little evidence for deep regional groundwater inputs and no surface inflow from streams throughout the study area (Lenhart et al. 2012). The similar seasonal pattern in WT observed among years demonstrates this, where the WT was high early in the growing season following snowmelt and spring rain prior to leaf out, followed by a gradual drawdown as ET increased during the growing season (Fig. 2). Vegetation has a large control on the rate of WT drawdown via its influence on ET, and a reduction in the amount of leaf area associated with disturbance such as EAB mortality or harvesting should reduce the rate at which the WT drawdown occurs. In turn, effects on WT arising from lower ET may change vegetation communities and alter ecosystem properties if the ecosystem shifts to an alternative stable state (Ridolfi et al. 2006; Asbjornsen et al. 2011). In the case of black ash wetlands, there is particular concern that water table rise could favor more water-tolerant vegetation, causing a shift from forested wetland to open marsh with long periods of inundation. Ridolfi et al. (2006) point out that recovery following such a shift is unlikely unless external factors that control the WT (e.g., precipitation, succeeding vegetation, drainage) change and revert the WT to its pre-disturbance level.

Harvest treatment effects

The significant increase in WT height following clear-cutting is most likely due to a reduction in ET following harvest of the overstory. A number of studies have documented similar responses across a range of climate and site types, with increases in WT height similar to those observed here (Bliss and Comerford 2002; Xu et al. 2002; Marcotte et al. 2008; Pothier et al. 2003). Clear-cutting largely caused a delay in WT drawdown that was more pronounced in 2012 than 2013, and heights were higher than the control in both years, primarily when the WT was deeper than 30 cm. These responses likely reflect a shift in vegetation composition to more early successional herbaceous forms, which established later in the growing season immediately after harvesting (R. Slesak, personal observation), resulting in a more prolonged drawdown delay in that year (Fig. 3). Vegetation data collected across clear-cut treatments support this, as communities shifted towards dominance by graminoid species, particularly lake sedge (*Carex lacustris*) (A. D'Amato, unpublished data). Recently-established herbaceous vegetation is also generally shallow-rooted and has lower influence on WT dynamics with increasing depth (Shah et al. 2007), which would result in treatment effects being more pronounced when the WT is relatively deep. Increased evaporation in the clear-cuts due to greater wind speed and radiation inputs at the soil surface may have also contributed to the drawdown (Verry 1986), but this mechanism is probably minor given the timing and rapidity at which drawdown occurred. Despite the delay in drawdown following clear-cutting, most of the treatment effects occurred when the WT was deeper than 30 cm, which reduces the likelihood of impacts to vegetation dynamics arising from an altered hydrologic regime.

In contrast to the clear-cut treatment, the effect of the group selection treatment was more muted and limited to the latter portion of the growing season when the WT was higher than the control, probably because the overall reduction in ET would have been much lower with only 20% of the stand basal area removed. Pothier et al. (2003) found that WT rise following thinning of lowland spruce-fir forests was linearly related to the amount of basal area removed, which they partly attributed to a decreasing rate of rainfall interception with increasing removal. Here, reduced interception would have also occurred to some extent in the central portion of each gap, but the amount may have been small because the gap size was relatively small. The muted response relative to the clear-cut treatment may have also been due to a compensatory effect, where reduced transpiration and interception following tree removal were offset by increased losses from surface evaporation (Kelliher et al. 1995). However, since most differences occurred when the WT was relatively deep and less susceptible to evaporation losses (Gardner 1958), it seems more likely that the response was driven by a shift to shallow-rooted vegetation in the gaps, which would have had lower influence as the depth to WT increased. Given that treatment effects were limited to periods when the WT was deep, and there was no evidence of a delay in WT drawdown, it seems unlikely that

Fig. 4. Weekly mean (n = 6) change in water table depth relative to controls after treatment application in the winter of 2011–2012. Weekly pre-treatment difference between control and treatment plots was used as a covariate in the analysis; weeks with a * indicate significant difference among treatments. Letters under a given week when present indicate that a treatment value is significantly different from the control treatment (CC = clear-cut, GI = girdle, GR = group selection).

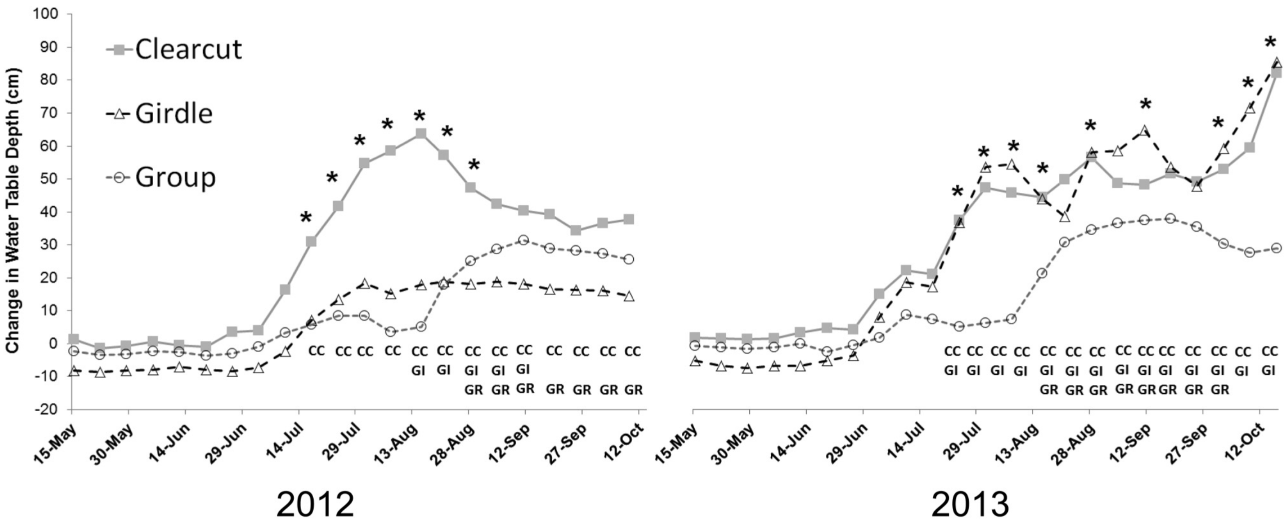


Table 3. Change in the number of growing-season days (May 16–Sept. 30) when the water table was above the surface, within 30 cm of the surface, or below 30 cm depth between pre-treatment (2011) and post-treatment measurement years by experimental treatment.

Treatment	Change in days above surface	Change in days within 30 cm of surface	Change in days below 30 cm of surface
2012			
Control	−12.8 (4.2) a	−6.1 (4.5) a	22.9 (3.5) a
Clearcut	−12.1 (4.1) a	4.4 (44) a	3.1 (3.5) b
Girdle	−20.4 (4.1) a	−0.9 (4.3) a	22.7 (3.4) a
Group selection	−16.1 (4.1) a	−9.8 (4.3) a	25.0 (3.4) a
2013			
Control	−19.7 (5.2) a	−9.0 (4.8) a	28.3 (6.2) a
Clearcut	−18.4 (5.3) a	1.5 (4.7) a	17.2 (6.2) ab
Girdle	−32.8 (5.2) a	24.7 (4.6) b	8.2 (5.9) b
Group selection	−22.3 (5.2) a	−6.7 (4.6) a	29.0 (5.9) a

Note: Mean values within a depth category and year with the same letter are not significantly different. Negative values indicate a reduction in number of days compared to pre-treatment, positive indicate an increase. Standard error in parenthesis (n = 6).

the group selection treatment will negatively shift plant community composition to more water-tolerant forms due to altered hydrology.

Simulated EAB mortality effects

The limited effect of girdling in the first year after treatment application was not surprising, as girdling initially allows xylem water transport to continue unabated and generally has limited effects on soil moisture for at least the first season after treatment (Hogberg et al. 2001; Zeller et al. 2008). Girdled trees did not show any obvious signs of stress from the treatment in 2012, but crown dieback was prevalent in 2013 and clearly visible in aerial photography (R. Slesak, personal observation). This response to simulated EAB had a notable effect on WT height in 2013, which was greater than the clear-cut treatment given pre-treatment WT patterns (i.e, clear-cut treatments had higher water tables than other treatments; Fig. 3). The greater response is also reflected in the change in the number of days when water was above and below 30 cm of the soil surface, as the results indicated much wetter surface conditions in the girdling treatment compared to the other treatments (Table 3). The greater response compared to the clear-cut treatment is likely because of different effects of girdling on the overstory and its influence on ET components. The reduction in leaf area in the second year, combined with reduced stomatal (Domec and Pruyn 2008) and xylem conductance (Zwieniecki et al. 2004) following girdling, would have reduced transpiration and contributed to the WT rise. At the same time, shade from the remaining foliage and tree trunks and branches would have inhibited establishment of understory vegetation and additional transpiration associated with it and also reduced disturbance-related increase in evaporation, because daily air temperature extremes were lower compared to the clear-cut (R. Slesak, unpublished data). Given the above, it seems likely that the risk of altered vegetation dynamics and a shift to an altered ecosystem state (e.g., open marsh condition) would be greatest following EAB mortality, given the increased duration when the WT is near the surface.

Caveats related to inter-annual variation in precipitation

Given the large influence that precipitation has on the growing season hydrology of these wetlands, it is worthwhile to consider precipitation patterns among years and any potential influence on the response. All years of the study period were relatively dry compared to the 50 year average, and the post-treatment period was drier than pre-treatment, as evidenced by the change in days when water was above or below the surface in 2012 and 2013, regardless of treatment (Table 3). Given these conditions, it is unclear if similar results would be observed in a more typical year with higher precipitation and a higher WT. For example, Pothier et al. (2003) attributed successful regeneration following clear-cutting of lowland conifers to dryer than typical conditions, as they noted a regeneration failure in nearby strip cuts during a wetter year. Conditions during the study period may have facilitated vegetation establishment following harvesting and reduced WT response, compared to years with greater precipitation. It is unlikely that the girdling treatment would have had a larger effect initially as transpiration would have been similar to the control treatment regardless the amount of precipitation, but it is possible that effects in 2013 could have been more pronounced if the WT had been near the surface for longer time periods. Greater precipitation in future years could also exacerbate impacts associated with an elevated WT following girdling, especially if they occur during the period prior to complete stand mortality when a partial canopy inhibits understory growth and increased evaporation.

Conclusions

These results indicate that both simulated EAB mortality and forest harvest can alter WT dynamics in black ash wetlands, generally causing a delay in WT drawdown and an increase in WT height. The mechanisms driving WT response appear to be largely associated with changes to the forest canopy and leaf area and its influence on vegetation establishment following treatment, but may have also been due to differences in evaporation among the treatments. Such short-term shifts in WT dynamics can have long-term effects on future compositional conditions of these sites, given their differential effects on facultative and obligate wetland species. Based on the timing and depth at which the treatments modified the WT for two years, risk of vegetation shifts and ecosystem alteration are greatest with simulated EAB mortality and clear-cutting and lowest with the group selection treatment. Risk of ecosystem alteration will vary depending on site hydrology (higher with shallow WTs), annual precipitation (higher with greater precipitation), and period of time necessary for establishment of deeper rooted vegetation (higher with prolonged crown dieback following EAB infestation or high herbaceous competition following clear-cutting). Since many of these factors will be difficult to predict, it may be necessary for forest managers to implement active mitigation strategies prior to EAB infestation to ensure maintenance of ecosystem processes in these forested wetland systems. Based on these early results, the group selection technique appears to be a suitable strategy to attempt modification of stand composition to other forest tree species prior to EAB infestation, while reducing risks of WT rise and ecosystem alteration.

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