

YOUNG VOICES AND VISIONS FOR THE UN DECADE OF RESTORATION

RESEARCH ARTICLE

Lianas (*Vitis* spp.) reduce growth and carbon sequestration of light-demanding tree species in a temperate forest

Jacob D. J. Peters^{1,2} , Julia M. Portmann³ , Bronson W. Griscom⁴

Governments worldwide are seeking natural climate solutions that can provide economic stimulus while meeting climate goals. Forests provide essential carbon dioxide (CO₂) sequestration services, but their potential may be limited by elevated liana densities often resulting from human disturbance. Here we report the first estimate of liana (*Vitis* spp.) impacts on CO₂ removal rates by trees in temperate Appalachian forests and suggest liana removal with biodiversity safeguards as a potential strategy for improved forest management. Shade-intolerant tree species without lianas removed CO₂ 20% faster than conspecifics with lianas ($p = 0.025$). We did not detect significant impacts of liana presence on the CO₂ removal rates for shade-tolerant species ($p = 0.838$). Additionally, the merchantable boles of trees without lianas increased in volume 9% faster annually compared to trees with lianas, for all species ($p = 0.003$). Our findings indicate that thinning lianas—particularly from light-loving species such as *Liriodendron tulipifera*—may improve sustainability of forest management and increase carbon storage while mitigating climate change.

Key words: carbon storage, forest restoration, improved forest management, natural climate solutions, shade tolerance

Implications for Practice

- Releasing forest stands from heavy liana infestation offers a simple natural climate solution while improving forest stand growth.
- Removal of lianas from light-demanding commercially valuable tree species hosting elevated liana loads is most likely to deliver both strong positive growth response and a cost-effective improvement to timber yields.

Introduction

Improved land stewardship can provide a major contribution to national climate targets in both industrialized and developing countries. Therefore, governments around the world are looking for interventions that link economic stimulus with climate action. A promising approach is through natural climate solutions (NCS), defined by Griscom et al. (2017) as terrestrial conservation, restoration, and improved practices pathways that include safeguards for food, fiber, and habitat. Improved management of natural forests harvested for timber is the second largest of such options for the United States (Fargione et al. 2018). This NCS is among the most impactful and cost-effective options globally and delivers a range of ecosystem services (Griscom et al. 2017; Roe et al. 2021). Here we explore the potential of an improved natural forest management technique: thinning lianas in temperate Appalachian forests of eastern

United States. Cutting liana stems (primarily of the genus *Vitis*) in timber-producing native forests where human activities have elevated liana abundance has the potential to accomplish both globally and locally important outcomes simultaneously: (1) faster tree growth and removal of carbon dioxide (CO₂) from the atmosphere, and (2) improved sustainable economic value of forest stands due to increased timber volume and improved growth form over time.

Incentives for Improving Forest Growth

The Appalachian region of Eastern North America is home to valuable timber species such as tulip poplar (*Liriodendron tulipifera* L.), northern red oak (*Quercus rubra* L.), and sugar maple (*Acer saccharum* Marshall) (Weakley et al. 2012; USDA-NRCS 2021; Virginia Botanical Associates 2021).

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¹The Forest School at The Yale School of the Environment, 360 Prospect Street, New Haven, CT 06511, U.S.A.

²Address correspondence to J. D. J. Peters, email jacob.peters@yale.edu

³Biology Department, James Madison University, 951 Carrier Drive, Harrisonburg, VA 22807, U.S.A.

⁴Conservation International's Center for Natural Climate Solutions and Moore Center for Science, 2011 Crystal Drive, Arlington, VA 22202, U.S.A.

Harvesting these tree species for wood products can be worth up to \$5,000 per acre as of 2022 and is a significant source of income for landowners and agencies such as the United States Department of Agriculture Forest Service (Jacobson 2008; Ray 2022). Naturally, landowners and agencies interested in timber harvest want to maximize profit per effort which can be aided by fast growth and optimal growth form. However, landowners of small nonindustrial Appalachian forests seldom invest in improving tree growth and form for many reasons, including the extended time delay for a return on investment in silviculture and a possible lack of access to forest management education (Joshi & Arano 2009).

Restoring tree growth and carbon storage in degraded forests can provide a promising NCS if safeguards for biodiversity and food and fiber security are met (Fahey et al. 2009; Griscom et al. 2017; Fargione et al. 2018). Restoring chronically degraded forests could be accomplished by accelerating forest growth, which may provide carbon, biodiversity, and resilience benefits for centuries into the future through enhancing forest successional regimes and allowing them to reach a steady-state system sooner (Smithwick et al. 2002; Luyssaert et al. 2008).

Liana Impacts

Lianas are structural parasites that, unlike trees, minimize allocation of resources to structural support, and hence have a dramatically lower ratio of carbon storage per unit leaf area (Darwin 1865; Della-Bianca 1979; Putz & Mooney 1992). Furthermore, lianas have been shown to significantly impact tropical forest carbon stocks (Durán & Gianoli 2013; Schnitzer et al. 2014; Finlayson et al. 2022). In tropical forests, large trees comprise upwards of 90% of overall biomass, but are frequently colonized by lianas, reducing their carbon storage potential (Durán & Gianoli 2013). Aboveground carbon storage in the tropics across five continents has been estimated at 146.5 ± 6.04 Mg C/ha (megagrams of carbon per hectare) in adult trees greater than 10 cm diameter, compared to 5.56 ± 0.47 Mg C/ha by lianas, which may reduce large trees' carbon storage by up to 50% (Durán & Gianoli 2013). Furthermore, the preference of tropical lianas to colonize gaps reduces overall carbon sequestration rate by $0.06\text{--}0.12$ Mg C ha⁻¹ yr⁻¹ because tree growth is hindered (Schnitzer et al. 2014; Schnitzer et al. 2021). The established relationship between carbon storage and liana prevalence in tropical regions emphasizes the importance of determining the impact of lianas on carbon sequestration rates in temperate regions.

Lianas have the capacity to reduce the economic value forest stands by slowing growth and increasing mortality of trees (Featherly 1941; van der Heijden & Phillips 2009; Schnitzer et al. 2014). Trees hosting lianas may suffer not only reduced photosynthesis, but also broken branches, twisted growth patterns, and uprooting or breaking of the main stem during severe weather events (Siccama et al. 1976; Trimble & Tryon 1979; Smith 1984). Smith (1984) proposed that if lianas are allowed to establish themselves in a young stand, the potential for yield of quality timber products is severely reduced. It may therefore be pragmatic to thin lianas in young (10–15 years), closed canopy forests that are poised to resist reinvasion—owing to the

inability of many lianas to climb adult trees without large canopy openings (Trimble & Tryon 1979).

Prior research reporting on the impacts of lianas on carbon storage has focused on the neotropics (van der Heijden et al. 2013; Schnitzer et al. 2014; Finlayson et al. 2022), and we are unaware of such studies in temperate forests. However, unlike tropical forests, trees in temperate forests including Appalachia produce annual growth rings: reliable indicators of tree age that allow us to quantify relationships between liana presence and tree growth rate over many decades with one season of data collection. In this way, Matthews et al. (2016) found reduced growth rates of temperate tree species when lianas were present in tree crowns compared to trees without lianas but did not link these impacts to carbon. Here we present the first quantification of liana impacts on tree CO₂ removal rates in a temperate forest and the first report linking species-specific impacts of lianas to forest restoration and climate mitigation.

Highlighting Temperate Lianas

Several groups of lianas in Eastern North American have the potential to impact adult trees due to their ability to either climb adult trees or attach themselves when trees are younger and grow with the tree, combined with their large adult size: wild grapes (*Vitis* spp., Vitaceae), Virginia creeper (*Parthenocissus quinquefolia* [L.] Planch., Vitaceae) Dutchman's pipe (*Isotrema macrophyllum* Lam., Aristolochiaceae), and nonnative lianas English ivy (*Hedera helix* L., Araliaceae), Oriental bittersweet (*Celastrus orbiculatus* Thunb., Celastraceae), Japanese honeysuckle (*Lonicera japonica* Thunb., Caprifoliaceae), and kudzu (*Pueraria montana* [Loureiro] Merritt, Fabaceae) (Lutz 1943; Ladwig & Meiners 2009; Weakley et al. 2012). We focus on wild grapes due to their prevalence throughout the study region and the hypothesized ease of treatment when compared to other lianas.

Wild grapes are ubiquitous in North America and are represented by many native species and a few introduced species (USDA-NRCS 2021) that provide substantial food and nesting habitat to varied wildlife (Martin et al. 1961; Della-Bianca 1979; Sanderson et al. 1980). Based on our observations that members of *Vitis* spp. have a consistent growth form, as well as the difficulty of differentiating species from the ground, we did not identify wild grapes to the species level.

Wild grapes attach to young trees using tendrils and are subsequently carried into the canopy as the tree grows (Darwin 1865; Trimble & Tryon 1979; Kadir 2006). They are shade-intolerant and can establish only in areas with ample sunlight, such as canopy gaps, and are therefore abundant in disturbed forest stands such as those subjected to selective timber harvest operations common throughout central Appalachia (Trimble & Tryon 1979; Putz 1984; Schnitzer et al. 2021). This growth strategy is important as it renders wild grape unable to reinvade mature trees in closed-canopy forests after removal treatments.

Most lianas—including wild grapes—have a main stem that roots it into the ground, near its host (Darwin 1865; Della-Bianca 1978; Keller 2020). Previous research concludes that cutting a wild grape near the ground severs its “lifeline” and causes mortality (Smith & McCay 1979; Trimble & Tryon 1979;

Smith 1984). Given the impacts of lianas on host trees reviewed here, thinning wild grapes—with safeguards in place to avoid perverse impacts on biodiversity, food availability, and habitat—may provide significant and long-lasting benefits to the growth of the host trees, bolstering both economic and climate benefits with minimal effort.

Goals and Predictions

For many decades researchers have been documenting the negative impacts of lianas on tree growth and calling for the cutting of lianas and other climbers as a simple means of increasing the productivity of forest stands (Featherly 1941; Trimble & Tryon 1974; Smith & Smithson 1975; Smith & McCay 1979; Putz 1984; Smith 1984; McNab & Meeker 1987; Schnitzer et al. 2000; Griscom & Ashton 2003; Schnitzer et al. 2014). While the cost of thinning lianas has rarely been justified given the delayed financial returns and the discounted rate of money, the emergence of a value for carbon storage could shift this silvicultural practice from being rare to common. Our goal was to estimate additional CO₂ removal by trees in the absence of common lianas (*Vitis* spp.) in central Appalachian forests. We predicted that trees hosting lianas experience significantly slower growth over their lifetime, and hence lower CO₂ removal rates than control trees of the same species growing in similar habitat but not hosting lianas, with increased impacts for shade-intolerant tree species.

Methods

Survey Methods and Data Collection

We collected field data across 97.12 ha of temperate forest, represented by four 24.28 ha forested parcels that were sampled between January and June 2021. All parcels were in the ridge and valley ecoregion of the eastern United States (USEPA

2013). We placed two of these 24.28-ha parcels (parcels 1 and 2) on private property and two within national forests (parcels 3 and 4) (Fig. 1). Parcels 3 and 4 were randomly placed within the ridge and valley ecoregion portions of the George Washington and Jefferson National Forests in northwestern Virginia, U.S.A., to provide locations that are distinct in geography and tenure (public vs. private), while remaining in the same ecoregion (Fig. 1).

Parcel 1 was subject to a typical regional form of timber harvest in 2006: diameter limit harvest of trees ≥ 30 cm diameter at breast height (dbh) (Fig. 1). Several stands at parcel 1 have not been harvested for longer time periods due to access constraints. Parcel 2 has not been harvested since approximately 1960 and is interspersed with a few stands that regenerated from fields about 30 years prior to our survey (Fig. 1). Elevation, precipitation, temperature, and soil types for all parcels are described in Table 1.

Within each parcel (1–4), we established 10–15 circular plots with 15-m radii in random locations using ArcGIS. Plots are centered on the nearest tree (>20 cm dbh) to their respective GPS point. We utilized a paired tree-based approach for this study, where each tree hosting a liana received a unique tag number (i.e. each host tree represented a sample size of $n = 1$). We collected data on any tree >20 cm dbh that was supporting a liana >1 cm dbh (host trees) as well as their paired control tree: the closest liana-free individual (which may occur outside of the plot) that is of the same species and of similar size (within 10 cm dbh).

We limited our analyses to species for which we were able to document at least three pairs (i.e. three host trees paired with controls), allowing a minimum level of replication. In order of increasing shade tolerance scores derived from Lopez et al. (2008), those species were *Liriodendron tulipifera*, *Betula lenta* L., *Quercus rubra*, *Quercus montana* Willd., *Fagus grandifolia* Ehrhart, *Acer saccharum*, and *Tsuga canadensis* L.

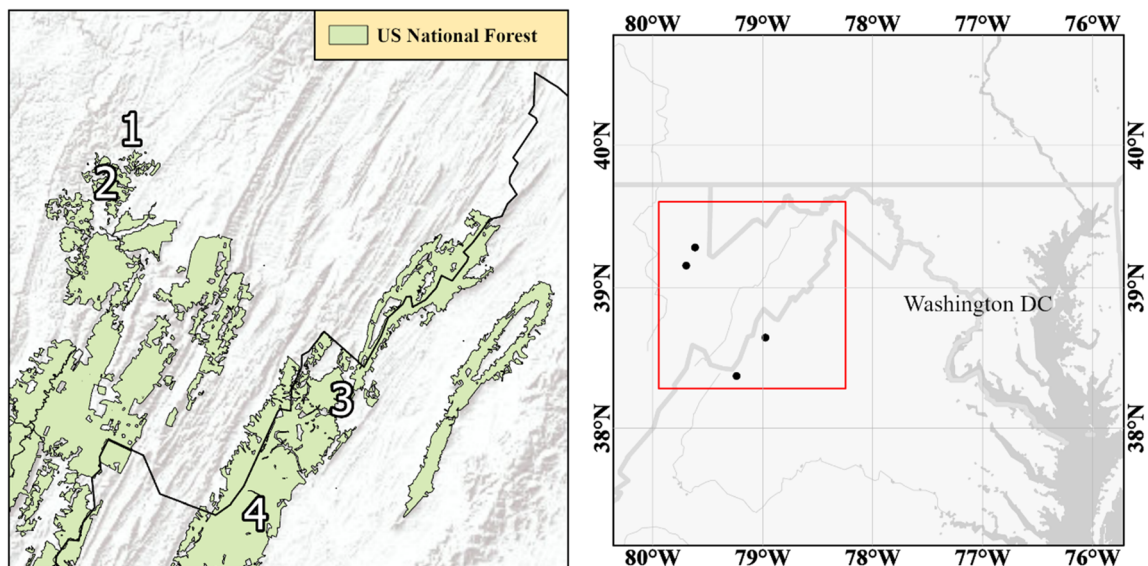


Figure 1. Study area map displaying the location of our four 24.28-ha (60 acres) parcels. Parcels 1 and 2 are on private property in West Virginia, U.S.A. Parcels 3 and 4 are in the George Washington and Jefferson National Forests in Virginia, U.S.A. Each parcel was surveyed using 10–15 randomly placed circular plots (15-m radii).

Table 1. Site conditions for the four parcels surveyed. Temperature data represent the 30-year average of monthly mean temperature obtained from PRISM. Soil types include inceptisols (I: stony, moist soils) and ultisols (U: silt loam). * denotes parcels located on private property in West Virginia.

Parcel	Elevation (m)	Precipitation (cm/year)	Temperature (°C)	Soil Type
1*	650–850	131	0.7	I, U
2*	450–550	137	1.2	I, U
3	460–580	97	1.6	I
4	720–800	104	0.7	I

Table 2. Liana (*Vitis* spp.) extent rank scale.

Rank	Description
1	liana prevalence is minimal—a small liana is present on very few branches
2	liana is present and growing throughout less than half of canopy
3	liana perceived as larger (heavier branching, longer lianas) and is growing throughout at least half of canopy
4	liana perceived as very large (at least 5 cm dbh, heavily branched, long lianas) and is growing abundantly throughout most of the canopy/canopies of one or more trees
5	liana perceived as very large (at least 5 cm dbh, heavily branched, long lianas) and has high prevalence throughout the canopy. May be completely covering treetops and is prevalent throughout neighboring trees

We collected tree cores from all trees using a Haglof increment borer (Torsång, Sweden) and stored them in 11" paper straws. We measured dbh and calculated merchantable tree bole height to estimate tree volume. To measure merchantable bole height, we measured angles to the base and top of each tree bole as well as the distance from which those angles were measured (with a goal of approximately 20 m when permitted by vegetation and topography) using a Nikon Forestry Pro II Laser Rangefinder (Minato City, Tokyo, Japan). We sanded and mounted tree cores, then counted tree rings to estimate the age of each tree (Fritts 1976). We collected dbh and "extent of infestation" data on every liana attached to documented host trees. We defined

liana extent as the apparent canopy prevalence of an individual liana, rather than the perceived impact on the host tree, ranked on a scale of 1–5 (Table 2).

Statistical Methods

We performed all calculations and analyses in an R programming environment (R version 4.0.4). The merchantable volume of each tree bole was estimated in cubic meters (m³) with the equation:

$$V = \pi * R^2 * MTH$$

where V represents the volume of the main bole (m³), R is the radius of the tree (m), and MTH is the merchantable tree bole height (m). Volume of the main bole was not used to estimate carbon but was used to estimate economic impacts. We estimated aboveground biomass in kilograms using biomass equations from Chojnacky et al. (2014):

$$AGB = e^{(\beta_0 + \beta_1 \ln [dbh])}$$

where AGB represents the total estimated biomass of the tree (kg), β_0 and β_2 are parameters from species-specific allometric equations from Chojnacky et al. (2014), and dbh is the tree's diameter at breast height. Belowground biomass was estimated with allometric equations from Chojnacky et al. (2014) to estimate root biomass as a function of dbh :

$$BGB = e^{-1.4485 + -0.03476 \times \ln (dbh)} + e^{-1.8629 + -0.77534 \times \ln (dbh)}$$

where BGB represents the belowground biomass. Carbon dioxide removal was subsequently estimated using the equation:

$$CDR = (AGB + BGB) \times 0.5 \times 3.67$$

where CDR represents the estimated CO₂ removed by the tree (kg), AGB is the total aboveground biomass (kg), and BGB is the estimated belowground biomass (kg). The multiplier 0.5 is the elemental ratio of carbon in tree biomass (Leith 1963; Whitaker et al. 1973; McPherson et al. 2016). The multiplier 3.67 is the molecular weight of CO₂, used to convert carbon stored to

Table 3. Census data, total biomass, and merchantable bole volume for all individuals (liana hosts and control trees combined). dbh = diameter breast height. Note that n represents the number of pairs of hosts and control trees for that species. Values represent the median, with the interquartile range listed in parentheses.

Species	n (Pairs)	dbh (cm)	Height (m)	Age (Years)	Total Biomass (kg)	Merchantable Bole Volume (m ³)
<i>Acer saccharum</i>	6	22.4 (20.6–24.1)	10.0 (8.8–12.7)	57 (55–65)	275 (226–328)	0.36 (0.29–0.60)
<i>Betula lenta</i>	4	31.7 (24.3–37.9)	15.7 (12.7–17.3)	57 (54–74)	663 (340–1,091)	1.11 (0.71–2.19)
<i>Fagus grandifolia</i>	3	30.7 (28.3–33.3)	12.3 (11.9–13.1)	86 (74–98)	542 (441–655)	0.95 (0.83–1.06)
<i>Liriodendron tulipifera</i>	8	43.9 (26.9–46.4)	21.5 (19.0–23.5)	53 (45–57)	1,002 (293–1,153)	3.24 (1.09–3.9)
<i>Quercus montana</i>	7	31.0 (23.2–38.0)	12.4 (9.94–13.1)	89 (46–112)	550 (273–909)	0.87 (0.41–1.90)
<i>Quercus rubra</i>	11	27.4 (24.8–33.8)	14.0 (12.0–16.3)	47 (41–52)	408 (321–681)	0.80 (0.62–1.12)
<i>Tsuga canadensis</i>	5	42.5 (40.7–54.0)	14.9 (11.7–20.1)	95 (75–103)	737 (664–1,313)	1.96 (1.70–3.11)

Table 4. Comparison of liana hosts' height growth, merchantable volume growth, and CO₂ removal rates to their liana-free neighbors, stratified by shade tolerance cohorts. Values represent the median, with the interquartile range listed in parentheses. Note that *n* represents the number of pairs of hosts and control trees for that group. Significance is denoted with * and ** for $p < 0.05$ and $p < 0.01$, respectively.

	<i>n</i> (pairs)	Height Growth Rate (m/year)	Merchantable Volume Growth Rate (m ³ /year)	CO ₂ Removal Rate (kg/year)
All species	44	−0.038 (−0.080 to 0.009)**	−0.0016 (−0.0061 to 0.0015)*	−1.03 (−3.65 to 1.41)
Shade tolerant	21	−0.025 (−0.056 to 0.027)	−0.0005 (−0.0049 to 0.0012)	0.04 (−2.37 to 1.20)
Shade intolerant	23	−0.050 (−0.107 to 0.0004)**	−0.0040 (−0.0091 to 0.0014)*	−2.58 ± 7.22 (−6.13 to 1.46)*

estimated CO₂ removed (McPherson et al. 2016). Growth, carbon storage, and CO₂ removal rates are determined by dividing each variable by the tree's age.

For analyses, we focused primarily on comparing specific pairs (i.e. host trees are compared directly to their paired control tree). We conducted Shapiro–Wilk normality tests for all data and subsequent paired Wilcoxon rank sum tests to assess differences in all variables including CO₂ removal rates between treatments (host trees and their control trees). Finally, we conducted

linear regressions to compare CO₂ removal rates with tree biomass, stratified by shade tolerance and liana presence.

Results

Summary of Survey Data

Across 97.12 ha of temperate forest, we established 56 plots for a total subsampled area of 3.96 ha. Of these plots, only

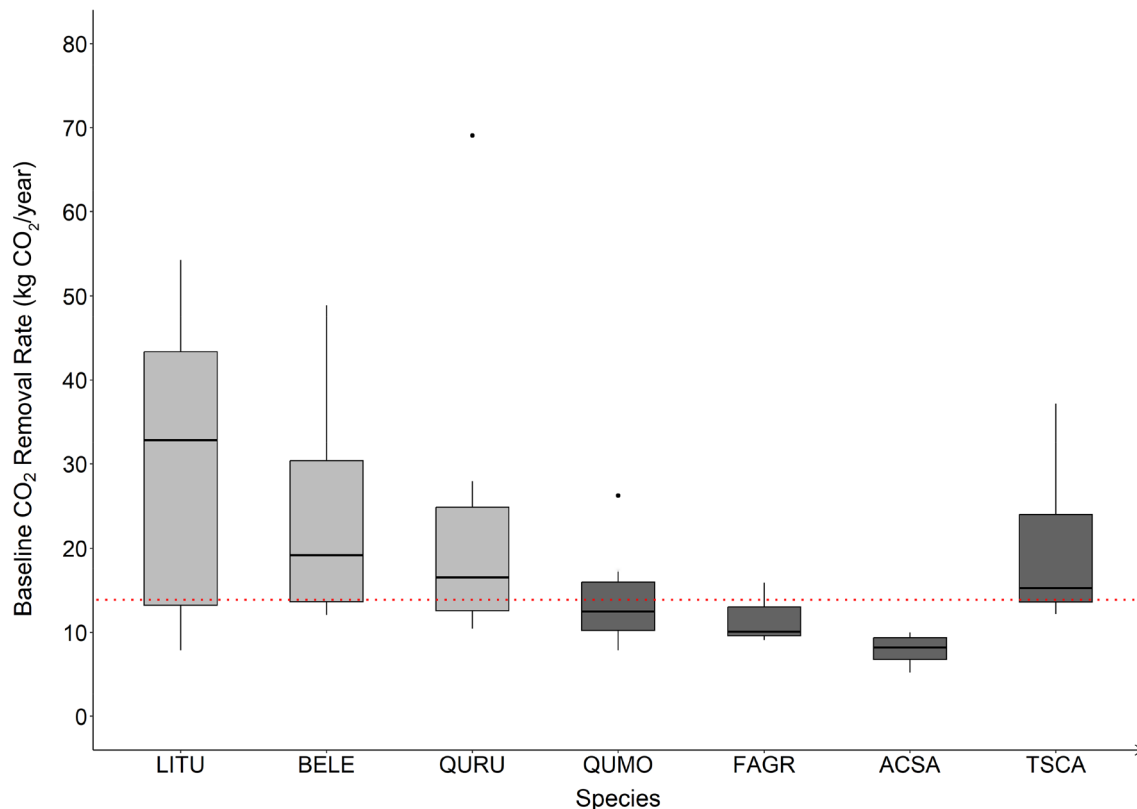


Figure 2. Baseline atmospheric carbon dioxide removal rates (kilograms of carbon dioxide per year [kg CO₂/year]) for each of the seven focal species. Data shown are from liana-free control trees only. Species are arranged by order of increasing shade tolerance (left to right, respectively). Lightly and darkly shaded boxes represent shade-intolerant species and shade-tolerant species, respectively. Dashed red line represents the combined median (13.85 kg CO₂/year). Black dots represent values more than 1.5 times the interquartile range over the 75th percentile. The bold horizontal line represents the median for each species (including outliers), the boxes represent the 25th and 75th quartiles, and the vertical lines extend to the minimum and maximum values (excluding outliers). Species codes are as follows: LITU (*Liriodendron tulipifera*), BELE (*Betula lenta*), QURU (*Quercus rubra*), QUMO (*Quercus montana*), FAGR (*Fagus grandifolia*), ACSA (*Acer saccharum*), and TSQA (*Tsuga canadensis*). Note the comparison of each species' median CO₂ removal rate to the baseline.

14 contained host trees. Liana abundance was found to be highly variable, with many plots having no trees hosting lianas, some plots having as few as two, and several plots with almost a dozen. We surveyed a total of 88 trees (44 trees hosting lianas) of the common species specified above (Table 3).

Among all trees >20 cm dbh, the median dbh was 29.5 cm, and the maximum was 63.5 cm (Table 3). The mean difference in dbh between hosts and their paired control trees was 0.8 (± 3.5) cm and we found no statistical difference in size between hosts and their paired controls ($p = 0.098$). Thus, control trees were successfully and consistently selected to be within the same size class as their paired host tree. Host trees were significantly older than their paired control trees ($p < 0.005$), with a mean age difference of 9 (± 16) years. The median volumetric growth rate of merchantable boles was 0.016 m³/year among all trees (Table 4). The median diameter of lianas surveyed (above >1 cm dbh) was 4.7 cm with an interquartile range (IQR) of 3.5–5.4, and largest liana we encountered was 11.9 cm dbh. The median liana extent (estimated between 1 and 5 via our extent ranking protocol) was 3.5 (Table 2).

Liana Impacts on Tree Growth Rates and Carbon Storage

Here we compare differences in merchantable growth, height, overall tree growth, and carbon dioxide removal between pairs of trees of the same species and size class (i.e. liana hosts and their paired liana-free neighbors). Across all tree species, liana hosts' merchantable bole growth was 9.7% slower for trees hosting lianas. We found that lianas (*Vitis* spp.) significantly reduced annual volumetric growth of the merchantable bole, with a median difference of -0.0016 m³/year (IQR = -0.0061 to 0.0015) for trees hosting lianas ($p = 0.031$). We did not detect a statistically significant difference in the heights of merchantable boles ($p = 0.258$) or in the ratio of bole height to dbh ($p = 0.087$). However, we observed a trend of shorter boles in host trees compared to their controls, with a median difference of -1.07 m (IQR = -2.40 to 2.00). Additionally, the rate of height growth of merchantable boles was significantly lower for trees hosting lianas ($p = 0.003$; Table 4). The median diameter growth rate was 0.51 cm/year (IQR = 0.39 – 0.65), and liana presence significantly correlates with the annual diameter growth rate of all trees: we observed a difference in diametric growth of -0.057 cm/year (IQR = -0.12 to 0.01) for trees hosting lianas ($p = 0.002$). Finally, the median atmospheric carbon dioxide removal rate by liana-free control trees (baseline) was 13.85 kg CO₂/year (IQR = 10.36 – 24.57), and we observed a median difference in CO₂ removal rates of -1.03 kg CO₂/year (IQR = -3.65 to 1.41) for trees hosting lianas, but this was just above a $p < 0.05$ significance threshold ($p = 0.073$; Table 4; Fig. 2). While the presence of lianas appears to significantly reduce CO₂ removal for shade-intolerant tree species ($p = 0.025$), we did not detect significant relationships between our estimated liana extent levels and CO₂ removal rates ($p = 0.235$; Table 4).

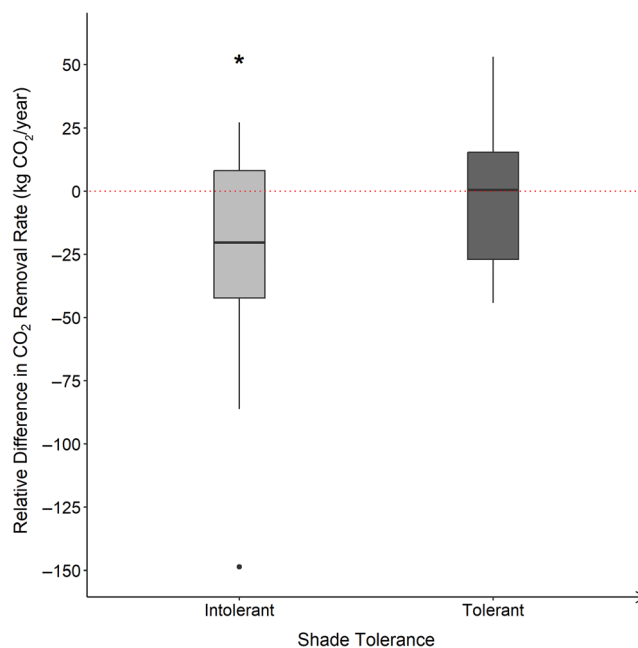


Figure 3. Relative difference in atmospheric carbon dioxide removal rates (kilograms of carbon dioxide per year [kg CO₂/year]) of lianas hosts and their paired neighbors, separated by their shade-tolerance designations. Lightly and darkly shaded boxes represent liana-free controls and liana hosts, respectively. Dashed red line represents the point of no difference (0). Black dots represent values more than 1.5 times the interquartile range over the 75th percentile. The bold horizontal line represents the median for each group (including outliers), the boxes represent the 25th and 75th quartiles, and the vertical lines extend to the minimum and maximum values (excluding outliers). Statistically significant differences are noted above boxes with an asterisk (*). Note the significant difference in CO₂ removal rates for shade-intolerant species hosts and controls.

We found atmospheric CO₂ removal rates to be negatively correlated with shade tolerance ($p < 0.005$; Figs. 2 & 3). For shade-tolerant species, the median difference in CO₂ removal rates of liana host trees versus their paired controls was 0.044 kg CO₂/year (IQR = -2.37 to 1.20), representing a slight (0.5%) increase ($n = 21$ pairs; $p = 0.838$; Fig. 3). However, for shade-intolerant tree species, the median difference in CO₂ removal rates of liana host trees versus their paired controls was -2.58 kg CO₂/year (IQR = -6.13 to 1.46), representing a 20% reduction ($n = 23$ pairs; $p = 0.025$; Fig. 3).

We also compared CO₂ removal rates (kg CO₂/year) with tree biomass (kg) for liana hosts and liana-free control trees among shade-tolerant and shade-intolerant groups using linear regression models (Fig. 4). Biomass increment and CO₂ removal rates are correlated with tree biomass ($p < 0.005$, $R^2 = 0.57$; Fig. 4). We also identified significant overlap between the 95% confidence intervals (CIs) of our regression models for shade-tolerant species (Fig. 4). Conversely, there was little overlap between confidence intervals and a visible distinction between regression lines of shade-intolerant liana

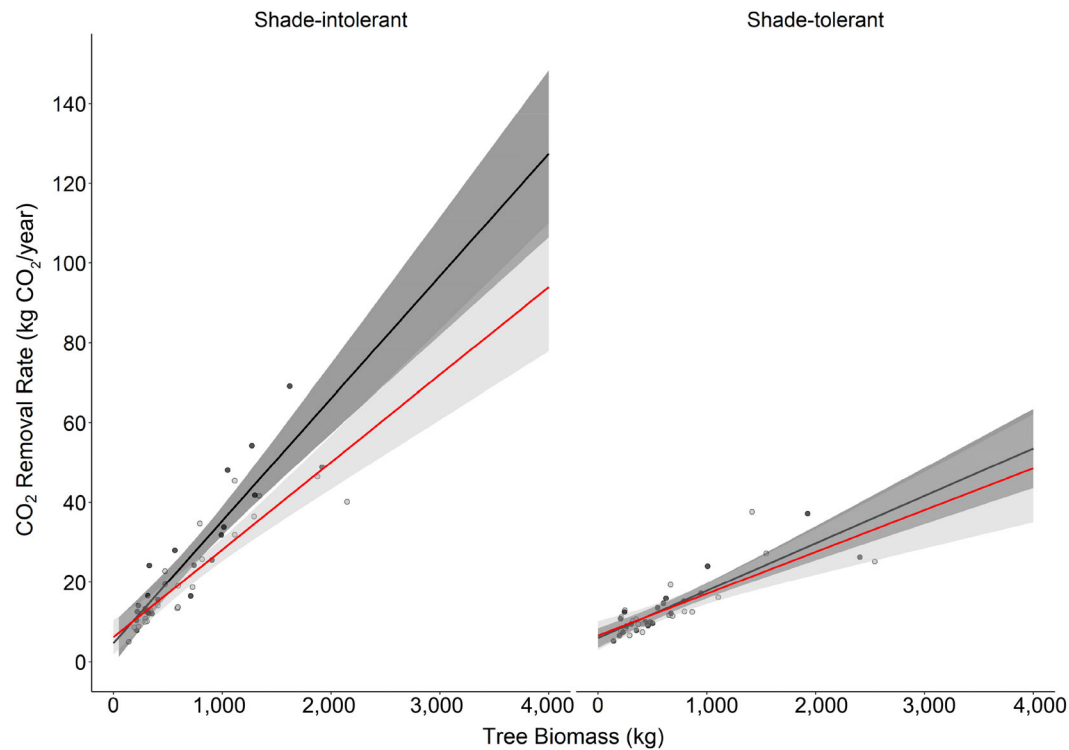


Figure 4. Carbon dioxide storage rates ($\text{kg CO}_2/\text{year}$) versus tree biomass (kg). Dark and light points represent liana-free hosts and controls, respectively. Black and red regression lines represent linear regressions for liana-free controls and liana hosts, respectively. Darkly shaded and lightly shaded area represents 95% CI for regression models of liana-free controls and liana hosts, respectively. Left and right side of plot isolates shade-intolerant and shade-tolerant species, respectively. Note the area of overlap in regressions and their 95% CI per shade tolerance.

hosts compared to their liana-free controls when plotted against their biomass. When lianas are present, light-demanding tree species experience significant reductions in their ability to remove carbon dioxide from the atmosphere (Fig. 4).

Discussion

Lianas (*Vitis* spp.) appear to impact fast growing shade-intolerant tree species more significantly than slow growing shade-tolerant species. This finding aligns with Visser et al. (2018), whose results indicate that tropical liana impacts depend on the host's ability to tolerate shade. We therefore suggest that thinning lianas from shade-intolerant tree species could maximize benefits per unit effort while providing a necessary safeguard for biodiversity. However, further research linking climate and carbon dioxide removal to host species-specific liana impacts is needed, as we represent the first study to present this linkage.

Tree growth and atmospheric CO_2 removal rates were higher for trees without lianas, although these trees tended to be younger than their similar-size neighbors and thus were initially at a disadvantage (lower position in the canopy). In other words, liana free trees appear to frequently overtake liana hosts that may be as much as a decade older. This result is particularly significant when considering that larger and older trees are known to store carbon at increased rates compared to smaller and younger trees and are more affected by vine

loading in tree canopies (Stephenson et al. 2014; Matthews et al. 2016). Hence, we suspect that a comparison of trees of matching ages (rather than matching sizes) would identify a larger impact on growth rates due to liana loading.

Regardless of these results, however, we must consider that most species of wild grape are native to North America and provide structural habitat for tree-nesting species, as well as food for a variety of wildlife species (Martin et al. 1961; Sanderson et al. 1980). We must take measures to avoid eradicating all lianas—however, many liana species have become overly abundant in disturbed forests, encouraged by human actions such as the creation of forest edges and repeated disturbance (Schnitzer et al. 2021). In temperate Mid-Atlantic forests, Matthews et al. (2016) found that native lianas were more abundant than exotic except on forest edges. We recommend that liana thinning efforts be adjusted as a function of host tree species, land manager goals, and the needs of local people and indigenous communities. For example, thinning either shade-intolerant species such as *Liriodendron tulipifera* to maximize carbon benefits or *Quercus* spp. to maximize timber value. Similarly, treating a subset of lianas on species more impacted by them will maximize climate and timber value benefits per unit effort and avoid excessive removal of lianas and the ecosystem services they provide.

Landscape context may also be important for identifying abundant liana loading, where treatments will be most effective.

It has been documented for decades that lianas prefer high-light levels such as those that exist in canopy gaps and edges (Putz 1984; Schnitzer 2005; Schnitzer et al. 2021). Currently, we are unaware of large-scale liana thinning efforts. Additionally, we propose that mapping abundant lianas in temperate forests is a logical next step, and that focusing liana thinning efforts in disturbed areas such as edges could reduce time and cost of treatment efforts.

Liana host trees in our study tended to have main boles that were shorter, grew significantly slower in height, and added volume at significantly lower rates than their liana-free neighbors. We also expect that lianas are reducing the true height to the top of the canopy. This is possible via liana colonization of regenerating canopy gaps (Schnitzer et al. 2000; Griscom & Ashton 2003), or through mechanical damage caused to tree canopies via liana loading (Siccama et al. 1976; Trimble & Tryon 1979; Smith 1984). These findings indicate that forest stands in central Appalachia will have significantly greater economic value in the decades following liana thinning.

We consider here as examples two common timber species, *Quercus rubra* and *L. tulipifera*. Annual volumetric merchantable bole growth was 4.38% higher for liana-free *Q. rubra*, and 21.68% higher for liana-free *L. tulipifera*. In Appalachia, *Q. rubra* stumpage per 1,000 board feet is worth \$210 and *L. tulipifera* is worth \$200 per 1,000 board feet (Timber Update 2021). Our results indicate that liana removal for a single *Q. rubra* containing 1,000 board feet could increase average annual stumpage value by \$9/year, while it would increase the equivalent value of a single *L. tulipifera* by as much as \$40/year. More work is needed to estimate potential profit from liana treatments over time, considering the discount rate of money and the upfront cost of liana removal. Additionally, carbon financing may offer a solution via providing upfront income for investing in stand improvements, rather than waiting for a return on investment solely through timber harvest.

A potential issue with liana removal is that perhaps we are removing another source of carbon sequestration. Several lianas in this study were greater than 10 cm diameter and spanning several trees. Although we did not attempt to estimate carbon stores in lianas in this study, it is known that, as structural parasites, lianas do not allocate as many resources to structural support compared to trees, and it is unlikely that the carbon stored in lianas equates to the carbon that their host tree would have stored in their absence (Schnitzer et al. 2014). Therefore, we suspect that lianas are less efficient at removing CO₂ per unit of growing space they occupy compared to trees. Finlayson et al. (2022) found large net benefits in carbon removal by trees following liana removal in tropical regions, after accounting for emissions due to decay of cut lianas. Such net CO₂ flux calculations are an important next step for research in Appalachian liana management.

Some studies have been able to reasonably assess the percent of crown cover by lianas from the forest floor using binoculars (Ingwell et al. 2010; Visser et al. 2018). However, we did not find this to be easy or accurate with a relatively small sample size, as we did not detect a trend in CO₂ removal rates as a function of liana extent as estimated from below. It is possible that with continued sampling this result may change. Furthermore, research has demonstrated that smaller-stemmed lianas can have sprawling crowns

situated over the host tree's canopy (Tobin et al. 2012). We propose that rather than judging liana infestation by stem size or "extent" estimated roughly from the ground, crown cover measured empirically from above using aerial imagery (i.e. images collected from drones or airplanes) may be useful.

Here we began to explore a research gap for temperate lianas while offering strategies for safeguarded liana removal. Additional work on mapping abundant lianas, estimating the climate impact of releasing temperate forests from lianas, and the differential impacts of lianas on host species in a broader geographical context is needed to elaborate on these findings for temperate forests of North America. Our results indicate that thinning lianas could deliver a cost-effective climate solution and forest restoration strategy, but that this practice should be explored as a function of tree shade tolerance and landscape characteristics that may influence the sensitivity of tree species to liana loading. Liana management offers a compelling opportunity across temperate forests to reverse chronic forest degradation while enhancing carbon dioxide removal and improving rural economies to advance towards our shared global climate and sustainable development goals.

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