SCIENTIFIC REPORT

NO SIGNIFICANT INCREASE IN TREE MORTALITY FOLLOWING CORING IN A TEMPERATE HARDWOOD FOREST

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ABSTRACT

The collection of tree-ring data from living trees is widespread and highly valuable in ecological and dendro-climatological research, yet there is concern that coring injures trees, potentially contributing to mortality. Unlike resinous conifers that can quickly compartmentalize wounds, less decay-resistant angiosperms may face more pronounced risk of injury from coring. To test if coring increases mortality rates in temperate hardwood trees, we leverage a unique dataset tracking the mortality of cored and uncored hardwood trees representing 19 species from 10 genera in a 26-ha plot in Virginia, USA. We compare mortality rates between 935 cored trees and 8605 uncored trees for seven years following coring. Annual mortality rates did not differ between cored trees (1.71% yr^{-1}; 95% CI 1.40 to 2.20) and uncored trees (1.85% yr^{-1}; 95% CI 1.70 to 2.28) across the full dataset, nor were there differences by genus or size class. These results indicate that the risk of hardwood mortality caused by increment coring is probably lower than previously assumed. However, these results cannot rule out the possibility that coring elevates hardwood mortality in other climates or when trees face multiple additional stressors that were not influential over the course of our study.

Keywords: coring, increment borer, wound, damage, mortality, temperate, hardwood, angiosperm.

INTRODUCTION

Tree rings provide valuable insight into forest age structures, stand dynamics, site productivity, and past climates. Many research questions require collecting data from living trees (Grissino-Mayer 2003; Speer 2010), despite the potential risk to tree injury or death. The most common method for collecting tree-ring data is by extracting small (*e.g.* 5-mm diameter) increment cores, which is clearly less impactful than cutting wedges or complete cross-sections of the tree bole. However, relatively few studies have evaluated the impacts of coring (Norton 1998; van Mantgem and Stephenson 2004; Wunder *et al.* 2011; Tsen *et al.* 2016), even as the collection and use of tree-ring data continues to expand.

It may take several years from the time of core extraction until new growth fully closes the cambial wound produced by the borer, depending on the species cored and the growth rate and other stresses affecting the individual tree (Hepting *et al.* 1949). Until then, the cambial wounds provide an entryway for fungi that accelerate wood decay. Trees can limit this damage to some degree by compartmentalizing the wounds, separating injured or decayed material from the uninfected wood (Shigo 1984). Compartmentalization may be effective at

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minimizing the impacts from coring on resinous conifers (van Mantgem and Stephenson 2004, Wunder *et al.* 2011a, 2013). Hardwoods, by contrast, tend to be less decay-resistant (Russell *et al.* 2014), and coring generally causes more immediate discoloration, staining, and fungal decay, potentially leading to major internal damage (Grissino-Mayer 2003). It is unclear, however, if such injuries lead to mortality, because relatively few medium- to long-term studies (*i.e.* 5–10 years) have documented the effect of coring on hardwood trees.

Here we leverage a unique dataset, in which the mortality of 935 cored trees and 8605 uncored trees was tracked for seven years in a large permanent plot within a temperate deciduous forest. The cored trees represent 19 species from 10 genera. This large sample size allows for a robust comparison of the impacts of increment coring on hardwood mortality.

METHODS

We conducted the study at a 26-ha large forest dynamics plot (Fig. 1) at the Smithsonian Conservation Biology Institute (SCBI) in northern Virginia, USA (38°53′36.6″N, 78°08′43.4″W; Bourg *et al.* 2013, Gonzalez-Akre *et al.* 2016). This plot is part of the Center for Tropical Forest Science-Forest Global Earth Observatory (CTFS-ForestGEO) network (Anderson-Teixeira *et al.* 2015). The climate is humid temperate, with a mean annual temperature of 12.9°C and mean annual precipitation of 1001 mm (Gonzalez-Akre *et al.* 2016). Elevation within the plot ranges from 273 to 338 m a.s.l.

The plot is a mature secondary mixed deciduous forest that developed with minimal direct human disturbances since agricultural abandonment in the mid-19th Century. The plot was established in 2008 following standardized protocols of the network (Condit 1998). Specifically, all woody stems ≥ 1 cm in diameter at breast height (dbh) were identified, mapped, tagged, and measured in dbh (Bourg *et al.* 2013). In order of decreasing basal area, the most abundant canopy tree genera are *Liriodendron*, *Quercus*, *Carya*, and *Fraxinus*. Canopy trees are primarily between 65 and 145 years old, though a few individuals up to 232 years old were also present.

Between August 2010 and April 2011, we collected increment cores at breast height (137 cm) from 997 randomly selected individuals from each genus with at least 30 trees > 10 cm dbh (Fig. 1; Bourg *et al.* 2013). We used a 5-mm diameter borer and did not plug the holes it created. All trees were ≥10 cm dbh at the time of coring, and the length of each core was just over half of tree dbh to ensure that we would reach the pith for future analyses of age structure and tree growth. We took a single core from each tree, except for 62 trees in which the initial core was too far off-center to accurately estimate the pith date. We extracted a second core from these trees at an angle oriented more directly toward the pith.

We tracked the mortality of all stems ≥ 10 cm dbh through censuses each year from 2013 to 2017 (Gonzalez-Akre *et al.* 2016). In total, 8605 of the trees ≥ 10 cm dbh in the initial census (2008) were not cored. Of the 997 cored trees, we retained 935 trees representing 10 genera. Samples from the one conifer in our plot (*Pinus strobus*) and large trees (>35 cm dbh) of *Fagus* and *Ulmus* were excluded from all analyses because of the small sample sizes available for comparing mortality rates.

We compared mortality rates between cored and uncored trees by genus and between two size classes: 10–35 cm and >35 cm dbh. The 35-cm dbh threshold was set to distinguish between large and small trees, whereby 35 cm dbh is the approximate size at which trees in this plot are more likely to be dominant or codominant than suppressed or intermediate, following the classification described in (Gonzalez-Akre *et al.* 2016). This separation by size class enables us to assess whether smaller trees are more severely impacted than the larger trees that dominate the forest canopy.

To calculate mortality rates, we repeatedly sampled random subsets of the uncored trees in which each genus–size class combination had twice the number of cored trees (n = 1870). Doubling the sample size relative to cored trees strengthened estimates of mortality rates while maintaining reasonably high independence among multiple samples drawn from the uncored population. Using R version 3.4.3 (R Core Team 2017), we repeated the random selection 1000 times and averaged the number of tree deaths across simulations. For each genus–size class combination we report the mean

Table 1. Mortality rates of 10 genera of hardwood trees in two size classes in Virginia, United States (See Eqn. 1 for the definitions of variables N_0, N_t, and m).

| Genus | Small trees (10–35 cm dbh) | | | | | | Large trees (>35 cm dbh) | | | | | |
| | Uncored* | | | Cored | | | Uncored* | | | Cored | | |
	N_0	N_t	m	N_0	N_t	m	N_0	N_t	m	N_0	N_t	m
Acer	50	44	1.5 (0.6, 3.2)	25	23	1.2 (0.1, 4.2)	16	15	0.7 (0.0, 4.0)	8	8	0.0
Carya	396	363	1.0 (0.7, 1.4)	198	177	1.6 (1.0, 2.4)	116	104	1.2 (0.7, 2.2)	58	55	0.8 (0.2, 2.2)
Fagus	172	171	0.1 (0.0, 0.4)	86	86	0.0	–	–	–	–	–	–
Fraxinus	34	25	3.5 (1.6, 6.5)	17	9	8.7 (3.7, 16.8)	30	23	3.0 (1.3, 6.2)	15	13	2.0 (0.2, 7.2)
Juglans	18	12	4.5 (1.8, 9.6)	9	8	1.7 (0.0, 9.0)	52	47	1.2 (0.4, 2.6)	26	25	0.6 (0.0, 3.1)
Liriodendron	90	77	1.8 (1.0, 3.1)	45	39	2.0 (0.7, 4.4)	90	89	0.1 (0.0, 0.7)	45	45	0.0
Nyssa	98	90	1.0 (0.5, 2.0)	49	45	1.2 (0.3, 3.1)	14	12	1.7 (0.2, 6.2)	7	5	4.7 (0.5, 16.2)
Quercus	168	117	4.0 (3.0, 5.3)	84	68	3.0 (1.7, 4.8)	386	319	2.2 (1.7, 2.7)	193	174	1.5 (0.9, 2.3)
Tilia	54	49	1.1 (0.4, 2.6)	27	27	0.0	18	15	2.1 (0.4, 5.9)	9	9	0.0
Ulmus	68	16	15.2 (11.1, 19.8)	34	13	12.9 (7.9, 19.4)	–	–	–	–	–	–
TOTAL	1148	964	2.0 (1.3, 2.3)	574	495	2.1 (1.7, 2.6)	722	624	1.7 (1.4, 2.0)	361	334	1.1 (0.7, 1.6)

*Uncored trees represent a subsample of all suitable uncored trees in the plot. N_0 represents the number by species and size class collected for each replicate. N_t represents the average number of trees alive across all replicates. Results for the uncored trees represent the mean mortality rate (m) and confidence intervals (in parenthesis) over 1000 replicates using a total of 1870 trees per replicate. High mortality rates for *Fraxinus* and *Ulmus* are likely caused by the emerald ash borer and Dutch elm disease that have caused extensive mortality to these species throughout their distribution in the eastern United States (Schlarbaum *et al*. 1998; Herms and McCullough 2014) and have been documented at this site (Gonzalez-Akre *et al*. 2016).

number of tree deaths, with confidence intervals, across 1000 random selections of uncored trees.

For each comparison, the annual mortality rate (*m*) was estimated following (Sheil *et al*. 1995):

$$m = \left[1 - \left(\frac{N_t}{N_0} \right)^{1/t} \right] \times 100 \quad (1)$$

Here, N_0 is the initial number of trees in the sample, and N_t is the number of surviving trees after a period of time (*t*). The time interval, *t*, was measured in years since the exact date of coring in 2010 (an average of 6.98 years) for cored trees or years since the date of the individual's initial census in 2008 (an average of 8.77 years) for uncored trees. We estimated 95% Confidence Intervals (CIs) using the normal approximation to the binomial variance for groups with more than five dead stems. When there were fewer dead trees, we estimated CIs using binomial probabilities (Sheil *et al*. 1995; Gonzalez-Akre *et al*. 2016). All data and R scripts are archived in public repositories at GitHub (https://github.com/SCBI-ForestGEO/Coring-Mortality) and with DOI on Zenodo (https://zenodo.org/record/1478207#.W-D6ZtVKjIV).

RESULTS AND DISCUSSION

We found no evidence that tree mortality rates increased over the seven years since coring. When pooled by genus and size class, the annual mortality rate of cored trees (1.71% yr^{-1}; 95% CI: 1.40–2.2% yr^{-1}) was slightly lower than that of uncored trees (1.85% yr^{-1}; 95% CI: 1.70–2.28 yr^{-1}). This difference was not statistically significant (p = 0.85) based on a Welch's two-sample t-test to account for unequal sample sizes. Differences in mortality rates between cored and uncored trees remained insignificant when making comparisons for large (p = 0.54) and small (p = 0.86) trees but continuing to pool across genera (Table 1).

The cored tree mortality rate fell within the 95% CI for uncored trees in 12 of 18 genus-size class combinations. Cored mortality rates were lower than uncored mortality rates for small *Juglans* (1.7% *vs*. 4.5%), large *Quercus* (1.5% *vs*. 2.2%), small *Tilia* (0.0% *vs*. 1.1%), and large *Tilia* (0.0% *vs*. 2.1%), but these differences were not statistically significant (p > 0.10). The only genus–size class combinations in which we found higher annual mortality rates in cored than uncored trees were for small *Fraxinus* (8.7% *vs*. 3.5%) and *Carya* (1.6% *vs*. 1.0%), but these rates were not statistically
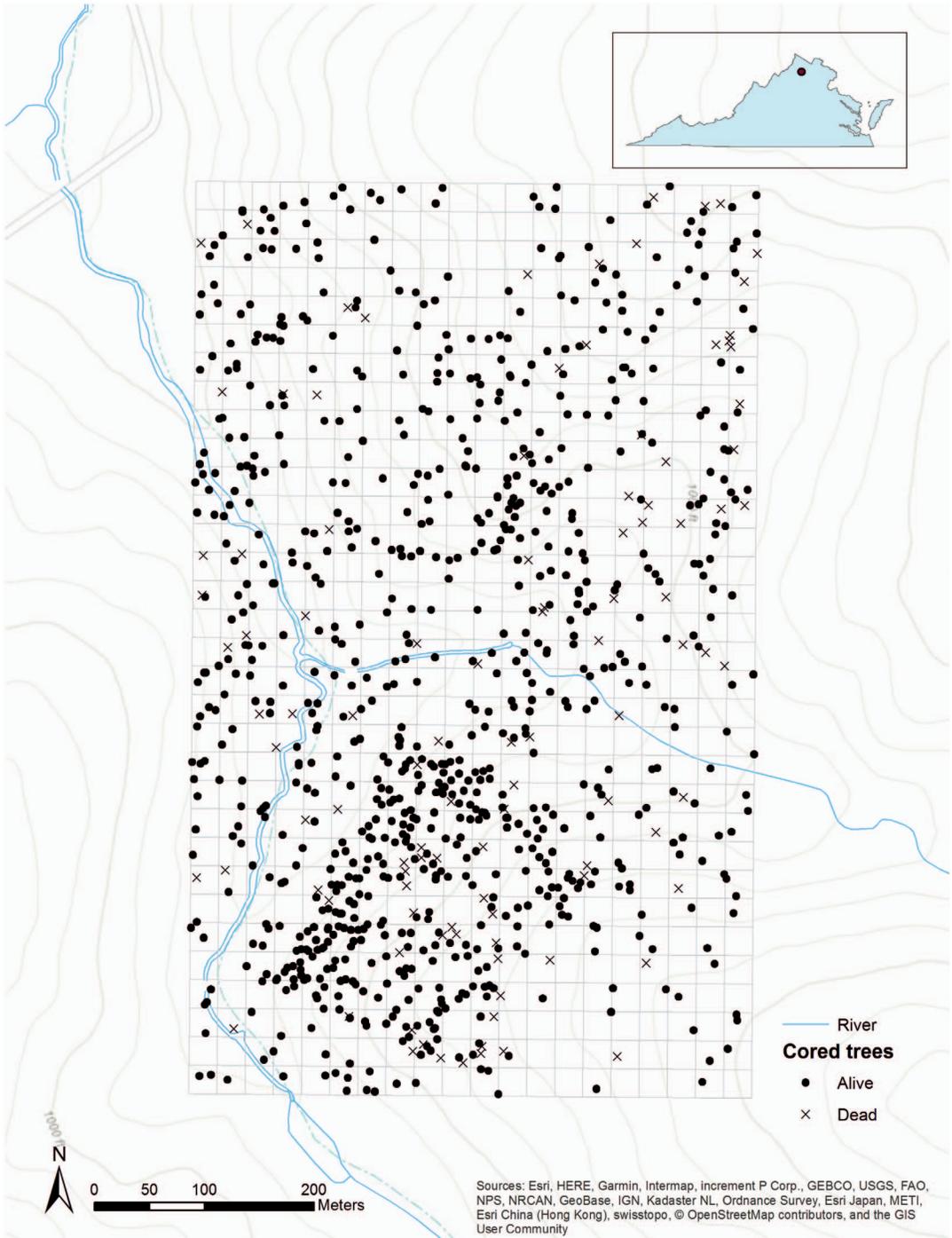
Figure 1. Location of each tree cored at the Smithsonian Conservation Biology Institute's 26-ha large forest dynamics plot in 2010. All trees were alive at the time of coring, and the trees marked as dead are those that had died by the 2017 census.

significant (p > 0.10). Differences in mortality rates are likely an artefact of our small samples size of cored trees for small *Fraxinus* and *Juglans* (n = 17 and 9, respectively) and large *Tilia* (n = 9; Table 1).

Despite the potential for coring to facilitate localized wood discoloration and decay (Hepting *et al*. 1949; Grissino-Mayer 2003), or for borers to break into a zone of compartmentalized decay in trees that had localized decay (*e.g.* heartrot) prior to coring (Norton 1998), we found no evidence that these factors increased mortality rates seven years after coring hardwood trees in our study. A seven-year interval is sufficient for cambial wound closure in most species (Tsen *et al*. 2016). However, we cannot rule out the possibility that coring could contribute to the mortality of some individuals over longer time scales. Such risk would most likely be limited to slow-growing individuals where the hole has not fully healed and the tree faces additional stressors. Even when core extraction does not cause mortality, it could reduce tree vigor because compartmentalizing the wound incurs energy costs (Shigo 1984; Smith 1988). Furthermore, our results do not necessarily apply to other hardwood forest types; coring may cause more severe wood damage with greater potential to contribute to tree mortality in wetter or warmer environments (*e.g.* wet tropics) where decay rates are faster, although this risk may be counteracted at the more productive sites where fast growth leads to relatively rapid wound closure (Neo *et al*. 2017).

Though further work will be required to rule out negative impacts over longer time scales or for applications where larger core diameters are needed (*e.g.* 12-mm diameter cores used for wood anatomy), our results show that the risk of temperate hardwood mortality from coring appears to be low for at least the first seven years following coring using a 5-mm diameter borer. These findings support the continued use of core extraction from live trees as a relatively low-impact technique in forest ecology and dendro-climatological research. Yet, it is still important to recognize that the holes produced by increment borers are an additional form of tree stress that could be harmful to slow-growing trees that already face multiple stresses and may take longer to heal the wound. Thus, when developing research plans, we support consideration of the best-practices, as laid out by Tsen *et al*. (2016),

to limit impacts on particularly vulnerable species and species with high conservation status.

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