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LETTER

Thermal stress in degraded forests in the Brazilian Amazon Arc of Deforestation

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Abstract

Understanding thermal stress in tropical forests has taken on new urgency in light of accelerating climate change and expansion of deforestation and forest degradation. Degraded tropical forests in particular may be approaching critical temperature thresholds even more rapidly than intact forests, with implications for tree survival and ecosystem recovery. We investigate thermal stress in degraded tropical forests within the Brazilian Amazon Arc of Deforestation. Using land surface temperature data from the ECOsystem Spaceborne Thermal Radiometer Experiment on the international Space Station (ECOSTRESS), we compared canopy temperatures of intact, selectively logged, and burned forests in Feliz Natal, Mato Grosso, Brazil. Upper canopy temperatures in previously burned forests were 4.1% higher (mean = 36.5 °C) and 50.9% more variable compared to intact and logged forests, which showed remarkably similar temperature distributions (means of 34.9 °C and 35.1 °C, respectively). Modeled leaf temperature distributions based on canopy temperature measurements from one of the warmest days in a 2 year dry season record indicated that 87% of leaves in the warmest burned forest patches exceeded the temperature threshold where respiration surpasses photosynthesis, compared to approximately 72%–74% in intact and logged forests, respectively, in the same time period. After controlling for environmental factors, burned forests were predicted to be 2.6 °C warmer on average [1.39 °C–3.96 °C, 95% credible interval] than intact forests across a 5–40 m canopy height range. Burned forests showed modest thermal recovery over time, with temperatures decreasing by approximately 1.2 °C over a 30 year recovery period. In contrast, logged forests showed minimal thermal differences from intact forests (−0.353 on average, [−1.125–0.274, 95% credible interval]) and negligible change (0.15 °C) over the same timeframe. While the absolute probabilities of exceeding damaging thermal thresholds remain low across all forest types under current climate conditions, the probability of leaves exceeding temperatures that cause permanent leaf damage was ten times higher in burned forests, with implications for the future of burned forest regeneration in water-limited regions of the Amazon basin. In particular, the combination of higher mean temperatures, greater temperature variability, and more frequent exposure to damaging thermal thresholds implies that burned tropical forests will experience substantially higher mortality rates and slower biomass recovery compared to intact and selectively logged forests, especially in water-limited regions where trees cannot rely on evaporative cooling to moderate canopy temperatures.

1. Introduction

Tropical forest degradation has been associated with a greater loss of carbon in the Brazilian Amazon than through deforestation directly (Csillik *et al* 2024). This degradation occurs through multiple pathways including selective logging, understory fires, and edge effects from fragmentation, often acting in combination to reduce biomass and alter forest structure. While degraded forests retain significant conservation value (Chazdon 2014) and carbon stocks (Heinrich *et al* 2023), their modified canopy architecture, reduced leaf area, and simplified vertical structure make them more vulnerable to additional disturbances and environmental stresses. Critically, these structural changes affect the forest microclimate by increasing light penetration and reducing evaporative cooling capacity (Longo *et al* 2020). This increased vulnerability is concerning as climate change accelerates, because degraded forests may be less resilient to extreme temperatures and drought than their intact counterparts (McDowell *et al* 2020), potentially creating feedback loops that further compromise forest recovery (Anderson-Teixeira *et al* 2013). Understanding how degradation affects forest thermal regimes is therefore crucial for predicting the fate of these ecosystems and their capacity to maintain critical functions under future climate conditions.

Degraded forests experience higher surface and canopy temperatures compared to intact forests due to changes in forest structure, reduced evapotranspiration (Rangel Pinagé *et al* 2023a), and increased exposure of understory and soil (Longo *et al* 2020). Tropical forests may be approaching critical temperature thresholds that could lead to widespread tree mortality (Esquivel-Muelbert *et al* 2020) and ecosystem state shifts (Anderson-Teixeira *et al* 2013). In the Amazon, peak canopy temperatures during dry periods can exceed 40 °C, where there is often a long-tailed distribution of leaf temperatures with some leaves reaching temperatures near upper physiological thresholds (Doughty *et al* 2023). One such threshold is the temperature at which leaf respiration exceeds photosynthesis, typically in the range ∼30 °C–35 °C for most tropical tree species (Slot and Winter 2017). Other critical thresholds between 46.7 °C (Doughty *et al* 2023) and 49.9 °C or warmer (Winter 2024) reflect the temperatures at which physiological leaf functioning declines and leaf necrosis may occur (Winter 2024).

Recent research has demonstrated that thermal remote sensing data can detect changes in forest canopy temperature related to disturbance and stress at large spatial scales (Doughty *et al* 2023). The ecosystem spaceborne thermal radiometer experiment on the international space station (ECOSTRESS) provides 70 m spatial resolution thermal infrared observations that inform our view of degradation severity and recovery potential currently not captured by optical sensors. However, few studies have leveraged the wealth of available remotely sensed data for comprehensive assessments of thermal stress in degraded areas of the Amazon or other tropical forests (de Oliveira *et al* 2021, Doughty *et al* 2023, Rangel Pinagé *et al* 2023a).

ECOSTRESS land surface temperature (LST) measurements represent the integration of thermal emissions from surfaces visible to the sensor, which in forests are predominantly light-exposed leaves. Previous research demonstrated strong agreement between ECOSTRESS LST and tower-based pyrgeometer measurements ($r^2 = 0.75$) (Doughty *et al* 2023), both of which capture area-integrated thermal signals. Another study used high-resolution thermal imaging to reveal that canopy temperature distributions are typically right-skewed, with the mode and median temperatures representing the most common leaf temperatures in sun-exposed portions of the canopy (Still *et al* 2019). This finding aligns with (Pau *et al* 2018), who showed that thermal infrared emissions are dominated by leaves receiving direct solar radiation, making LST particularly representative of median temperatures in light-exposed canopy regions. The physical basis for using LST as an estimate of median leaf temperature is further supported by the Stefan–Boltzmann law: while extreme leaf temperatures contribute non-linearly to the integrated thermal signal, the large number of leaves at moderate temperatures means the bulk signal is dominated by the central tendency of the distribution. In tropical forests specifically, leaf temperature distributions have been shown to vary predictably with the light environment (Ray *et al* 2005), suggesting LST can effectively characterize the temperature regime experienced by light-exposed leaves across forest conditions.

The Brazilian Amazon, particularly the *Arc of Deforestation* (Fearnside 2017) along its southern and eastern edges, provides an ideal case study for examining patterns of thermal stress in degraded forests. This region has experienced extensive deforestation and degradation over recent decades, creating a mosaic of intact, burned, logged, and regenerating forests (Souza-Filho *et al* 2016, Csillik *et al* 2024). The region also experiences water limitation during normal dry season conditions (Guan *et al* 2015) and therefore is particularly prone to excess heat (Brodribb *et al* 2020).

We leveraged active as well as passive thermal and optical remote sensing to assess the thermal state of degraded forests under the current climate. Specifically for our study area, we asked two questions: How do logging and burning affect forest upper canopy and leaf temperatures as measured by ECOSTRESS thermal infrared data? And what is the proportion of upper canopy leaves that exceed critical temperature thresholds, estimated through a combination of ECOSTRESS thermal data and global ecosystems dynamics investigation (GEDI)-derived structural metrics? We expected burned and logged forests to exhibit higher temperatures than intact forests, due to differences in structure (Longo *et al* 2020). In particular, we expected that leaf temperatures would exceed physiological thresholds most frequently in burned forests and least frequently for intact forests.

2. Methods

2.1. Study area

Our study site, in the municipality of Feliz Natal, is located in the state of Mato Grosso and falls within the Brazilian Amazon Arc of Deforestation (Fearnside 2017) (figure 1). Feliz Natal contains a range of forest degradation types and intensities. It features land uses including pastures and mechanized agriculture that have replaced part of the original forest cover. Most of the remaining forests in this region that are outside of designated conservation areas or Indigenous territories have been logged and a substantial portion of the forests have burned at least once (Rappaport *et al* 2018).

We obtained land cover classification data from Rangel Pinagé *et al* (2023b), which includes intact, logged, and burned forest categories across our study area as well as a reference forest degradation dataset built from visual interpretation of Landsat TM, ETM + and OLI time-series, from 1984 to 2020 (see SI: Expanded Methods). The classification has an overall accuracy (i.e. the ratio of correctly classified pixels across all classes divided by the total number of samples) of 0.86 ± 0.1 (95% CI 0.85–0.87) among all sites and an overall accuracy of 0.69 (95% CI 0.65–0.72) for the Feliz Natal site specifically. The data from Rangel Pinagé *et al* (2023b) is based on 3 m resolution PlanetScope images acquired on 17 June 2018 with the output classification aggregated to ∼500 m resolution.

2.2. Canopy height

All available canopy height estimates were obtained from the GEDI Level 2a data product (Dubayah *et al* 2020) between 2018–2020. To minimize the effects of GEDI's geolocation error, we excluded all observations that fell within 10 m of a forest edge (Dubayah *et al* 2020, Cooley *et al* 2024). We also excluded any observations with a poor data quality flag. We selected canopy height as the variable to quantify the relationship between forest structure and upper canopy temperature to build on existing work showing the negative, non-linear relationships of height and aboveground biomass with canopy temperature (Longo *et al* 2020, Rangel Pinagé *et al* 2023a). We also examined how canopy temperature varied across other forest structure metrics, including the height of median energy (relative height 50) (figure S2). To address instances of potential land use change between the land cover classification date and GEDI acquisition date, we established minimum AGB thresholds for each degradation type based on a previous study by Rangel Pinagé *et al* (2023b) that estimated AGB distributions estimates derived from high point density airborne lidar data acquired in 2017–2018 over Feliz Natal (see SI Methods).

2.3. Upper canopy temperature

We obtained all available best quality, high accuracy LST observations from ECOSTRESS (Hulley *et al* 2022, Hulley and Freepartner 2018) for the dry season months (May–October) of 2018–2020 between 10:00–18:00 local time. ECOSTRESS is a thermal radiometer mounted on the International Space Station and thus has variable overpass times, with an average revisit period of 3–5 d over the Feliz Natal site. We masked the LST images using the ECOSTRESS QC bit flag (value for bits 1&0 = 00 indicating best quality) and high LST accuracy (bits 15 & 14 = 10 or 11 for accuracy of good or excellent performance). We applied the standard ECOSTRESS cloud mask. We selected the warmest

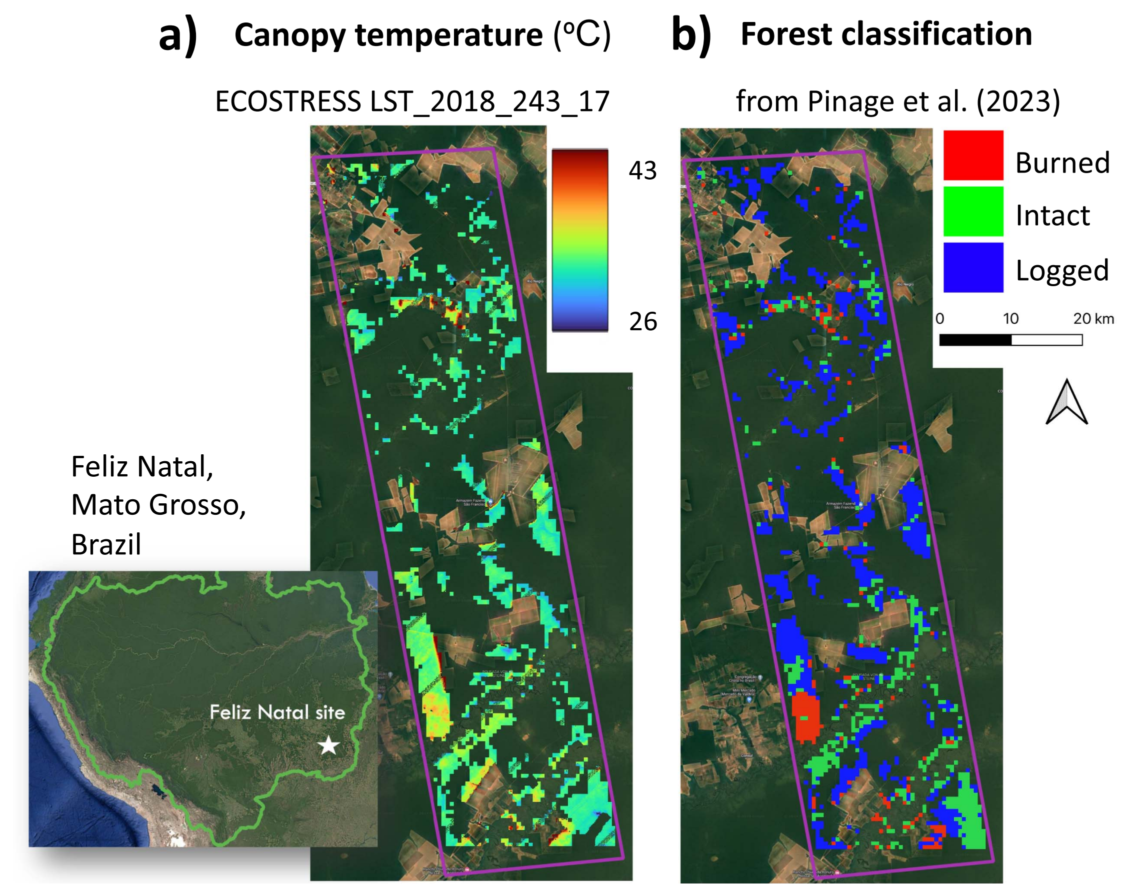

Figure 1. (a) Forest upper canopy temperature in Feliz Natal, MT study site for a single day of 31-08-2018 corresponding to the warmest, least cloud-contaminated ECOSTRESS acquisition available in the 2 year dry season record; and (b) classification of intact, logged and burned forest types from Rangel Pinagé *et al* (15). Non-forest pixels were removed.

and least cloud-contaminated LST observation on record during this time period for further analysis (31 August 2018 captured by ECOSTRESS at 17:38:03 UTC/13:38:03 local time). Leveraging the forest classification carried out by Rangel Pinagé *et al* (2023b), we spatially filtered the results to only include intact or degraded forest (selectively logged or burned within 34 years prior to the image date, noting that the degradation time series began in 1984 due to the Landsat historic record) (figure 1). Similar to (Doughty *et al* 2023), we assumed that each LST pixel corresponded to the mean upper canopy temperature value, as ECOSTRESS primarily measures thermal emissions from the uppermost light-exposed portions of the canopy. Our final data set consisted of all best quality ECOSTRESS observations that spatially intersected with the filtered GEDI footprints. Due to the difference in spatial resolution between ECOSTRESS LST (70 m) and GEDI (25 m diameter footprints), we assumed that each GEDI observation corresponded to the LST value of the ECOSTRESS pixel that each footprint centroid fell within. The final data set consisted of 6767 observations across three forest categories: logged (n = 3,997), intact (n = 1,859), and burned (n = 911) forests. logged forests had similar aboveground biomass (mean ± SD of 120 ± 72 Mg ha^{-1}) compared to intact forests (122 ± 72 Mg ha^{-1}) while burned forests showed notably lower biomass (102 ± 79 Mg ha^{-1}).

2.4. Modeling the influence of degradation on canopy temperature

We used a hierarchical Bayesian model with the following power law function to investigate the relation between degradation and upper canopy temperature:

$$\text{UpperCanopyTemp} = d * \text{CanopyHeight}^{z} \quad (1)$$

where UpperCanopyTemp is the upper canopy temperature [°C] and CanopyHeight is the canopy height from GEDI [m].

We allowed d and z to depend on degradation, bare ground fraction (22) and several best-performing, uncorrelated and ecologically meaningful fixed-effect variables identified from a random forest variable importance measure (figure S3). We tested eight model specifications, all using 20 evenly-spaced spatial quadrats (table S1). The model comparison results revealed that simpler models with fewer parameters performed better overall according to the deviance information criterion (table S2), with the best-performing model representing the spatially-explicit relationships of parameters d and z as follows:

$$\mathrm{d}[q] \sim A_0 + A_1 * \mathrm{Burned}[q] + A_2 * \mathrm{Logged}[q] + A_3 * \mathrm{AirTemp}[q] + A_4 * \mathrm{Burned}[q] * \mathrm{TimeSinceDis}[q] + A_5 * \mathrm{Logged}[q] * \mathrm{TimeSinceDist}[q] + \varepsilon \quad (2)$$

$$z[q] \sim B_0 + B_1 * \mathrm{Burned}[q] + B_2 * \mathrm{Logged}[q] + B_3 * \mathrm{Elevation}[q] + B_4 * \mathrm{Burned}[q] * \mathrm{TimeSinceDis}[q] + B_5 * \mathrm{Logged}[q] * \mathrm{TimeSinceDist}[q] + \varepsilon \quad (3)$$

where q is the index for the spatial quadrat;

A_0, B_0 are the intercepts for parameters d and z, respectively;

A_1, B_1 are the hyperparameters that quantify the normalized influence of burned forest on d and z, respectively;

A_2, B_2 are the hyperparameters that quantify the normalized influence of logged forest on d and z, respectively;

A_3, B_3 are the hyperparameters that quantify the normalized influence of air temperature at center height of lowest atmospheric model layer (2 m) on d and elevation on z, respectively;

A_4, B_4 are the hyperparameters that quantify the normalized influence of the interaction between burned forest and time since disturbance on d and z, respectively;

A_5, B_5 are the hyperparameters that quantify the normalized influence of the interaction between logged forest and time since disturbance on d and z, respectively;

and ε is unexplained residual variation.

We implemented the model using Just Another Gibbs Sampler (JAGS) with Markov Chain Monte Carlo simulation (Plummer *et al* 2024) with three chains, each with 15 000 iterations of burn-in and 40 000 iterations of sampling.

2.5. Leaf temperature distributions

To model sub-pixel leaf temperature distributions, we obtained canopy top leaf thermocouple data aggregated by Doughty *et al* (2023) from 11 individual leaves from a single site in the Amazon over 54 sunny periods lasting 20 min (figure S4). We used an implementation of JAGS with MCMC simulation (Plummer *et al* 2024) to parameterize a gamma distribution of upper canopy leaf temperatures for the warmest forest patches—quantified by the top quartile of ECOSTRESS LST pixels—for each forest type among intact, selectively logged and burned types. We chose a gamma distribution because it accommodates a long tail of extreme high values as identified by previous work (Pau *et al* 2018, Doughty *et al* 2023). We assumed that the observed temperatures in the warmest forest patches of each forest type represented the mean value of the upper canopy leaf temperature distribution (Pau *et al* 2018, Still *et al* 2019). We modeled the standard deviation of the estimated leaf temperature distribution as a function of the mean temperature for each forest degradation class assuming a linear relationship between mean and standard deviation based on the thermocouple data.

We then computed the probabilities that individual leaves within the warmest forest patches (i.e. the top quartile pixels of intact, logged and burned forest) would exceed three critical temperatures thresholds, including the temperature at which:

(1) respiration begins to exceed photosynthesis ($T_{AR} = 33.0$ °C) from (Slot and Winter 2017),
(2) the ratio of variable chlorophyll-a fluorescence yield to maximum fluorescence yield (F_v/F_m) starts to decline by 5% ($T_5 = 46.7$ °C) from (Doughty *et al* 2023), and
(3) the ratio F_v/F_m declines by 50% ($T_{50} = 49.9$ °C) from (Winter 2024).

The T_5 threshold indicates the beginning of leaf damage; whereas surpassing the T_{50} threshold is likely to cause irreversible leaf damage, resulting in necrosis (Winter 2024). Note that these temperature thresholds are not universal, and vary across taxa (Slot and Winter 2017) and acclimation conditions (Smith and Dukes 2017). Nonetheless, they signal the increased risk of widespread and irreversible damage in forest canopies.

3. Results

3.1. ECOSTRESS and GEDI observations

Upper canopy temperatures in burned forests (mean = 36.5 °C, median = 36.5 °C, standard deviation = 1.84 °C) were 4.1% higher and 50.9% more variable on average relative to selectively logged and intact forests (table 1, figures 2 and S1). Selectively logged and intact forests exhibited remarkably similar means and variability in their canopy temperature distributions (compare respective mean, median and standard deviations of 35.1 °C, 35.0 °C, and 0.93 °C

Table 1. Summary statistics of the canopy temperatures of the warmest and least cloudy day in the 2-year record (31-08-2018) among intact, logged and burned forests. The statistics correspond to the image in figure 1(b).

Forest type	Mean canopy temp (°C)	Standard deviation (°C)	75th percentile (°C)
Intact	34.9	0.87	35.2
Logged	35.1	0.91	35.4
Burned	36.5	1.81	37.3

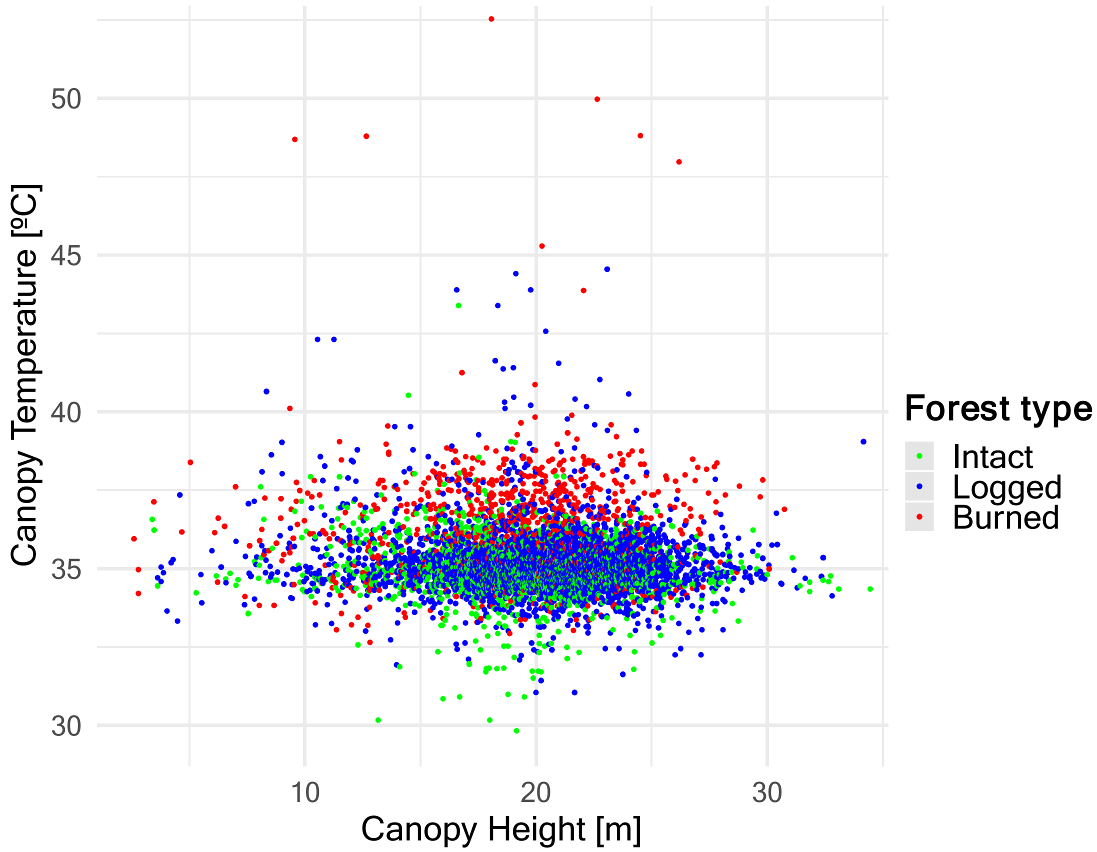

Figure 2. Plot of canopy height (m) versus upper canopy temperature (°C) colored by forest degradation type. Each point represents a single AGB sample from GEDI with a corresponding upper canopy temperature observation from ECOSTRESS for burned (red points), logged (blue points) and intact (green points) forest types.

for selectively logged forests with 34.9 °C, 34.8 °C, and 0.88 °C for intact forests; table 1).

We found similar mean canopy heights (rh95) calculated among forest types spanning the entire extent of each within our study area (18.9 m, 19.8 m and 20.4 m for burned, intact and logged forests, respectively) (figure 2). Despite height similarities, diverging structural patterns emerged when examining the height of median energy (rh50; figure S2). Burned forests exhibited 35%–32% lower rh50 values relative to intact and logged forests (burned forest mean rh50 was 7.41 m versus 11.5 m and 11.0 m for intact and logged forests, respectively). This suggests extensive structural degradation in the understory canopy of burned forest and only modest understory disturbance in logged forests relative to intact forests.

3.2. Modeled upper canopy temperatures

After controlling for environmental factors, forest degradation type had a substantial influence on upper canopy temperature. Based on our best-performing model (Model 5 c, RMSE = 0.76 °C, MAE = 0.47 °C, R^2 = 0.55), burned forests were substantially warmer than intact and logged forests (table S3). Across all canopy heights (5–40 m), burned forests were 2.6 °C warmer on average [1.39–3.96 °C, 95% credible interval] than intact forests (figure 3(A) and table S4). This temperature difference was greater at taller canopy heights, with burned forests showing a slight positive relationship between height and temperature. At 40 m height, burned forests were 2.8 °C warmer [1.57 °C–4.32 °C] than intact forests, compared to 2.2 °C warmer [1.39 °C–3.28 °C] at 5 m height. Across all heights, logged forests were slightly cooler

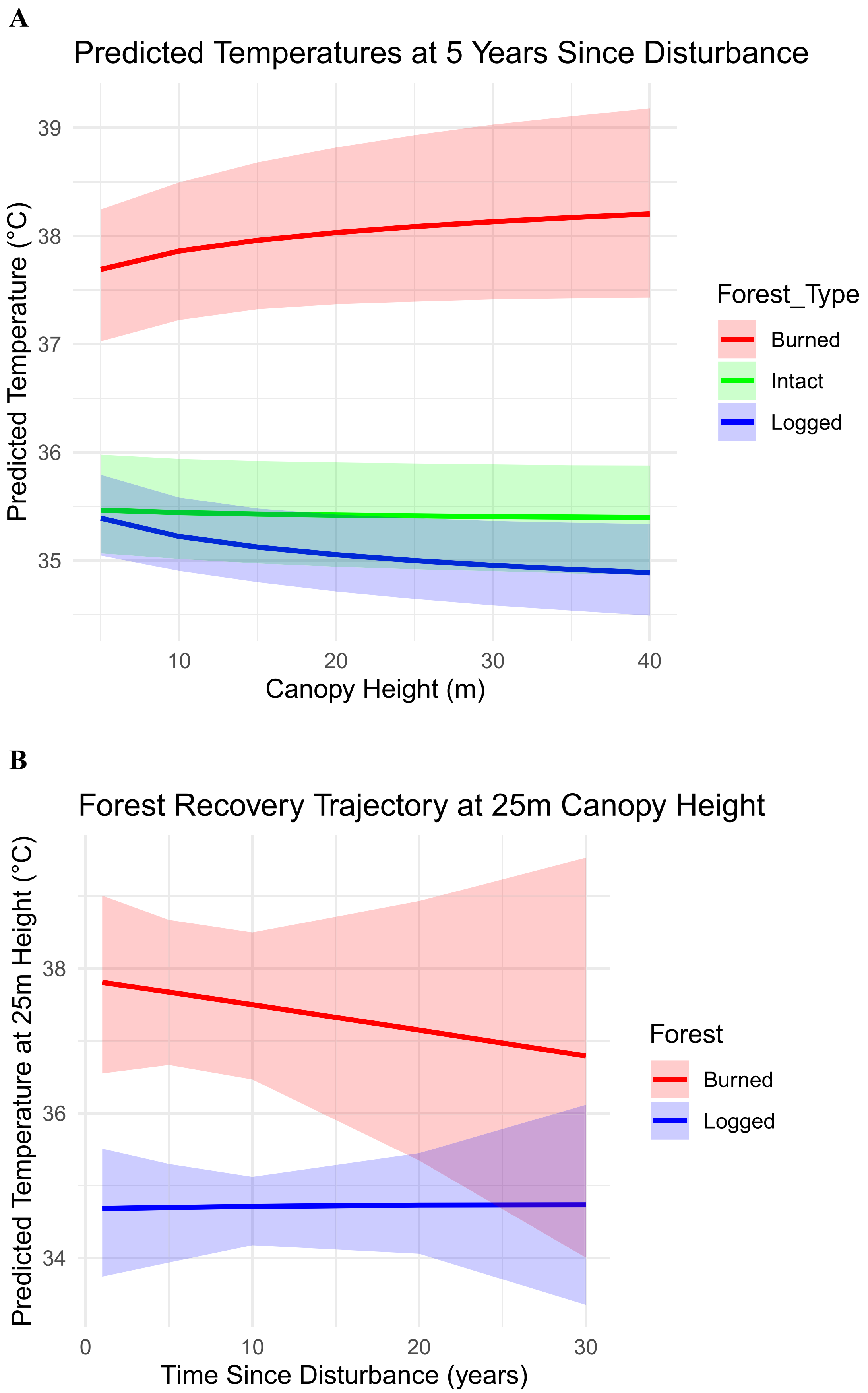

Figure 3. (A) Predicted upper canopy temperature with 5%–95% credible interval uncertainty bounds as a function of canopy height (modeled after 5 years of recovery for logged and burned forests). (B) Predicted upper canopy temperature (with 5%–95% CI) as a function of time since disturbance modeled at 25 m canopy height for all forest types. Estimates are based on posterior hyperparameter distributions for each forest type: intact, logged and burned. All other covariates are constant and equal to their mean.

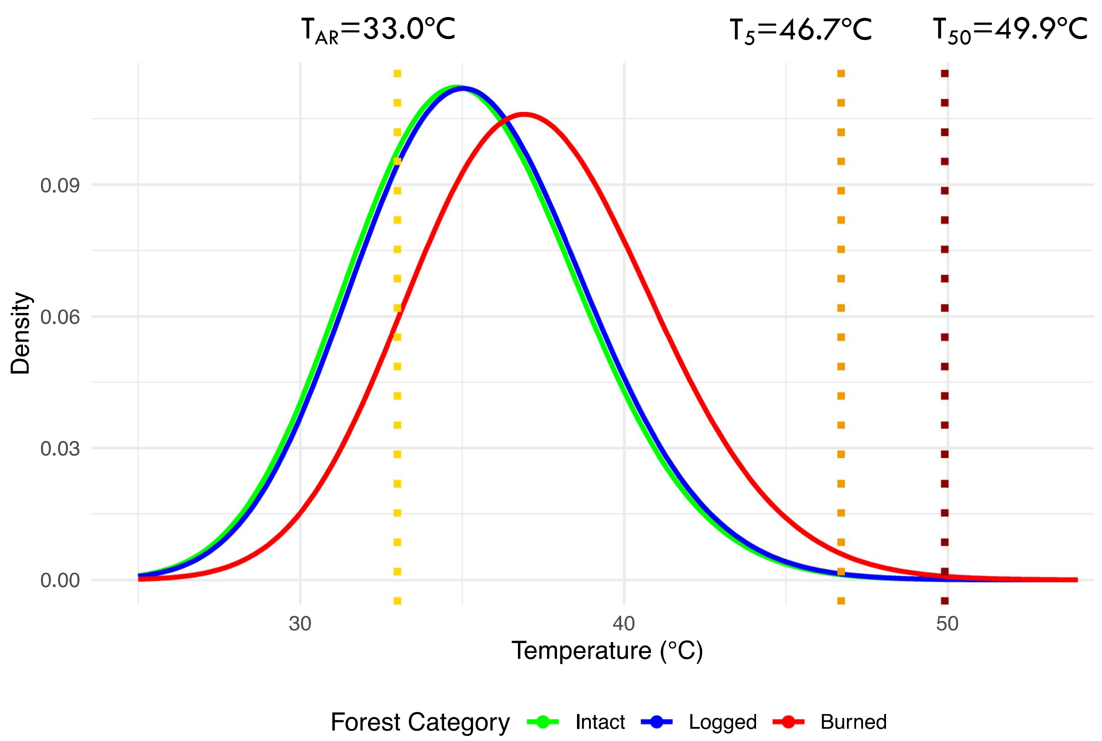

Figure 4. Modeled upper canopy leaf temperature distributions based on thermal observations from ECOSTRESS of forest patches (intact, logged and burned) on the warmest, cloud-free observation in a 2 year dry season record (31-08-2018). Results for the warmest patches (top quartile) are shown. See table 2 for gamma distribution parameters and uncertainties. Dashed lines represent three critical temperature thresholds: the temperatures at which (1) respiration begins to exceed photosynthesis (T_{AR}), (2) the ratio of variable chlorophyll a fluorescence yield to maximum fluorescence yield (Fv/Fm) starts to decline by 5% (T_5), and (3) the ratio Fv/Fm declines by 50% (T_{50}). See table 3 for quantified probabilities of each forest type exceeding the three temperature thresholds.

than intact forests by 0.35 °C on average [−1.13–0.27 °C], though the confidence intervals overlapped zero (table S4). Logged forests exhibited a negative relationship between canopy height and temperature, with temperatures decreasing by approximately 0.51 °C when comparing 5 m to 40 m heights. In contrast, intact forests maintained remarkably stable temperatures across all canopy heights, with only a 0.07 °C decrease from 5 m to 40 m height.

Time since last disturbance played a clear role in thermal recovery but varied by disturbance type (figure 3(B)). Burned forests showed a modest temperature decrease of approximately 1.2 °C over a 30 year recovery period following disturbance, supporting the hypothesis that some thermal recovery occurs as vegetation regrows after fire. In contrast, logged forests showed a minimal change of only 0.1–0.2 °C over the same 30 year period, suggesting that the selective logging practices in our study area had limited lasting impact on canopy thermal properties.

The model's parameter estimates indicate that burning increased the baseline temperature coefficient ($\beta_c_$burned = 1.46 [0.01, 3.02]), while logging showed no clear effect ($\beta_c_$logged = − 0.72 [−2.08, 0.99]). Neither disturbance type altered the temperature-height relationship exponent on its own (table S3), though the interaction between logging and time modified this relationship ($\beta_z_$logged_time = 0.009 [−0.008, 0.028]).

The model performance varied by forest type, with lower prediction errors for intact and logged forests (RMSE = 0.700 °C and 0.696 °C; MAE = 0.440 °C and 0.419 °C, respectively) compared to burned forests (RMSE = 1.05 °C; MAE = 0.764 °C). Mean residuals revealed negligible bias in the model (mean residual = 0.0068 °C, 0.033 °C and 0.145 °C for intact, logged and burned forests, respectively). The residuals suggest that our model provides a conservative estimate of the temperature differences between burned and intact forests, potentially underestimating the true thermal impact of burning on forest canopies.

3.3. Modeled upper canopy leaf temperature distributions

Similar to canopy temperatures, modeled upper canopy leaf temperatures of warmest quartile intact and selectively logged forest types also closely resembled one another, with their means differing less than 0.5 °C (figure 4 and table 2). Meanwhile, the leaf temperature distribution of burned forests exhibited a higher mean and spread. Nearly three quarters

Table 2. Fitted parameters (shape and rate) of the modeled gamma distribution of upper canopy leaf temperature distributions based on thermal observations from ECOSTRESS of the warmest (top quartile) patches of intact, logged and burned forests on 31-08-2018 (corresponding to figure 4). Parameter uncertainties are quantified with [2.5%, 97.5%] credible intervals.

Forest type	Shape [2.5%, 97.5%]	Rate [2.5%, 97.5%]
Intact	96.85 [91.19, 99.91]	2.75 [2.59, 2.84]
Logged	97.53 [93.27, 99.92]	2.76 [2.63, 2.83]
Burned	97.57 [91.99, 99.93]	2.62 [2.47, 2.69]

Table 3. Probability of an upper canopy leaf in the warmest forest patches (top quartile temperatures of intact, logged and burned forests) exceeding three critical temperatures thresholds at which: (1) respiration begins to exceed photosynthesis (T_{AR}), (2) the ratio of variable chlorophyll a fluorescence yield to maximum fluorescence yield (Fv/Fm) starts to decline by 5% (T_5), and (3) the ratio Fv/Fm declines by 50% (T_{50}).

Forest type	$P(x > T_{AR})$	$P(x > T_5)$	$P(x > T_{50})$
Intact	72.4%	0.155%	0.0114%
Logged	74.2%	0.184%	0.0140%
Burned	87.5%	0.972%	0.1142%

of the light-exposed leaves were expected to exceed the T_{AR} temperature threshold in the warmest forest patches for both intact (72%) and selectively logged (74%) forests (table 3 and figure 4). This probability increased to 87% in burned forests. Probabilities that individual leaves would exceed T_5 were over five times greater in burned forests (0.97%) relative to intact and selectively logged forests (0.15% and 0.18%, respectively) and were an order of magnitude higher than the probabilities of exceeding T_{50} for all forest types (compare 0.11% probability of exceeding T_{50} in burned forests with 0.011% and 0.014% probabilities for intact and selectively logged forests, respectively) (table 3).

4. Discussion

Burned forests exhibited substantially altered thermal regimes even after 15 years of recovery, with consistently elevated temperatures across all canopy heights (figure 3). The slight increase in temperature with increasing height in burned forests contrasts with the patterns observed in both intact and logged forests, suggesting fundamental differences in energy balance mechanisms following fire disturbance.

Our model showed that intact forests maintain remarkably consistent temperatures across all canopy heights (only 0.07 °C difference from 5 m to 40 m), while logged forests exhibited a modest cooling trend with increasing height (0.51 °C cooler at 40 m compared to 5 m). In contrast, burned forests showed increasing temperatures with height (0.51 °C warmer at 40 m compared to 5 m). However, the temperature differences among forest types far exceed the within-type variations across heights: the average temperature difference between burned and intact forests (2.6 °C) is approximately 37 times larger than the height-related temperature variation within intact forests (0.07 °C) (figure 3).

Our results align with previous studies indicating that forest degradation—especially fire—alters canopy structure and energy balance (Longo *et al* 2020), leading to increased surface temperatures (de Oliveira *et al* 2021). Edge effects, with burned forests often closer to forest boundaries, may contribute to this pattern (Camargo and Kapos 1995) through advection of warm air from surrounding agricultural areas (Giancola *et al* 2024). The weak positive temperature-height relationship we observed in burned forests differs from previous work that showed negative relationships between height (or aboveground biomass) and canopy temperature across varying sites (Longo *et al* 2020) and degradation intensities (Rangel Pinagé *et al* 2023a). This difference may be explained by several factors. First, temporal differences in measurements could capture different post-disturbance recovery stages (our study used an ECOSTRESS scene captured in 2018, while Rangel Pinagé used 2019 MODIS and Landsat data and Longo *et al* used field measurements combined with airborne lidar from 2012–2015 to calibrate their Ecosystem Demography model). Second, methodological differences exist: our model focused on isolating the specific effects of canopy height and degradation while controlling for environmental variables including elevation and air temperature, whereas Rangel Pinagé *et al* (2023a) examined bivariate relationships across different forest structure metrics, and Longo *et al* (2020) used a mechanistic ecosystem model that simulates degradation impacts on evapotranspiration and associated temperature changes.

The thermal similarity between intact and selectively logged forests (−0.35 °C average difference [−1.13 °C–0.27 °C]) may be explained by several mechanisms. First, selective logging in our study region typically removes only a small fraction of trees, preserving most canopy structure. Indeed, our data showed minimal biomass differences between logged and intact forests (120 vs. 122 Mg ha^{-1}), suggesting much lower impact than fire damage (Rappaport *et al* 2018). Second, logging primarily targets specific valuable trees affecting less than 20% of stand biomass (Rocha *et al* 2024), while maintaining overall canopy cover. A study in the eastern Amazon found that the canopy over logged sites is mostly closed by three years after logging with the exception of small portions of the logged area (main roads, log decks, and the largest tree falls) (Asner *et al* 2004). Our logging sites, being 1–30 years post-disturbance, have likely experienced substantial canopy recovery. Third, although logged forests showed modest understory disturbance (rh50 reduced by 4.3% relative to intact forest), they maintained intact upper canopy layers that are critical for thermal regulation (Pau *et al* 2018).

In contrast, burned forests exhibited structural alterations across all canopy layers. The extensive midstory degradation (rh50 reduced by 35.6% compared to intact forests) combined with upper canopy damage creates a compounding effect that increases light penetration and heat loading throughout the profile. The slight increase in temperature with height in burned forests (0.51 °C from 5 m to 40 m) suggests that upper canopy damage may be prevalent, with taller burned forests potentially having less capacity for evaporative cooling due to hydraulic limitations, despite the gradual recovery observed over time.

The ecological implications of observed thermal stress depend on exposure frequency and duration. Our ECOSTRESS measurements capture instantaneous canopy temperatures, but we observed similar high-temperature patterns at least three times since 31 August 2018 over a monitoring period between 2018–2024 (figure S8), suggesting these are not rare events. At the T_{AR} threshold (typically 30 °C–35 °C), negative carbon balance can begin within hours of sustained exposure, though trees can often recover overnight (Slot and Winter 2017). More acute thermal stress begins at T_5 (46.7 °C) with initial chlorophyll fluorescence decline (Berry and Bjorkman 1980, Slot *et al* 2021), while at T_{50} (49.9 °C), 50% of leaf area becomes necrotic after 15 min exposure (Krause *et al* 2010). However, experimental evidence shows that intact tropical seedlings can survive and produce new leaves even after experiencing T_{50}, suggesting some resilience at the whole-plant level (Krause *et al* 2010).

Recent studies have quantified several substantial carbon investments including root exudation (∼10% of gross primary productivity globally; (Chari *et al* 2024)) and mycorrhizal associations (estimated at 13.12 Gt CO2e yr^{-1}; (Hawkins *et al* 2023)). While our estimates of leaves exceeding respiratory thresholds represent a snapshot rather than a comprehensive carbon cost assessment, they suggest degraded forests experience more frequent exposure to temperatures that affect carbon balance through multiple pathways. In water-limited regions like our study area where dry season rainfall averages <100 mm month^{-1} (Rangel Pinagé *et al* 2023a), thermal stress compounds with drought stress because trees must balance carbon uptake against water loss. This matters because carbon reserves are critical for maintaining hydraulic function (Adams *et al* 2017)—depleted reserves reduce a tree's capacity to repair embolized xylem vessels, which requires significant energy investment estimated at 7%–10% of daily carbon fixation (Secchi and Zwieniecki 2011).

Probabilities that individual leaves would exceed T_{50} were 10 times greater in burned forests (0.11%) relative to intact and selectively logged forests (0.011% and 0.014%, respectively). The low absolute probabilities indicate substantial thermal tolerance in these forests under current climatic conditions. However, the relative differences among forest types suggest that degradation by fire increases vulnerability to thermal damage through greater leaf maintenance and replacement costs. Such conditions are already leading to carbon losses, reduced forest productivity, and mortality (Esquivel-Muelbert *et al* 2020), potentially creating a positive feedback loop that further exacerbates forest degradation and climate change (Anderson-Teixeira *et al* 2013). Our finding that selectively logged forests maintain thermal regimes remarkably similar to intact forests, however, suggests an important limitation to this feedback concern. It indicates that certain forms of forest degradation—particularly selective logging as practiced in our study area—may not substantially alter the forest's thermal environment or trigger cascading thermal stress effects. This represents a potentially optimistic finding for forest management, suggesting that carefully implemented selective logging practices could maintain many of the climate regulation services of intact forests despite some loss of biomass, though longer-term monitoring would be needed to confirm this result.

Higher temperatures and increased thermal variability in burned forests may impede natural regeneration processes by affecting seed germination, seedling establishment, and sapling growth (Uriarte *et al* 2016). This could necessitate more intensive restoration interventions in heavily degraded areas to overcome these thermal barriers to forest recovery. Early successional trees like *Cecropia* tend to have faster growth rates (Rozendaal and Chazdon 2015), which are generally associated with higher mortality risk across Amazonia (Esquivel-Muelbert *et al* 2020). However, the relationship between successional status and thermal tolerance appears more nuanced. Studies comparing the pioneer *Ficus insipida* with the late-successional *Virola sebifera* found that neither species showed heat acclimation capacity in their sunlit leaves, with both operating close to their thermal tolerance limits (Krause *et al* 2010). This suggests that both early and late successional species may be vulnerable to increasing heat stress, though potentially through different mechanisms. While pioneers' rapid growth rates and limited access to deeper water tables may make them more susceptible to drought-induced mortality during heat waves, late-successional species' slower growth and denser wood may provide some buffer against short-term thermal stress but less capacity for acclimation over time.

The limited capacity for thermal acclimation observed across tropical species compared to temperate ones (Cunningham and Read 2003) suggests that Amazon forests may be particularly vulnerable to the combined effects of rising temperatures and increasing drought frequency. In the driest Amazonian region—which coincides with our study site—species-level bioclimatic distributional patterns were found to predict the risk of tree death (Esquivel-Muelbert *et al* 2020), suggesting that these forests are already experiencing climatic conditions of thermal stress and water stress beyond their adaptive limits. Climate mitigation strategies may need to account for reduced forest carbon sequestration for tropical forests, especially those recovering from fires.

5. Conclusion

This study revealed clear differences in thermal stress patterns among burned versus selectively logged and intact forests in Feliz Natal Municipality in the Brazilian Amazon Arc of deforestation. Burned forests exhibited markedly higher canopy temperatures and a greater probability of exceeding critical physiological thresholds, highlighting their vulnerability to current and future climate warming. The similarity in thermal patterns between intact and selectively logged forests suggests that logging has less severe immediate impacts on forest thermal regimes compared to burning. Our findings underscore the benefits of incorporating thermal remote sensing in forest degradation assessments. As global temperatures increase, understanding and mitigating thermal stress in degraded tropical forests will be crucial for maintaining their ecological functions and carbon storage potential.

Data availability statement

The data that support the findings of this study are openly available at the following URL/DOI: https://doi.org/10.6084/m9.figshare.27923028.

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Conflict of interest

The authors have no competing interests to declare.

Author contributions

S.C. wrote the main manuscript text. M.K., S.C., M.L., and D.M. contributed with early stages of conceptualization and provided key suggestions throughout the analysis and writing of the manuscript. S.C. performed all data processing and statistical analyses. M.K., M.L., O.C., A.D., R.C., V.S., M.A., P.D., D.A., M.G, D.M., and K C-N. contributed in conceptualization and interpretation of results. All authors contributed to editing and reviewing the manuscript.

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References

Adams H D *et al* 2017 A multi-species synthesis of physiological mechanisms in drought-induced tree mortality *Nat. Ecol. Evol.* **1** 1285–91

Anderson-Teixeira K J, Miller A D, Mohan J E, Hudiburg T W, Duval B D and DeLucia E H 2013 Altered dynamics of forest recovery under a changing climate *Glob. Change Biol.* **19** 2001–21

Asner G P, Keller M and Silva J N M 2004 Spatial and temporal dynamics of forest canopy gaps following selective logging in the eastern Amazon *Glob. Change Biol.* **10** 765–83

Berry J and Bjorkman O 1980 Photosynthetic response and adaptation to temperature in higher plants *Annu. Rev. Plant Physiol.* **31** 491–543

Brodribb T J, Powers J, Cochard H and Choat B 2020 Hanging by a thread? Forests and drought *Science* **368** 261–6

Camargo J L C and Kapos V 1995 Complex edge effects on soil moisture and microclimate in central Amazonian forest *J. Trop. Ecol.* **11** 205–21

Chari N R *et al* 2024 The global root exudate carbon flux *bioRxiv Preprint* https://doi.org/10.1101/2024.02.01.578470 (posted online 6 February 2024, accessed 16 January 2025)

Chazdon R L 2014 Synthesis: the promise of tropical forest regeneration in an age of deforestation *Second Growth: The Promise of Tropical Forest Regeneration in an Age of Deforestation* (University of Chicago Press) ch 15, pp 304–16

Cooley S S *et al* 2024 Combining spaceborne lidar from the global ecosystem dynamics investigation with local knowledge for monitoring fragmented tropical landscapes: a case study in the forest–agriculture interface of Ucayali, Peru *Ecol. Evol.* **14** e70116

Csillik O *et al* 2024 A large net carbon loss attributed to anthropogenic and natural disturbances in the Amazon Arc of Deforestation *Proc. Natl Acad. Sci.* **121** e2310157121

Cunningham S C and Read J 2003 Do temperate rainforest trees have a greater ability to acclimate to changing temperatures than tropical rainforest trees? *New Phytol.* **157** 55–64 de Oliveira G, Brunsell N A, Chen J M, Shimabukuro Y E, Mataveli G A V, Dos Santos C A C, Stark S C, de Lima A and Aragao L E O C 2021 Legacy effects following fire on surface energy, water and carbon fluxes in mature Amazonian forests *J. Geophys. Res.* **126** e2020JG005833

Doughty C E *et al* 2023 Tropical forests are approaching critical temperature thresholds *Nature* **621** 7977

Dubayah R *et al* 2020 The global ecosystem dynamics investigation: high-resolution laser ranging of the Earth's forests and topography *Sci. Remote Sens.* **1** 100002

Esquivel-Muelbert A *et al* 2020 Tree mode of death and mortality risk factors across Amazon forests *Nat. Commun.* **11** 5515

Fearnside P 2017 Deforestation of the Brazilian Amazon *Oxford Research Encyclopedia of Environmental Science* (Oxford University Press) (https://doi.org/10.1093/acrefore/9780199389414.013.102)

Giancola D T, Escada M I S, Rosa M G B, Andrade A C, Laurance S, Laurance W F, Vicentini A and Camargo J L C 2024 Degradation exposure scenario in the Brazilian Amazon: edge effect on hyperdominant C-cycle tree species *For. Ecol. Manage.* **562** 121926

Guan K *et al* 2015 Photosynthetic seasonality of global tropical forests constrained by hydroclimate *Nat. Geosci.* **8** 284–9

Hawkins H-J, Cargill R I M, Van Nuland M E, Hagen S C, Field K J, Sheldrake M, Soudzilovskaia N A and Kiers E T 2023 Mycorrhizal mycelium as a global carbon pool *Curr. Biol.* **33** R560–73

Heinrich V H A *et al* 2023 The carbon sink of secondary and degraded humid tropical forests *Nature* **615** 7952

Hulley G C, Göttsche F M, Rivera G, Hook S J, Freepartner R J, Martin M A, Cawse-Nicholson K and Johnson W R 2022 Validation and quality assessment of the ECOSTRESS level-2 land surface temperature and emissivity product *IEEE Trans. Geosci. Remote Sens.* **60** 1–23

Hulley G and Freepartner R 2018 *ECOsystem Spaceborne Thermal Radiometer Experiment on Space Station (ECOSTRESS): Level-2 Land Surface Temperature and Emissivity Algorithm Theoretical Basis Document* (California Institute of Technology)

Krause G H, Winter K, Krause B, Jahns P, García M, Aranda J and Virgo A 2010 High-temperature tolerance of a tropical tree, Ficus insipida: methodological reassessment and climate change considerations *Funct. Plant Biol.* **37** 890

Longo M *et al* 2020 Impacts of degradation on water, energy, and carbon cycling of the Amazon tropical forests *J. Geophys. Res.* **125** e2020JG005677

McDowell N G *et al* 2020 Pervasive shifts in forest dynamics in a changing world *Science* **368** eaaz9463

Pau S, Detto M, Kim Y and Still C J 2018 Tropical forest temperature thresholds for gross primary productivity *Ecosphere* **9** e02311

Plummer M, Stukalov A and Denwood M 2024 rjags: Bayesian graphical models using MCMC (version 4–16) *Computer Software* (available at: https://cran.r-project.org/web/packages/rjags/index.html)

Rangel Pinagé E, Bell D M, Longo M, Silva C A, Csillik O and Huete A 2023a Surface energy dynamics and canopy structural properties in intact and disturbed forests in the Southern Amazon *J. Geophys. Res.* **128** e2023JG007465

Rangel Pinagé E, Keller M, Peck C P, Longo M, Duffy P and Csillik O 2023b Effects of forest degradation classification on the uncertainty of aboveground carbon estimates in the Amazon *Carbon Balance Manage.* **18** 2

Rappaport D I, Morton D C, Longo M, Keller M, Dubayah R and dos-Santos M N 2018 Quantifying long-term changes in carbon stocks and forest structure from Amazon forest degradation *Environ. Res. Lett.* **13** 065013

Ray D, Nepstad D and Moutinho P 2005 Micrometeorological and canopy controls of fire susceptibility in a forested Amazon landscape *Ecol. Appl.* **15** 1664–78

Rocha N C V, Adami M, Galbraith D and de Freitas L J M D 2024 Signature of logging in the Brazilian Amazon still detected after 17 years *For. Ecol. Manage.* **561** 121850

Rozendaal D M A and Chazdon R L 2015 Demographic drivers of tree biomass change during secondary succession in northeastern Costa Rica *Ecol. Appl.* **25** 506–16

Secchi F and Zwieniecki M A 2011 Sensing embolism in xylem vessels: the role of sucrose as a trigger for refilling *Plant Cell Environ.* **34** 514–24

Slot M, Cala D, Aranda J, Virgo A, Michaletz S T and Winter K 2021 Leaf heat tolerance of 147 tropical forest species varies with elevation and leaf functional traits, but not with phylogeny *Plant Cell Environ.* **44** 2414–27

Slot M and Winter K 2017 In situ temperature response of photosynthesis of 42 tree and liana species in the canopy of two Panamanian lowland tropical forests with contrasting rainfall regimes *New Phytol.* **214** 1103–17

Smith N G and Dukes J S 2017 Short-term acclimation to warmer temperatures accelerates leaf carbon exchange processes across plant types *Glob. Change Biol.* **23** 4840–53

Souza-Filho P W M, de Souza E B, Silva Júnior R O, Nascimento W R, Versiani de Mendonça B R, Guimarães J T F, Dall'Agnol R and Siqueira J O 2016 Four decades of land-cover, land-use and hydroclimatology changes in the Itacaiúnas River watershed, southeastern Amazon *J. Environ. Manage.* **167** 175–84

Still C, Powell R, Aubrecht D, Kim Y, Helliker B, Roberts D, Richardson A D and Goulden M 2019 Thermal imaging in plant and ecosystem ecology: applications and challenges *Ecosphere* **10** n/a

Uriarte M, Schwartz N, Powers J S, Marín-spiotta E, Liao W and Werden L K 2016 Impacts of climate variability on tree demography in second growth tropical forests: the importance of regional context for predicting successional trajectories *Biotropica* **48** 780–97

Winter K 2024 Are tropical forests approaching critical temperature thresholds? *Plant Biol.* **26** 495–8