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Effects of Nitrogen Enrichment on Early-Stage Wood Decomposition in a Southern Appalachian Forest

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ABSTRACT

Nitrogen (N) availability is shifting across forest ecosystems due to spatial variation in atmospheric N deposition and changing biogeochemical dynamics under global change. These shifts in N supply have implications for woody debris decomposition, a key process regulating carbon (C) turnover and storage in forests. Because wood is typically low in N, changes in external N availability can alter microbial activity and decomposition pathways, influencing both the rate and fate of C during decay. This study investigates how low-level N enrichment (5 kg N ha^{-1} yr^{-1}) affects the wood decomposition of two common tree species, *Liriodendron tulipifera* and *Pinus strobus*, in Southern Appalachian forests over a one-year period. We hypothesized that *L. tulipifera* would experience greater mass loss than *P. strobus* due to faster decomposition, and that N enrichment would accelerate decomposition of both species. Species *Liriodendron tulipifera* and *P. strobus* differed in wood chemical composition, particularly in lignin and cellulose content. In contrast, N enrichment had a minimal impact on decomposition dynamics. These results suggest that species is the dominant driver of early-stage wood decomposition. Given the short duration of this study and low rate of N enrichment, further research is needed to determine how these dynamics unfold over time and at what threshold N enrichment alters decomposition trajectories. As global changes reshape forest composition and nutrient patterns, understanding the combined influence of wood chemistry and nutrient availability will be essential for accurately predicting C storage in forest ecosystems.

Key words: carbon cycle, decomposition, forests, nitrogen enrichment, woody debris

INTRODUCTION

Forests play a central role in the global carbon (C) cycle, storing vast quantities of C and mediating its movement between the atmosphere, vegetation, and soils (Schulze et al. 2000; Jandl et al. 2007; Piao et al. 2022). A substantial portion of this C enters the detrital pool as woody debris (Harmon et al. 1986; Environmental Protection Agency 2024), where decomposition governs whether C is rapidly returned to the atmosphere or retained in the soil over longer timescales. Unlike leaf litter, which decomposes quickly and fuels short-term nutrient cycling, woody material persists for years to decades, making it a key control point for long-term C storage (Kueppers et al. 2004; Shannon et al., 2022). However, the rate and pathway of wood decomposition are not fixed; they are strongly influenced by nutrient availability, particularly nitrogen (N), which can both stimulate and constrain microbial activity depending on context (Bebber 2021; Roy et al. 2023). As N deposition and nutrient imbalances intensify under global change, understanding how N regulates wood decomposition is critical for predicting forest C dynamics and feedbacks to climate.

Decomposition is a complex, multi-stage process regulated by both environmental conditions and intrinsic wood properties. While early decomposition research emphasized the role of climate, more

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recent studies highlight the importance of local factors such as debris quality and moisture availability in driving decomposition (Bradford et al. 2014). Woody debris decomposition involves physical fragmentation, microbial colonization, and leaching, with rates spanning years to decades, varying according to chemical composition, size, and debris position (i.e., standing, fallen, or suspended; Harmon et al. 1986; Zhou et al. 2007; Wambsganss et al. 2017). Fine woody debris (FWD; <10 cm diameter) and coarse woody debris (CWD; ≥10 cm diameter) serve as a critical habitat and substrates for fungi and bacteria, which release enzymes that break down complex molecules like cellulose and lignin (Crawford 1981).

Lignin content plays a key role in decomposition dynamics (Harmon et al. 1986). Its complex chemical structure makes it more resistant to microbial degradation than other cell components. Microbes typically decompose cellulose and hemicellulose first, leaving lignin to accumulate and serve as an indicator of decay progression (Crawford 1981; Campbell et al. 2019). Lignin content varies across tree species, with coniferous species averaging higher concentrations than deciduous species (30% vs. 24%, respectively; Harmon et al. 1986). Consequently, deciduous wood tends to decompose faster than coniferous wood due to lower lignin concentrations (Weedon et al. 2009; Lukac and Godbold 2011; Shorohova and Kapitsa 2014; Herrmann et al. 2015). Moreover, studies comparing deciduous and coniferous wood decomposition often span years to decades with little research investigating the early stages of decomposition (Kueppers et al. 2004).

Because decomposition plays a central role in C cycling, differences in decay rates directly influence long-term forest C storage. External factors such as N availability can further influence these processes; however, most N enrichment research has focused on litter and soil rather than wood. For example, Edwards et al. (2011) found that N enrichment enhanced cellulolytic enzyme activity while simultaneously suppressing ligninolytic enzyme production in soils. Similarly, Van Miegroet and Jandl (2007) observed that high N inputs required substantial microbial biomass formation to process the excess N leading to increased microbial respiration rates and microbial biomass C. In contrast, Frey et al. (2014) reported a reduction in soil microbial biomass and enzymatic activity following N enrichment. Collectively, these findings suggest that N enrichment may either promote soil C sequestration by slowing lignin degradation or accelerate C loss via elevated microbial activity, allowing litter and soils to act as either C sinks or sources (Jandl et al. 2007; Hobbie et al. 2012; Frey et al. 2014). However, woody debris differs fundamentally from litter in its higher lignin content, structural complexity, and elevated C:N ratios (Harmon et al. 1986), characteristics that can impose stronger constraints on microbial colonization and enzymatic processing under N enrichment and delay decomposition (Stewart et al. 2015).

Despite interest in N effects on litter decomposition and soil C dynamics, relatively few studies have examined its N influences on wood decomposition (Qiao et al. 2016; Purahong et al. 2018; Bebber 2021; Roy et al. 2023). Bebber (2021) reported faster decay of fungi-inoculated wood blocks under high N, while other studies found mixed results—showing both inhibition and stimulation of decomposition depending on fungal communities and their enzyme responses (Purahong et al. 2018; Roy et al. 2023). Qiao et al. (2016) demonstrated that N enrichment can alter C:N ratios of leaf litter, wood, and soil, often reducing decomposition in high C:N substrates like wood. Together, these findings indicate that the direction and magnitude of N effects on wood decomposition remain unresolved, limiting our ability to predict whether elevated N suppresses or accelerates woody debris decay and, consequently, how N enrichment regulated forest C cycling.

In the Southern Appalachian region, forest composition reflects a legacy of natural succession and human intervention. Roughly 55% of the region is dominated by broadleaf, deciduous forests, while 19% is composed of planted stands of needle-leaf conifers, primarily Eastern White Pine, *Pinus strobus* L. (Wear and Greis 2012). Many pine stands were established during 20th century restoration efforts, including those led by the Civilian Conservation Corps (Vimmerstedt 1962). These coniferous-deciduous mosaic forests differ in soil properties such as low N in coniferous systems (Gosz 1981), increased soil moisture retention by broadleaf, deciduous litter (Babl et al. 2020), and

increased soil acidity from coniferous needles (Handley 1954; Abrahamsen et al. 1976). These factors have important implications for decomposition rates and nutrient dynamics.

This study leverages these forest differences to investigate the effects of low-level N enrichment (5 kg N ha^{-1} yr^{-1}) on wood decomposition from two common trees. This region is especially suited for such a study as recent reviews have indicated that N limitation may be more pronounced in mountainous soils and in forests located away from major urban centers (Du et al. 2020; Du et al. 2024). Specifically, we asked:

(i) How does wood decomposition differ between *Liriodendron tulipifera* L. in broadleaf-dominated deciduous forest and *P. strobus* in a conifer-dominated forest over one year?
(ii) Does N enrichment impact wood decomposition in either species?

We hypothesized that *L. tulipifera* would exhibit greater mass loss after one-year due to lower lignin concentration relative to *P. strobus*. Additionally, we hypothesized that N enrichment would accelerate decomposition for both species, due to heightened microbial activity facilitated by sufficient N availability.

MATERIALS AND METHODS

Study Site

This study was conducted at the Robert Gilley Field Station (36.2791, –81.5815), a property of Appalachian State University located in western North Carolina, United States between fall 2023 and spring 2025. This region's mean annual precipitation is 1,404.85 mm and has a mean annual temperature of 9.48°C (PRISM 2023). This site supports a mix of broadleaf, deciduous hardwoods and needle-leaf conifers, with species primarily belonging to Aceraceae, Betulaceae, Juglandaceae, and Pinaceae (Swanson et al. 2011). The soil texture is loam to sandy loam (Soil Survey Staff 2025). In fall 2023, six 30 m × 30 m plots were established—three coniferous-dominant forest (hereafter, "CF") plots and three deciduous-dominant forest (hereafter, "DF") plots—across three elevational blocks each approximately 30 m apart (low, mid, high, Figure 1; Thompson et al. 2011). We used a two-way fixed block design, with plots stratified by elevation (low, mid, high) and forest type (coniferous vs. deciduous), resulting in six plots (one of each forest type per elevation, Table 1). Both factors were observational and not manipulated.

Study Design

To test how the early stages of wood decomposition differ between coniferous wood and deciduous wood and how N enrichment influences this process, we established two 2 m × 4 m nutrient treatment subplots (N addition and control) within each of the six forest plots. To prevent N runoff contamination, we positioned the control subplots in the upslope NW corner of each plot and the N enrichment subplots in the downslope SE corner. In the N enrichment subplots, we applied 26.09 g of time-released urea (46% N) every three months, resulting in a N deposition rate of 5 kg N ha^{-1} yr^{-1}. This resulted in a total N deposition rate of approximately 11.7 kg N ha^{-1} yr^{-1}, representing an increase of about 75% over the ambient background level of ~6.7 kg N ha^{-1} yr^{-1} (Walker et al. 2023).

For the decomposition experiment, *Liriodendron tulipifera* was selected as the deciduous species and *Pinus strobus* as the coniferous species (Roy et al. 2023) due to their dominance in their respective forests. We cut 24 logs of each species to a length of 15 cm and a circumference of 15 ±2 cm, recording the initial wet mass of each log (Shannon et al. 2022). Each log was tagged with a numbered marker before gently laying four logs lengthwise across the 2 m width of each subplot, thereby spaced evenly within five 0.8 m sections along the 4 m length. *Liriodendron tulipifera* was placed only in deciduous-dominated forest plots, and *P. strobus* only in conifer-dominated plots, to reflect natural species distributions and decomposition conditions. Logs were harvested after six months and one year, with two replicate logs retrieved from each subplot at each harvest. Soil samples (10 cm deep) were collected prior to experiment setup, air dried, sieved, and analyzed on an elemental analyzer at the Kansas State University Soil Testing Lab for total C

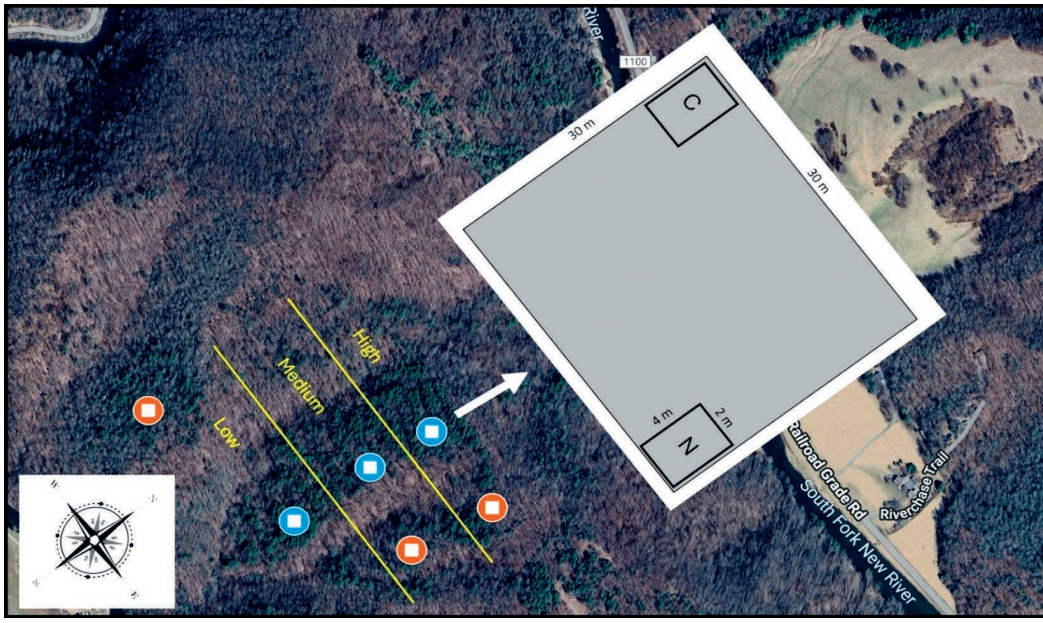

Figure 1. Satellite image of the Robert Gilley field station with an inset picture depicting plot layout. Red points represent DF plots, while blue points are the CF plots. Within the inset picture, C denotes control subplots and N denotes N treatment subplots.

Table 1. Experimental Design

Levels (nested)	Replicates	Notes
Forest Type	2	CF, DF
Plots per Forest Type	3	Low, Medium, High
Subplots per Plot	2	Control, Nitrogen
Timepoints per Subplot	2	6 months, 1 yr
Logs per Timepoint	2	Rep 1, Rep 2

and N %. Initial soil samples revealed DF plots had 5.08% C and 0.37% N while CF plots had 4.7% C and 0.26% N.

To assess log decomposition, we first measured their wet mass, length, and diameter before placing them in a drying oven at 60°C for 48 hours (Laiho and Prescott 1999). Once dry, mass was measured again to quantify mass loss after each harvest. Next, we prepared the log samples for shipment to the Kansas State University Soil Testing Lab for total N and C analysis. To prepare these samples, we used an Irwin Speedbor 9.5 mm spade bit to produce wood shavings by drilling twice—once on the top and once on the bottom—positioning each hole 5 cm from the log ends (Fukasawa et al. 2019; Roy et al. 2023). To collect the wood shavings produced during drilling, we placed a piece of folded paper beneath each log. The shavings from both drill sites of the same log were combined into a single Whirl-Pak bag to create a mixed sample (Fukasawa et al. 2019). Next, we cut two 1 cm thick disks from the middle of each log, avoiding the drilled areas. These disks were sent to DairyOne (Ithaca, New York) for lignin, cellulose, and hemicellulose analyses. Lignin content was determined using sulfuric acid treatment. Cellulose was calculated as the difference between acid detergent fiber (ADF) and lignin, while hemicellulose was estimated as the difference between neutral detergent fiber (NDF) and ADF (Stewart et al. 2015).

To evaluate FWD decomposition, FWD was collected from the same *Pinus strobus* and *Liriodendron tulipifera* trees used to create the logs. Debris was processed into sections ~14 cm in length and separated into two diameter classes: small (≤0.6 cm) and large (≥2.5 cm). Two large and three small pieces of FWD were added to 20 cm^2 bags that were constructed from 1 mm mesh material. The initial mass of FWD in each mesh litter bag was recorded before three bags were placed in each subplot, and additional FWD was brought back to the lab, dried, and weighed for a dry-weight correction. Like the logs, *L. tulipifera* FWD was placed only in deciduous-dominated forest plots, and *P. strobus* FWD only in conifer-dominated plots, to reflect natural species distributions and decomposition conditions. FWD litterbags were harvested at six weeks, three months, and 12 months, with one replicate litter bag collected from each subplot at each harvest. Upon returning to the lab, all samples were transferred to a refrigerator and stored at 4°C until processing (Stewart et al., 2015). To assess FWD decomposition, litterbags were dried at 60°C for 48 hours and FWD was weighed to quantify mass loss at each harvest time. FWD samples were not assessed for wood chemistry.

Statistical Analysis

All statistical analyses were performed in R version 2023.12+369 (R Core Team 2023). We used three-way ANOVAs to test if log species, N treatment, or harvest time, and their interactions impacted woody mass loss for logs and FWD, log tissue properties (i.e. lignin, cellulose, and hemicellulose content), and log C and N content (R package stats). Statistical significance was assessed at α=0.05. Datasets for this study are archived in Dryad as DOI: 10.5061/dryad.zpc866tnk.

RESULTS

Total log decomposition (measured as % mass remaining) was strongly influenced by species (p<0.001, F=23.3; Table 2, Fig. 2A), with *Liriodendron tulipifera* exhibiting consistently greater decomposition than *Pinus strobus* logs across all harvest times and treatments. By 12 months, *L. tulipifera* had 92.32% remaining mass while P. strobus had 112.10% remaining mass, a percentage difference of 17.65%. Furthermore, harvest time affected decomposition: *L. tulipifera* lost mass whereas *P. strobus* gained mass overtime (p<0.001, F=9.1; Table 1, Fig. 2A). Nitrogen enrichment showed slowing decomposition trends for *L. tulipifera* but did not affect *P. strobus* (p=0.058, F=3.7; Table 1, Fig. 2B). After 12 months, *L. tulipifera* mass remaining increased from 87.31% in controls to 97.33% with nitrogen, indicating an 11.47% reduction in decomposition. A significant harvest time × species interaction was detected (p<0.001, F=25.2; Table 2), where *L. tulipifera* experienced substantial mass loss after 12 months, whereas *P. strobus* logs experienced mass gain over time (Fig. 2A).

Wood density differed by species, where the *Pinus strobus* had overall lower density than the *Liriodendron tulipifera* logs (p<0.001, F=221.6 Table 2, Fig. 2C). Over time of decomposition, *L. tulipifera* logs decreased in density, while *P. strobus* logs did not change (p=0.01, F=4.8; Table 2; Fig. 2C). Harvest time (p=0.01, F=4.8; Table 2, Fig. 2C) and N enrichment treatment (p<0.001, F=18.0; Table 2, Fig. 2D) also had significant effects on log density. Furthermore, significant interactions were detected between species and N enrichment treatment (p<0.001, F=16.6; Table 2; Fig. 2D), where N increased density of the *L. tulipifera* logs, but the *P. strobus* logs were not affected by N enrichment treatment.

Lignin content varied significantly by species (p<0.001, F=470.8; Table 3, Fig. 3A), with *Pinus strobus* exhibiting higher lignin concentrations than *Liriodendron tulipifera* logs overall. Both harvest time (p=0.01, F=7.8; Table 3, Fig. 3A) and N enrichment treatment (p=0.0081, F=8.3; Table 3, Fig. 3B) also influenced lignin content: relative lignin content increased over time in *P. strobus* logs in both treatments, whereas relative lignin content increased under N enrichment treatment but not control treatment in *L. tulipifera* logs after 12 months.

Cellulose content was significantly greater in *Liriodendron tulipifera* logs compared to *Pinus strobus* logs (p<0.001, F=145.8; Table 3, Fig. 3C). Harvest time also affected cellulose content (p=0.0035, F=10.5; Table 3, Fig. 3C), with both species generally exhibiting relative increases over

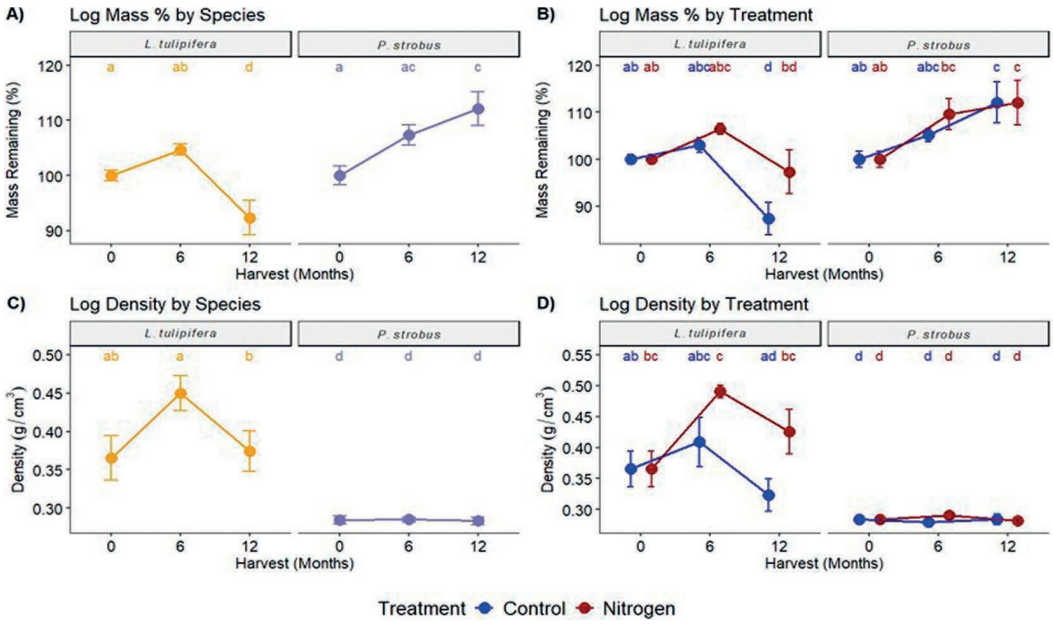

Figure 2. Percent change in log mass (**A**, **C**) and wood density (**B**, **D**) by species (**A**, **B**; n=12) and by species × treatment (**C**, **D**; n= 6). Points represent means, and error bars indicate standard error (SE). Species colors are yellow for *Liriodendron tulipifera* and purple for *Pinus strobus*; N treatments are shown in blue (control) and red (N addition). Different letters indicate statistically significant differences (e.g., "a", "b").

Table 2. ANOVA results for log mass % and density. Df=degrees of freedom, F=F-statistic, *p*=p-value (α=0.05, *: *p*<0.05).

		Log Mass %		Log Density	
	Df	F	p	F	p
Species	1	23.3	**<0.001***	221.6	**<0.001***
Treatment	1	3.7	0.058	18.0	**<0.001***
Harvest	2	9.1	**<0.001***	4.8	**0.01***
Species:Treatment	1	0.9	0.34	16.6	**<0.001***
Species:Harvest	2	25.2	**<0.001***	4.8	**0.01***
Treatment:Harvest	2	1.9	0.16	0.4	0.69
Species:Treatment:Harvest	2	1.9	0.15	0.4	0.70

the 12-month period. However, no effect of N enrichment treatment on cellulose content was detected in either species (p=0.49, F=0.5; Table 3, Fig. 3D). There was significant interaction between species and harvest time (p=0.0064, F=8.9; Table 3), driven by an increase in cellulose content over time in *L. tulipifera* logs, while *P. strobus* logs instead remained unchanged.

In contrast to lignin and cellulose, hemicellulose content exhibited no significant patterns. Harvest time (p=0.35, F=0.9; Table 3, Fig. 3E), species (p=0.15, F=2.2; Table 3, Fig. 3E), N enrichment treatment (p=0.56, F=0.3; Table 3, Fig. 3F), and their interactions (p>0.05; Table 3) had no effect on hemicellulose concentrations.

Total C % differed significantly by species (p<0.001, F=165.4; Table 4, Fig. 4A) and harvest time (p=0.047, F=3.3; Table 4, Fig. 4A), with *Pinus strobus* consistently exhibiting higher C % than *Liriodendron tulipifera* logs across the 12-month period. In contrast, no effect of N enrichment

Table 3. ANOVA results for log lignin %, cellulose %, and hemicellulose %. Df=degrees of freedom, F=F-statistic, *p*=p-value (α=0.05, *: *p*<0.05).

		Lignin %		Cellulose %		Hemicellulose	
	Df	F	p	F	p	F	p
Species	1	470.8	<**0.001***	145.8	<**0.001***	2.2	0.15
Treatment	1	8.3	**0.0081***	0.5	0.49	0.3	0.56
Harvest	1	7.8	**0.010***	10.5	**0.0035***	0.9	0.35
Species:Treatment	1	4.2	0.051	0.05	0.83	0.8	0.37
Species:Harvest	1	12.7	**0.0016***	8.9	**0.0064***	4.1	0.053

treatment was detected (p=0.77, F=0.09; Table 4, Fig. 4A), and there were no significant interactions among harvest time, species, and N enrichment treatment (p>0.05, Table 4). Total N % also differed significantly by species (p<0.001, F=15.9; Table 4, Fig. 4C) and harvest time (p<0.001, F=103.5; Table 4, Fig. 4C), whereas N enrichment treatment had no significant effect (p=0.17, F=1.9; Table 4, Fig. 4C). *Liriodendron tulipifera* consistently exhibited higher total N % than *P. strobus* logs, and both species showed no change in total N % between 0 and 6 months but experienced significant increases by 12 months.

Similarly, species and harvest time significantly affected wood C:N ratios (p=0.0062, F=8.2 and p<0.001, F=18.5, respectively; Table 4, Fig. 4E), with no effect of N enrichment treatment (p=0.55, F=0.4; Table 4, Fig. 4E). A significant harvest time × species interaction was detected (p=0.048, F=3.3; Table 4, Fig. 4E), reflecting a decrease in C:N ratios for *Pinus strobus*, but not *Liriodendron tulipifera*, over the 12-month period.

Overall, *L. tulipifera* FWD exhibited greater mass loss compared to *P. strobus* FWD at the 3- and 12-month harvest periods (Fig. 5A). Total FWD decomposition was significantly affected by species (measured as % mass remaining; p<0.001, F=63.2; Table 5, Fig. 5A), where *L. tulipifera* FWD lost more mass by 12 months than *P. strobus* FWD despite significant declines in mass between the 6- and 12-month harvests for both species. While mass decreased significantly for both species over time of decomposition (p<0.001, F=202.2; Table 5, Fig. 5A), a significant interaction between species and harvest time was detected (p<0.001, F=16.5; Table 5) due to the faster decomposition of *L. tulipifera*. Nitrogen enrichment treatment had no effect (p=0.97, F=0.002; Table 5, Fig. 5B).

DISCUSSION

This study revealed that early-stage wood decomposition is primarily driven by species identity, with low-level nitrogen enrichment exerting little influence on the initial decomposition processes. Over a 1-year period, *Pinus strobus* logs retained more mass than *L. tulipifera* logs, with the same pattern observed for FWD, regardless of N treatment. These results support well-established findings that coniferous wood decomposes more slowly than deciduous wood, largely due to differences in lignin concentrations and structural composition (Weedon et al. 2009; Shorohova and Kapitsa 2014).

Species chemical composition drives decomposition rates

Wood chemical properties, particularly lignin content, strongly influenced decomposition. *Pinus strobus* had higher lignin concentrations than *Liriodendron tulipifera*, consistent with prior studies reporting coniferous trees possess more lignin than deciduous trees (Lukac and Godbold 2011; Herrmann et al. 2015). Because lignin is resistant to microbial degradation, microorganisms break down less complex molecules such as cellulose and hemicellulose first. Since deciduous wood, such as that of *L. tulipifera*, contains less lignin, decomposition tends to proceed at a faster rate. Thus, this difference in chemical composition influenced the trends observed in wood chemical contents over time.

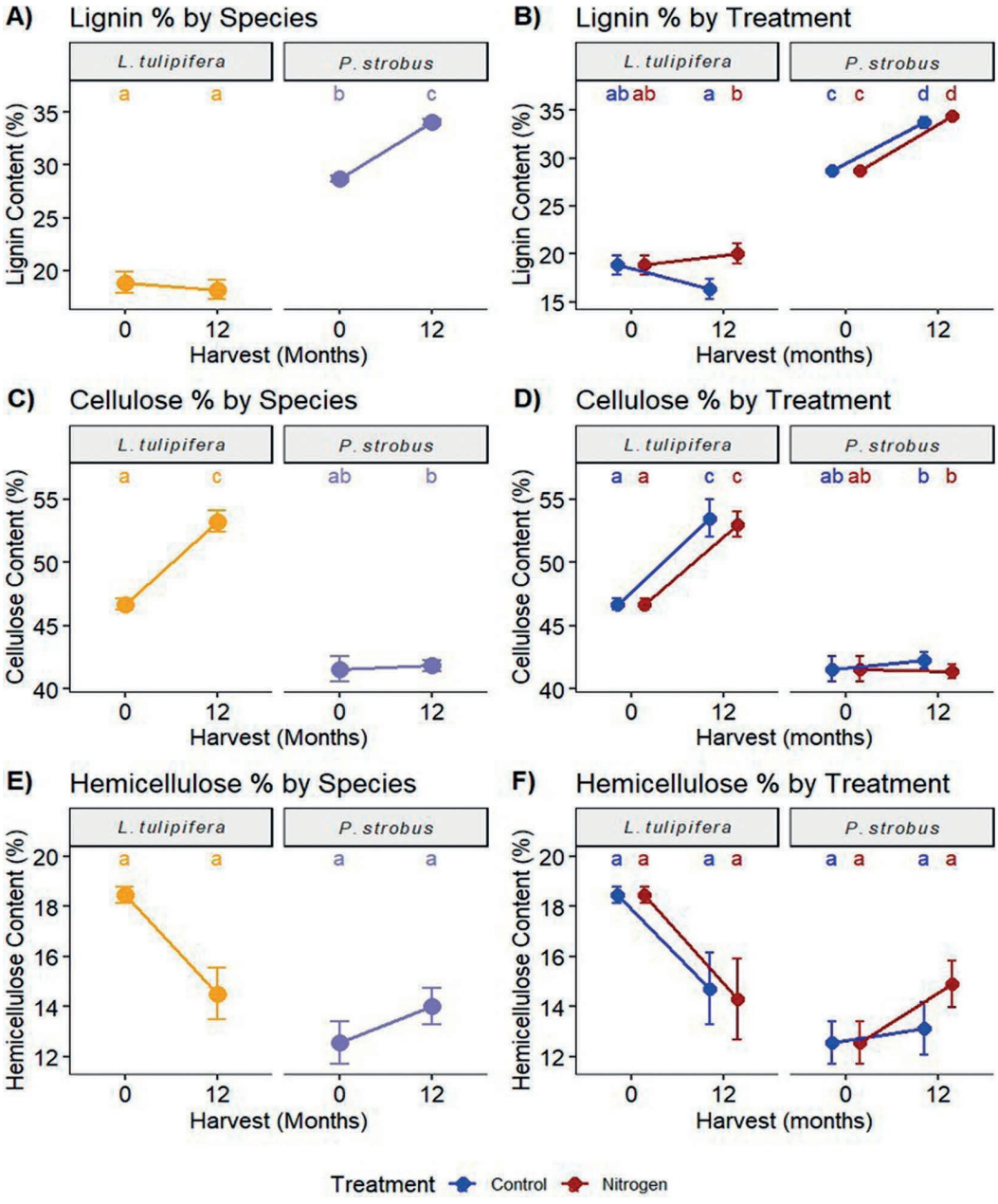

Figure 3. Lignin (**A**, **D**), cellulose (**B**, **E**), and hemicellulose (**C**, **F**) content (%) by species (**A**–**C**; n=12) and by species × treatment (**D**–**F**; n=6). Data points represent means, with error bars indicating standard error (SE). Species colors are yellow for *Liriodendron tulipifera* and purple for *Pinus strobus*; N treatments are shown in blue (control) and red (N addition). Different letters indicate statistically significant differences (e.g., "a," "b").

Table 4. ANOVA results for log C %, N %, and C:N. Df=degrees of freedom, F=F-statistic, *p*=p-value (α=0.05, *: *p*<0.05).

		Log C %		Log N %		Log C:N	
	Df	F	p	F	p	F	p
Species	1	165.4	<**0.001***	15.9	<**0.001***	8.2	0.0062*
Treatment	1	0.09	0.77	1.9	0.17	0.4	0.55
Harvest	2	3.3	**0.047***	10.35	<**0.001***	18.5	<**0.001***
Species:Treatment	1	0.04	0.85	1.2	0.27	0.1	0.76
Species:Harvest	2	0.2	0.80	2.1	0.14	3.3	**0.048***
Treatment:Harvest	2	0.02	0.98	0.2	0.78	0.2	0.82
Species:Treatment: Harvest	2	0.02	0.98	0.9	0.41	0.01	0.99

Interestingly, *Liriodendron tulipifera* also showed elevated cellulose levels during decomposition. This may reflect not only chemical differences in wood content, but also possible fungal colonization due to methodological factors. Here, the ADF method used to determine cellulose content may also capture some fungal material (Soest 1963), which can inflate cellulose estimates. These changes may signal early microbial colonization, as microorganisms often accumulate nutrients from the environment before breaking down structural components (Cornwell et al. 2009). This colonization and accumulation of nutrients may also explain the observed density increase in the *L. tulipifera* logs and the mass increase in the *Pinus strobus* logs.

Log and soil C:N ratios further support species-driven decomposition patterns. *Liriodendron tulipifera* logs exhibited consistently lower C:N ratios than the *Pinus strobus* logs, consistent with faster decomposition as lower C:N ratios are characteristic of a rapid decrease in C content with decomposition (Cornwell et al. 2009). Thus, further indicates the role of species as a primary driver of early decomposition. Similarly, soils beneath *L. tulipifera* logs showed lower C:N ratios than soils beneath *P. strobus* logs, suggesting species-specific influences on nutrient cycling extend beyond the debris itself. Long-term forest composition—specifically coniferous- versus deciduous-dominant systems—may play a more influential role in shaping soil nutrient balance than short-term N enrichment (Van Der Sande et al. 2023). Overall, these results highlight the complexity of decomposition dynamics and emphasize the importance of considering broader environmental controls, such as species composition, when evaluating soil nutrient patterns in forest ecosystems.

Limited effects of nitrogen enrichment

Contrary to expectations, N enrichment (5 kg ha^{-1} yr^{-1}) did not significantly affect mass loss in either species. However, *Liriodendron tulipifera* logs harvested after 12 months showed an increase in lignin content under N addition. This pattern may suggest a suppression of microbial lignin-degrading activity by fertilization, a phenomenon reported in previous studies (Hobbie et al. 2012; Frey et al. 2014). However, studies such as Hobbie et al. (2012) applied relatively high N concentrations (~50 kg N ha^{-1} yr^{-1}), which have been criticized for exceeding realistic environmental levels (Bebber 2021). While our lower N application (5 kg N ha^{-1} yr^{-1}) more closely reflects natural deposition rate for the Southern Appalachian region (~6.7 kg N ha^{-1} yr^{-1}; Walker et al. 2023), the lack of a strong treatment effect suggests that small increases in N may not be sufficient to alter decomposition dynamics over short timescales.

While few studies have explored the effects of N enrichment on fungal community shifts and enzymatic activity in woody debris (i.e., Qiao et al. 2016; Purahong et al. 2018; Bebber 2021; Roy et al. 2023), fewer have evaluated how N availability influences physical decomposition outcomes. This study provides a complementary perspective by testing N enrichment effects by assessing both mass loss and changes in wood chemical properties to advance the understanding of how changes in N deposition may alter early-stage decomposition and forest nutrient cycling.

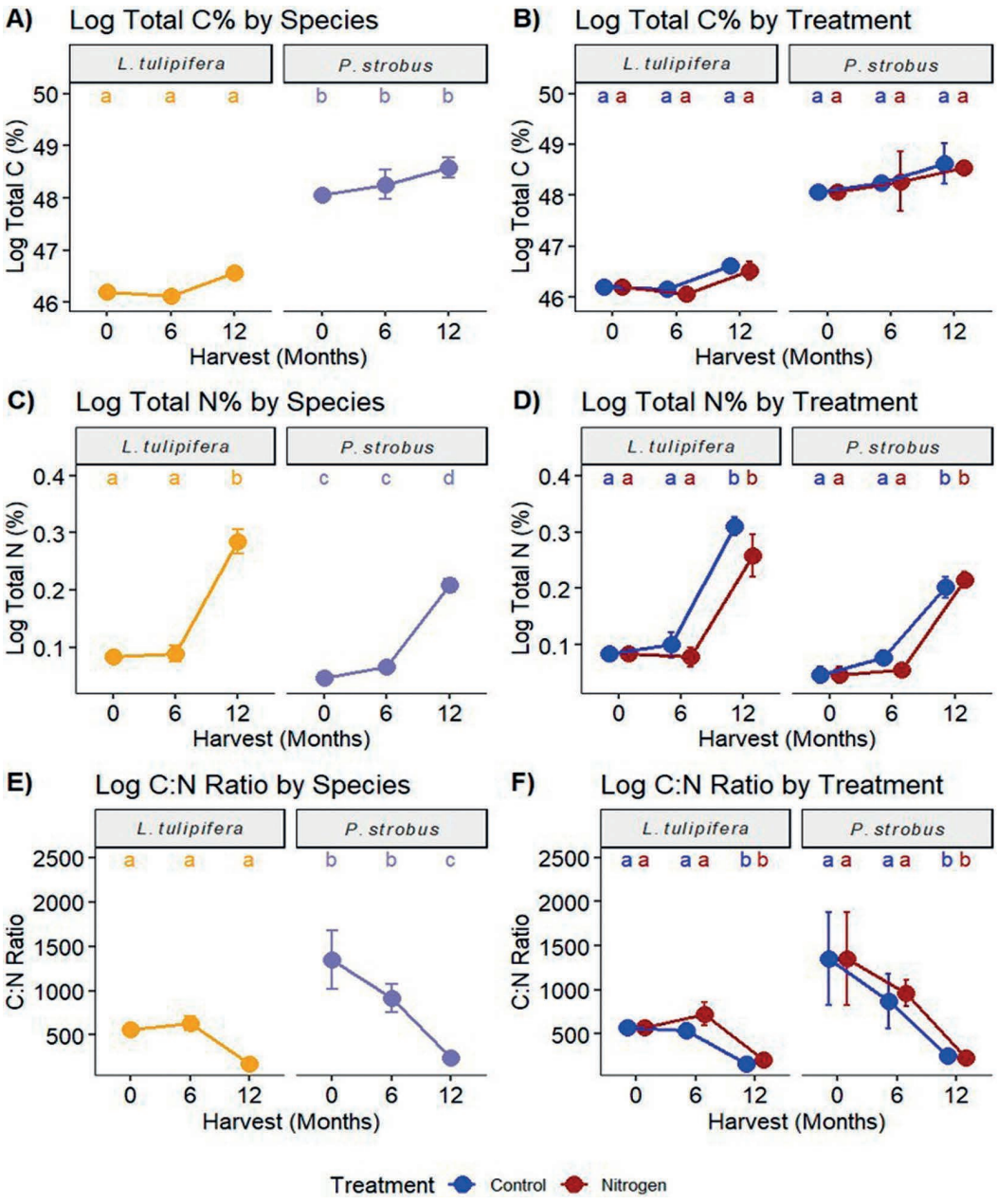

Figure 4. Total carbon (C) % (**A**, **B**), total nitrogen (N) % (**C**, **D**), and C:N ratio (**E**, **F**) by species (**A**–**C**; n=12) and by species × treatment (**D**–**F**; n=6). Data points represent means, with error bars indicating standard error (SE). Species colors are yellow for *Liriodendron tulipifera* and purple for *Pinus strobus*. N treatments are shown in blue (control) and red (N addition). Different letters indicate statistically significant differences (e.g., "a", "b").

Table 5. ANOVA results for fine woody debris (FWD) mass %. Df=degrees of freedom, F=F-statistic, *p*=p-value (α=0.05, *: *p*<0.05).

	FWD Mass %		
	Df	F	p
Species	1	63.2	<**0.001***
Treatment	1	0.002	0.97
Harvest	3	202.2	<**0.001***
Species:Treatment	1	0.4	0.53
Species:Harvest	3	16.5	<**0.001***
Treatment:Harvest	3	0.4	0.73
Species:Treatment:Harvest	3	0.6	0.62

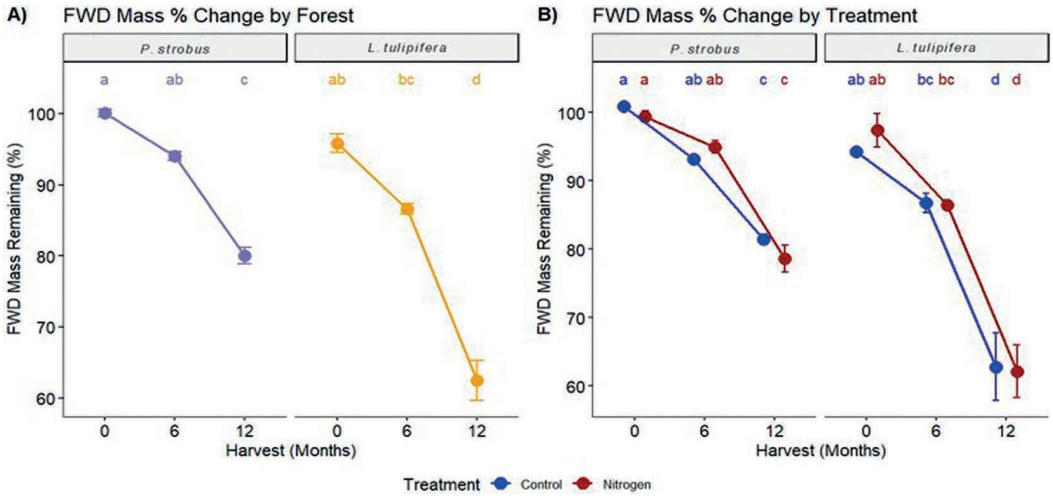

Figure 5. Percent change in FWD mass by species (**A**; n=12) and by species × treatment (**B**; n=6). Data points represent means, with error bars indicating standard error (SE). Species colors are yellow for *Liriodendron tulipifera* and purple for *Pinus strobus*; N treatments are shown in blue (control) and red (N addition). Different letters indicate statistically significant differences (e.g., "a," "b").

Future directions

There are several important caveats to consider when interpreting the results of this study. First, species were assigned to their respective dominant forest types (*Liriodendron tulipifera* in deciduous plots; *Pinus strobus* in coniferous plots), which precluded testing for any interactions between species and forest environment. Thus, the potential environmental effects on species-specific decomposition were not directly addressed. A full factorial design would allow more precise assessment of species × environment interactions. Second, the application of N across the subplots followed a uniform distribution, yet background N deposition can vary within forests from canopy architecture and the pooling of nutrients around tree trunks (Butler and Likens 1995; Levia et al. 2011). Thus, future studies that apply a gradient of N could determine the threshold at which elevated N alters microbial activity and decomposition dynamics, thereby offering a more comprehensive understanding of its effects on forest processes. Similarly, using a broader gradient of wood sizes could reveal size-dependent decomposition rates and nutrient effects that our data and other studies have confirmed; larger logs decompose more slowly than smaller woody debris (Graham 1981; Graham and Cromack 1982; Harmon et al. 1986;). Third, the experiment was established on a south-facing slope, which receives

high solar exposure and therefore experiences drier conditions than north-facing slopes. If the study had been conducted on a north-facing aspect, the cooler and moister microclimate may have facilitated more rapid decomposition (Amstutz et al. 2024). Thus, inclusion of multiple aspects is important to consider for future studies. Finally, the duration of this study is notably short relative to the time scales over which wood decomposition typically occurs. While this timescale limits the ability to draw conclusions about decomposition rates and total carbon turnover, these findings highlight the interaction of microbial colonization and nutrient abundance in early-stage wood decomposition. To build on these findings, future research should extend decomposition monitoring beyond one year to determine whether the species-specific trends observed here persist, amplify, or shift over longer timescales. Incorporating microbial community analyses—such as characterizing fungal and bacterial assemblages—would offer valuable insight into the biological mechanisms driving decomposition. In addition, measuring enzyme activity alongside changes in wood chemical properties could help clarify how N availability interacts with intrinsic wood traits to regulate decomposition processes.

Conclusions

This study highlights the central role of species and intrinsic chemical chemistry on early-stage wood decomposition. The limited response to low-level N enrichment suggests that modest increases in atmospheric N deposition may have negligible effects on short-term wood decomposition dynamics in this system. To advance understanding, future research should extend the duration of decomposition monitoring, incorporate microbial community analyses, and measure enzyme activity alongside nutrient cycling metrics. Together, these approaches will clarify how decomposition processes respond to changes in forest composition and nutrient inputs in a changing climate.

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