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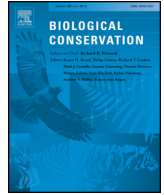
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## Short communication

## Evaluating climber cutting as a strategy to restore degraded tropical forests



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## ABSTRACT

A substantial share of the remaining tropical forest cover is represented by historically degraded fragments exposed to severe edge effects, where ruderal plants proliferate vigorously and may arrest succession. We tested climber plant cutting as strategy to restore a semideciduous tropical forest remnant that is dominated by ruderal climbers. We compared control (unmanaged) plots with plots subjected to climber cutting at 1-m height with recutting one (after 8 months) or three times (8, 24 and 36 months). We monitored: 1) tree and shrub biomass gain and canopy openness for three years; 2) tree and shrub growth and recruitment of regenerating seedlings for one year; and 3) planted seedling survival for two years. Climber cutting increased biomass gain by ~51% for smaller trees and shrubs ( $1.58 \leq \text{dbh} < 5 \text{ cm}$ ) only, regardless of the number of re-cuts. Canopy openness increased following climber cutting, but recovered after ten months due to rapid growth of the tree canopies. Growth of regenerating seedlings, but not abundance, was favored by climber cutting. Initial cutting of climbers enhanced survival of enrichment plantings, but this benefit declined with canopy re-occupation by tree foliage. Although longer-term research is needed, cutting ruderal climbers in degraded forest remnants was shown to be a promising approach to enhance forest regeneration and carbon sequestration, justifying its consideration in the restoration agenda as a complementary activity to increasing forest cover in former agricultural lands.

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## 1. Introduction

Many small forest remnants are scattered throughout human-modified landscapes, exposing them to edge effects and chronic human-mediated disturbances that lead to reduced biomass (Chaplin-Kramer et al., 2015) and proliferation of ruderal plant species (Tabarelli et al., 2012). Ruderal plant species are more resilient to disturbances typical of small degraded forest remnants and grow rapidly when light is not a limiting factor – as in forest gaps, edges, or disturbed sectors (Tabarelli and Lopes, 2008; Schnitzer and Carson, 2010). In this context, ruderal trees, ferns, bamboo, or climber species may become hyperabundant and arrest successional processes (Pinard et al., 1999; Tabarelli et al., 2012). Climbers, in particular, directly compete with trees, reducing their growth, and increasing mortality (Paul and Yavitt, 2010).

Although degraded and isolated, some forest remnants may still house a considerable share of local biodiversity (Chazdon et al., 2009), provide valuable ecosystem services (Ferraz et al., 2014), and represent the main sources of plant propagules and fauna to young second-growth and restored forests in fragmented landscapes (Brancalion et

al., 2013). Nevertheless, restoration projects usually focus on planting trees in completely deforested areas, despite the fact that active restoration is expensive and rarely reaches biodiversity levels of reference sites (Suganuma and Durigan, 2015; Shoo et al., 2016). In contrast, few resources are directed to improve existing forest remnants through management and enrichment plantings (Bertacchi et al., 2016).

We tested climber plant cutting and enrichment plantings as restoration strategies in a semideciduous tropical forest remnant in southeastern Brazil dominated by ruderal climbers. We monitored: 1) tree and shrub biomass gain and canopy openness for three years; 2) tree and shrub growth and recruitment of regenerating seedlings for one year; and 3) planted seedling survival for two years. We hypothesized that climber cutting would increase biomass gain of trees and establishment and growth of spontaneously regenerating and planted seedlings.

## 2. Methods

## 2.1. Study site

This study was carried out in a 14-ha tropical forest remnant in the municipality of Piracicaba, São Paulo state, southeastern Brazil (lat 22° 42' 40"S, long 47° 37' 30"W), at ~550 m elevation. The climate is classified as Cwa according to the Köppen–Geiger classification system, with

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hot, humid summers and dry winters (Alvares et al., 2013). During the study period (2012–2015), average monthly temperature was 22.7 °C, average annual precipitation was 1377 mm in the first two years (2012–2013) and 877 mm in the third year (2014). A hurricane with wind speeds of up to 180 km/h struck our study area in the second year. Vegetation is characterized as a semideciduous seasonal tropical forest, which is part of the “interior” biogeographical sub-region of the Atlantic Forest global biodiversity hotspot, one of its most threatened sub-regions with only 7.1% forest cover remaining (Ribeiro et al., 2009).

Like most tropical forest remnants embedded in human-dominated landscapes, the study area has been impacted by several types of anthropogenic disturbances, such as selective logging, cattle grazing, and hunting. The most recent severe impact was a fire that burned most of the fragment 35 years before our study. Since then, tree community composition has changed little and climbers have proliferated vigorously. The dense liana cover in the canopy led us to infer that successional processes have been slowed in the study area, i.e., arrested succession (Pinard et al., 1999; Schnitzer et al., 2000). We conducted our experiment in sectors of the forest remnant that had lower abundance of medium- to large-sized trees and were dominated by ruderal climbers (hereafter “degraded sectors”); abundance of rooted climbers > 1 cm diameter at 1.3 m from the rooting point was ~6500–7000 individuals/ha in these degraded sectors, and ~30% of individuals belonged to only three species: *Mansoa difficilis* (Cham.) Bureau & K. Schum. (Bignoniaceae), *Lundia obliqua* Sond. (Bignoniaceae), and *Dicella bracteosa* (A. Juss.) Griseb (Malpighiaceae; Mello, F. A., unpublished data).

## 2.2. Experimental design

We refer to *climbers* as all vines and lianas that grow a substantial distance upward from the ground but require support of a host plant to ascend to the canopy, recognizing that for some species the need for support is facultative. We use the term *liana* to refer to only woody climber plants (Putz and Mooney, 1991). In this work, we cut all climber plants. In degraded forest sectors, we installed 30 circular plots with a 10-m radius (314.15 m<sup>2</sup>) separated by a minimum of 4 m. The plots were grouped in 10 blocks of three plots separated by a minimum of 6 m, based on spatial proximity and forest structure. We randomly selected two plots in each block for climber cutting (CC plots – 20 plots) and the other ‘control’ plots were not managed (10 plots). Tree and shrub aboveground biomass, abundance, and species density, and liana abundance were similar in all treatments one month before climber cutting (data not shown). We also marked five plots in areas of the forest remnant with larger trees and low liana cover to serve as a reference for some forest community and biomass comparisons (hereafter “less disturbed sectors”). We thought these areas were a more realistic

target for restoration than contiguous forest with minimal human disturbance.

We cut all climber plants in CC plots (hereafter ‘climber cutting’) at 1-m height using a machete, and left the biomass on the ground or hanging on trees, since biomass removal from the canopy could cause accidents, damage trees, and increase labor requirements (Fig. 1). We cut all climbers up to 2 m away from the plot border to create a buffer; although climbers can grow for several meters horizontally in the forest canopy, we did not observe substantial climber cover from plant rooted outside the CC plots. We randomly selected one plot in each block in which we re-cut climbers once, 8 months after initial cutting (CC1 – 10 plots), and one plot to receive repeated re-cutting climbers: three times at 8, 24 and 36 months after initial cutting (CC3 – 10 plots). The CC1 and CC3 treatments did not differ with respect to climber cutting frequency during the first 24 months, so for analyses prior to 24 months, we combined the results from the CC1 and CC3 plots and refer to them as CC plots.

## 2.3. Data collection

### 2.3.1. Canopy openness

We measured canopy openness 2, 6, 10, 14, 20, 32 and 38 months after climber cutting by sampling four points, one in each corner of the plot, using a convex densiometer and averaging the values.

### 2.3.2. Tree and shrub community

We measured abundance, height, and dbh of all trees and shrubs with dbh ≥ 1.58 cm in all plots 1 month before, and 18 and 36 months after climber cutting. We considered 1.58 cm dbh as the minimum sampling size in order to include smaller individuals that could respond at faster rates to climber cutting and because forest inventories in our study region usually sample trees with circumference at breast height ≥ 5 cm. All trees and shrubs sampled were identified to the highest taxonomic level possible according to the Angiosperm Phylogeny Group III (2016). Abundance, height, and identity of tree and shrub seedlings (height > 10 cm; dbh < 1.58 cm) were measured in four 1.5 × 1.5 m subplots in 12 plots (six CC and six control plots) and four 1 × 1 m subplots in eight other plots (four CC and four control plots) one month before climber cutting and one year after climber cutting, but before climber re-cutting. Seedling abundance data were standardized to the number of individuals per m<sup>2</sup>. We used the diameter-based model developed by Van Breugel et al. (2011) for trees dbh > 1 cm to estimate aboveground biomass of trees and shrubs dbh ≥ 1.58 cm.

### 2.3.3. Enrichment plantings

We planted two seedlings (20–40 cm in height) of each of ten native species in ten CC and control plots 1 and 10 months after climber cutting



Fig. 1. Ruderal climbers covering the crown of a native tree (A), dead climbers hanging on trees (B), and the same tree free of climbers, after spontaneous fall of dry climber foliage and stems (C).

(hereafter plantings I and II, respectively); both plantings were done during the rainy season. Seedlings were grown in  $9 \times 2.8$  cm ( $56\text{-cm}^3$ ) round tubes in a local forest nursery and during their last month in the nursery were fully exposed to sunlight and received reduced irrigation. Planted seedling survival and growth were monitored for 20 and 10 months for plantings I and II, respectively. A timeline of all data collection is provided in Supplementary File 1.

2.4. Data analysis

We analyzed biomass gain overall and separately for three diameter classes ( $1.58 \leq \text{dbh} < 5$  cm,  $5 \leq \text{dbh} < 15$  cm and  $\text{dbh} \geq 15$  cm), as well as canopy openness, using generalized mixed linear models ( $\alpha = 0.05$ ) considering lognormal distribution of data. We considered treatments, block, and time as main effects and specific treatments were compared using Tukey's multiple comparison procedure ( $\alpha = 0.05$ ). Planted seedling survival was compared between treatments using a paired *t*-test ( $\alpha = 0.05$ ). We calculated the relative change in seedling abundance and growth one year after climber cutting as compared to values collected one month prior to climber cutting; these values were compared using the Student *t*-test when data met assumptions of parametric statistics and using a Mann-Whitney test ( $\alpha = 0.05$ ) when they did not.

3. Results

The tree and shrub community included 129 species of which only two were exotic. Most individuals belonged to small, short-lived pioneer/early secondary native species and understory species, whereas canopy tree abundances were low; seven species represented ~40% of the individuals  $\text{dbh} \geq 1.58$  cm sampled (Supplementary File 2). Tree and shrub biomass increased over time in control, CC1 and CC2 ( $F_{\text{Time}(1,24)} = 7.73, p = 0.0104$ ; Fig. 2). On average, climber cutting increased relative biomass gain after three years by 52% for smaller trees and shrubs ( $1.58 \leq \text{dbh} < 5$  cm:  $F_{[2,18]} = 8.12, p = 0.0031$ ) when compared to control plots. There were no significant effects on other size classes separately or all together ( $5 \leq \text{dbh} < 15$  cm:  $F_{[2,18]} = 1.23, p = 0.3156$ ; and  $\text{dbh} \geq 15$  cm:  $F_{[2,16]} = 1.82, p = 0.1940$ , two plots did not have any trees  $\text{dbh} \geq 15$  cm; all size classes together:  $F_{[2,18]} = 0.76, p = 0.4801$ ; Fig. 2).

Climber cutting initially increased canopy openness ( $F_{[12, 162]} = 2.37, p < 0.01$ ), but the canopy was re-occupied by tree foliage by the 10-month measurements (Fig. 3). The hurricane increased canopy openness more in climber cutting than control plots, but canopy

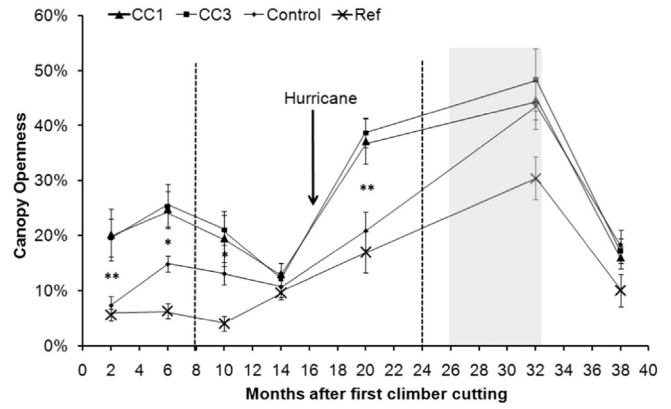


Fig. 3. Canopy openness dynamics after climber cutting. Control: plots without climber cutting; CC1: climbers re-cut 8 months after initial cutting; CC3: climber re-cut 8, 24, and 36 months following initial cutting; Ref: less degraded reference plots. The black arrow indicates a hurricane that struck the study area. The gray area indicates a severe dry season. \* indicates that CC3 differs from control ( $\alpha = 0.05$ ); \*\* indicates that both CC1 and CC3 differ from control ( $\alpha = 0.05, n = 10$  for each treatment and  $n = 5$  for reference plots). Reference plots were excluded from statistical analyses.

openness in all treatments (except less degraded plots) was similar by 16 months after the hurricane (Fig. 3). Canopy cover declined similarly in all treatments in the extremely dry third year of the study (Fig. 3).

Climber cutting increased seedling growth almost fivefold in only one year (CC:  $66.7 \pm 49.8\%$ , control:  $13.3 \pm 17.7\%$ ;  $F_{[1,18]} = 15.36, p = 0.0012$ ), and had no effect on seedling abundance (CC:  $-2.8 \pm 22.8\%$ , control:  $-11.6 \pm 21.3$ ;  $F_{[1,18]} = 0.77, p = 0.3917$ ). Survival of seedlings planted immediately after climber cutting was higher in CC plots after 10 months (CC:  $30.5 \pm 10.6\%$ , control:  $10.9 \pm 6.3\%$ ,  $t_{[1,9]} = 5.31, p < 0.001$ ); survival after 20 months was low overall, but remained higher in CC plots (CC:  $18.0 \pm 9.5\%$ , Con:  $3.0 \pm 2.5\%$ ,  $t_{1,9} = 16.34, p < 0.001$ ). Survival of seedlings planted 10 months after climber cutting did not differ among treatments (CC:  $25.5 \pm 11.3\%$ , control:  $17.5 \pm 11.2\%$ ,  $t_{[1,9]} = 1.32, p = 0.220$ ).

4. Discussion

Climber cutting with only one re-cut after eight months was enough to enhance growth of seedlings and smaller trees and shrubs, which may result in future biomass gains from lower to larger size classes as succession develops. Higher light availability following climber cutting

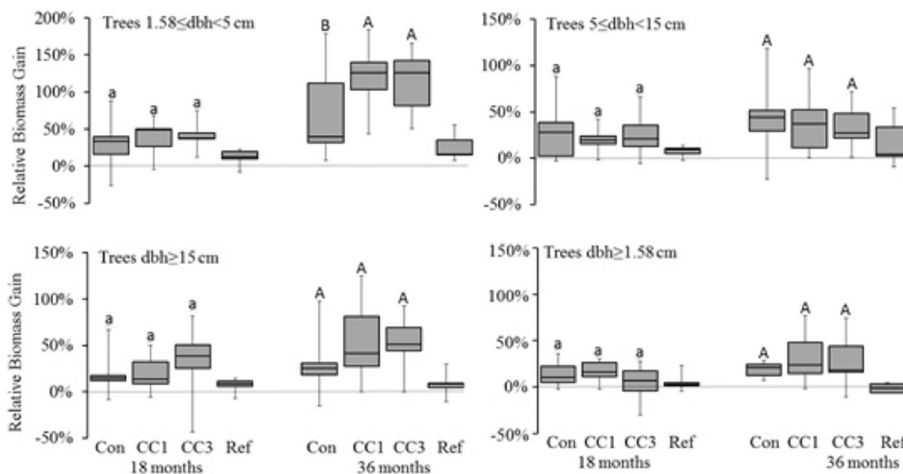


Fig. 2. Relative biomass gain 18 and 36 months after climber cutting for all trees and shrubs sampled (bottom right) and trees and shrubs in different size classes. Con: control plots without any intervention; CC1: climbers re-cut 8 months after first cutting; CC3: climbers were re-cut 8, 24 and 36 months after initial cutting; Ref: less degraded, reference sectors in the forest remnant studied that were not included in statistical analyses. Box plots show median values and upper and lower limits represent the third and first quartiles, respectively. Upper and lower error bars represent maximum and minimum values, respectively. Boxes followed by the same letter do not differ for  $\alpha = 0.05$ . Treatment effects in different periods are distinguished by lower- and uppercase letters.



may have particularly favored growth of both planted and spontaneously regenerating seedlings, as well as small- to medium-sized pioneer trees and shrubs dominating the smaller size class, for which biomass gain was detected. However, fast canopy recovery following climber cutting likely reduced the potential of climber cutting to favor further development of understory plants.

Similar to our results, Venturoli et al. (2015) found that basal area of trees dbh > 3 cm was 24% greater than control plots 4.8 years after liana cutting in a seasonally dry tropical forest of Brazil. Schnitzer et al. (2014) observed stronger effects than ours; biomass accumulation of trees > 1.3 m tall was 180% greater than control plots eight years after lianas were removed in seasonally moist tropical forests of Panama. Both studies were performed in more conserved forests, where climber abundance is substantially lower and there are more old-growth forest species. Our three years of monitoring may not have fully captured the effect of climber cutting on larger tree growth, and longer monitoring periods (e.g., eight years for Schnitzer et al., 2014, ten years for Ingwell et al., 2010; Kainer et al., 2014) may be necessary to determine whether there are biomass gains in larger individuals.

In spite of the biomass gain of smaller-sized trees and shrubs, mostly from pioneer species, further biomass gains may be hampered in degraded forests by reduced recruitment of large-sized, mid- to late-successional tree species due to dispersal limitation. The dominance of the forest community by pioneer species may mean that persistence of mid- to late-successional tree species was historically hampered by the lack of seed sources in a highly fragmented landscape and by competitive exclusion with ruderal plants (Tabarelli et al., 2012). Consequently, positive effects of climber cutting for biomass gain may not be long-lasting. Although young pioneer trees were favored by higher light availability following climber cutting, canopy re-occupation by trees may prevent their further development, and reduced abundance of late-successional would maintain the remnant in arrested succession. Assisting the recruitment of larger tree species through enrichment planting may thus be necessary for long-term recovery of structure and composition of degraded forests.

Enrichment planting has long been suggested as a method to reintroduce tree species in remnant forests for commercial timber production (Lamb et al., 2005), but is yet poorly tested in restoration (Bertacchi et al., 2016). In our study, enrichment planting resulted in low survival rates (<20%), which may be a direct consequence of the high canopy cover (>70%) in both climber cutting and control plots. Nonetheless, climber cutting favored planted seedlings survival, highlighting the value of this intervention for assisting the reintroduction of targeted tree species in forest remnants for both timber production and conservation purposes. This benefit was only obtained, however, when enrichment planting was implemented right after climber cutting. These results suggest that for enrichment planting to be successful, plantings should be concentrated in large gaps where light levels are higher or ongoing overstory clearing will be necessary (Bertacchi et al., 2016; Martínez-Izquierdo et al., 2016).

Although the above-mentioned restoration strategies may support the recovery of composition and structure of degraded fragments, unpredictable natural disturbances must be considered when managing such fragments for conservation purposes. Interestingly, the hurricane and drought had stronger effects on canopy openness than climber cutting treatments. Natural variation in weather may have a stronger impact on recovery than specific restoration strategies (Garrido-Pérez et al., 2008; Martínez-Garza et al., 2013; Wilson, 2015). Wind damage can be exacerbated by climber cutting (Garrido-Pérez et al., 2008); and water limitation and higher canopy openness resulting from drought events in seasonally dry forests may have strong consequences for the competitive balance between trees and climbers (Chen et al., 2015). Thus, further studies are necessary to understand the consequences of climate change on climber proliferation and their management in forest remnants (Schnitzer and Bongers, 2002).

#### 4.1. Conservation implications

In spite of the early benefits of climber cutting as forest restoration strategy, the long term effects on biomass gain, and recovery of late-successional species have yet to be investigated. However, our short-term, spatially limited study suggests cutting ruderal climbers as a promising approach for supporting forest recovery and provisioning of ecosystem services in degraded remnants, justifying its inclusion in the restoration agenda. Since implementation and maintenance costs of restoring degraded remnants can be much lower than active restoration, and their higher resilience compared to agricultural lands support faster biodiversity recovery, climber cutting and enrichment plantings can be more cost-effective than traditional restoration approaches. Other interventions for restoring degraded remnants, such as planting commercial trees around the remnants to create a buffer against edge effects, have yet to be tested. Ecological restoration in highly-fragmented landscapes relies on the potential of these remnants to act as biological sources for restoration. Therefore, conserving forest remnants is essential to long-term conservation, but restoring agricultural land is also important. It does not make sense to increase forest cover in former agricultural lands without taking care of the repositories of biodiversity and ecological memory in the landscape.

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.biocon.2016.07.031>.

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