Restoring logged tropical forests: the removal of woody, climbing plants

by

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AUTHOR'S DECLARATION

I, the author, confirm that the thesis is my own work and has not previously been presented for an award at this, or any other, university. This thesis is a collaborative piece of research from myself (C.F) and several other researchers: Robert P Freckleton (R.P.F), David P Edwards (D.P.E), Anand Roopsind (A.R), Bronson W Griscom (B.W.G), Matthew G Hethcoat (M.G.H), Patrick G Cannon (P.G.C), Robert G Bryant (R.G.B), and Kalsum M Yusah (K.M.Y). I am aware of the University's Guidance on the Use of Unfair Means (www.sheffield.ac.uk/ssid/unfair-means).

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ABSTRACT

In a world facing twin climate and biodiversity crises, the protection and restoration of logged tropical forests is pivotal. Woody, climbing plants (lianas) restrict the recovery of logged tropical forests, but their removal can restore timber and carbon value. While liana removal is employed to restore logged forests in several tropical countries, the efficacy, application, and monitoring of this technique to track the outcome for timber and carbon require further consideration before it can be rolled out widely. In this thesis I, firstly, use meta-analytic techniques to quantify the overall effect of liana removal on the growth of trees and aboveground biomass, and to explore the drivers of variation in efficacy. I find that liana removal significantly enhances tree growth and nearly doubles aboveground biomass accumulation, but the Neotropical bias in liana removal studies prevents me from drawing meaningful conclusions about the causes of variation in liana removal efficacy. Secondly, I conduct a novel liana removal experiment across 320 ha of logged forest in Sabah, Malaysian Borneo, in which I remove varying proportions of lianas. I acquire satellite data across this experiment and find that liana removal can be detected using Normalized Burn Ratio – a vegetation index based on spectral reflectance that differentiates leaf from non-photosynthesising material. In this chapter I also provide the first experimental evidence that partial removal has a smaller impact on the canopy, potentially minimising negative impacts on biodiversity. Finally, I find that satellite data can also detect commercial-scale liana removal (applied across 17,000 ha in Sabah). Overall, my thesis demonstrates the significant potential of liana removal as a restoration action to enhance timber and carbon in logged tropical forests and develops a simple remote sensing method to validate the application and monitor the influence of large-scale liana removal on the canopy. However, much is yet unknown about liana ecology and the myriad impacts of liana removal on biodiversity and forest function, so I urge further research into these questions and strongly recommend that at least 20% of lianas are retained if liana removal is rolled out to restore logged forests across the tropics. Further work should also focus on operationalizing the use of remote sensing for monitoring.

CHAPTER 1: General introduction

Global importance and degradation of tropical forests

Tropical forests present a dilemma: they are fundamental to the functioning of our planet (Bruijnzeel, 2004; Devaraju et al., 2015), yet their future is seriously threatened (Malhi et al., 2014). Covering just 10% of global land area, tropical forests hold over half of the world's biodiversity (Barlow et al., 2018; Pillay et al., 2022) and over half of the carbon stored in the world's forests (Pan et al., 2011). Hence, these forests hold even greater value today in a world that is facing both the climate change crisis (Solomon et al., 2009) and catastrophic rates of biodiversity loss (Ceballos et al., 2015). Tropical forests are also a quandary as they provide natural resources and ecosystem services that support over a billion people worldwide (Lewis et al., 2015), yet humans are the key driver of declines in this ecosystem (Asner et al., 2009; Gibbs et al., 2010; Malhi et al., 2014). Despite their importance, in 2017 it was estimated that only 20% of tropical forests remain intact (Potapov et al., 2017), and the over-exploitation for natural resources and loss and fragmentation of tropical forests does not seem set to change (Lewis et al., 2015).

Human activities have myriad impacts on tropical forests, and land-use change is the biggest threat (Asner et al., 2009; Hansen et al., 2013). Agricultural expansion is the leading driver of tropical deforestation, causing the loss of 0.4% of forests per year between 1980 and 2020 (Gibbs et al., 2010), but this thesis studies the insidious impact of selective logging that degraded roughly 14 times the area deforested between 2000 and 2005 (Asner et al., 2009). In this thesis, I define forest degradation as activities that change the structure, faunal and floral composition, or function (such as tree growth or carbon storage) of a forest away from old-growth conditions, but do not necessarily deforest an area. Timber harvesting is no new phenomenon, but industrialized, large-scale logging has engulfed tropical forests over the past century (Edwards et al., 2014; Shearman et al., 2012). In 2011, 400 million ha of tropical forests were classed as production forests (Blaser et al., 2011), and this is likely to increase with the growing global demand for timber (Malhi et al., 2014).

Selective logging and the value of logged forests

Selective logging is one of the most common land-uses in the tropics, affecting 20% of the humid tropical forests – 'rainforests' that represent over half of all tropical forests – between 2000 and 2005 (Asner et al., 2009). Selective logging is the harvest of stems from particular tree species and that are above a minimum cutting diameter, rather than the removal of all stems that is carried out in clear-cutting (Edwards et al., 2014). While selective logging allows forest cover to remain, there is still a substantial impact on the plant and faunal communities and timber and carbon storage (Baraloto et al., 2012; Edwards et al., 2011; Putz et al., 2012a). Modifications to selective logging, such as lower harvest volumes (Burivalova et al., 2014) and reduced impact logging practices (Bicknell et al., 2014; Griscom et al., 2019; Miller et al., 2011; Pereira Jr et al., 2002), can minimise these impacts, but logging practices are still largely unsustainable since successive rounds of logging are too quick for full timber recovery (Putz et al., 2022; Shearman et al., 2012). Consequently, timber harvesting is often a "boom and bust" process, where-by logged forests eventually become financially unviable, leading logging companies to expand timber harvesting into pristine forests and leaving an expanding scar of environmental destruction in their wake (Shearman et al., 2012). Reducing the contribution of timber harvesting to environmental issues including the biodiversity and climate crises, therefore, requires improved sustainability of logging practices that will increase timber recovery and decrease the area of land degraded by timber harvesting (Cerullo and Edwards, 2019; Gibson et al., 2011; Putz et al., 2022).

While reducing the area of land affected by logging is important, the huge area of forests that have already been degraded by logging (i.e. that have altered structure, composition, or function) also represents a significant opportunity to protect biodiversity and address global carbon emissions (Edwards et al., 2014; Philipson et al., 2020). Pristine forests are irreplaceable in terms of biodiversity and carbon (Gibson et al., 2011; Pan et al., 2011), but logged forests have substantial recovery capacity (Cook-Patton et al., 2020; Gourlet-Fleury et al., 2013; Rutishauser et al., 2015) and still retain crucial biodiversity and ecosystem services (Edwards et al., 2011; Putz et al., 2012b). Despite this, logged forests are threatened with conversion to non-forest land-uses that have greater financial value (Burivalova et al., 2020) but far worse consequences for biodiversity and carbon (Edwards et al., 2014; Lewis et al., 2015). Consequently, restoring logged tropical forests towards pre-logging structure, faunal and floral composition, and function may prevent the rampant conversion of logged tropical forests, protecting and enhancing the biodiversity and ecosystem services of logged tropical forests (Burivalova et al., 2020; Cerullo and Edwards, 2019; Harrison et al., 2020).

Restoration of logged tropical forests

Concerns about ecosystem degradation and its consequences have prompted an era of ecosystem restoration. This is exemplified by numerous global restoration initiatives, including the UN declaring 2021-2030 as the Decade on Ecosystem Restoration (United Nations Environment Agency, 2019). Programs exist to restore ecosystems ranging from freshwater to grassland (Harper et al., 2021; Török et al., 2021), including the Bonn Challenge and REDD+ that focus specifically on restoring hundreds of millions of hectares of forests in the coming years (Strassburg et al., 2020).

Restoration is generally defined as assisting the recovery of an ecosystem that has been degraded and often relates to bringing functions back to the land, such as biodiversity or other ecosystem services (Benayas et al., 2009; Martin, 2017). In the context of tropical forests, restoration often aims to restore the characteristics of old-growth forest – the condition of the forest prior to human disturbance (Crouzeilles et al., 2016). However, there are multiple ways to define old-growth forests, such as by forest structure or species diversity, multiple ways to restore degraded forests, and the full restoration of all ecological attributes of pre-disturbed forests is difficult (Crouzeilles et al., 2016; Lamb et al., 2005). Passive restoration, for example, allows degraded forests to recover forest structure and biodiversity naturally, but can be slow, requires specific conditions that allow for the recruitment of old-growth species, and may have limited success if perceived as land abandonment (Cerullo and Edwards, 2019; Ren et al., 2017; Zahawi et al., 2014). Active restoration methods, on the other hand, such as planting seedlings of old-growth tree species into logged forests, can enhance the speed and chance of reaching pre-logging floral composition, but can be prohibitively expensive (Benayas et al., 2009; Cerullo and Edwards, 2019; Ren et al., 2017).

In selectively logged forests, restoration could alternatively aim to restore timber and carbon stocks (Cerullo and Edwards, 2019; Philipson et al., 2020). While this does not explicitly focus on recovering old growth species composition, this could restore vegetation structure while the enhanced timber and carbon value of the forest could provide a reason to maintain the land as forest and reduce the expansion of logging into pristine forests, both of which ultimately benefit biodiversity (Edwards et al., 2014). Enhancing tree growth and carbon storage in selectively logged forests is the primary aim of restoration throughout this thesis. Post-logging silvicultural practices, such as removing competing trees, can be employed to encourage the growth of commercial timber species (Peña-Claros et al., 2008a) and an emerging method with huge potential to enhance timber growth and carbon storage in logged tropical forests is the removal of woody climbing plants (referred to herein as lianas or climbers) (Cerullo and Edwards, 2019; César et al., 2016; Marshall et al., 2016)..

Liana-tree competition and liana removal

Lianas, defined in the section above as woody, climbing plants, are often described as structural parasites (Visser et al., 2018) as they exploit the trunks of trees to reach sunlight in the canopy rather than investing in their own supportive tissue (Schnitzer and Bongers, 2002). Because they do not form thick trunks, lianas grow much faster than trees, especially in the increased light environment created by logging and other disturbances (Schnitzer and Bongers, 2002; Schnitzer and van der Heijden, 2019). Lianas quickly reach the canopy and allocate more resources into producing large quantities of leaves, competing heavily with trees for light as a result (Putz, 1983; Song et al., n.d.). Research also shows that lianas have alternative water use strategies to trees, allowing them to continue growing in drier conditions during which trees tend to have lower growth rates (Schnitzer and van der Heijden, 2019), resulting in greater liana abundance in seasonally dry tropical forests (Toledo-Aceves, 2014).

Given their biology, it is no surprise that lianas negatively impact numerous aspects of tree performance, including tree growth and fruit production (Estrada-Villegas and Schnitzer, 2018; Grauel and Putz, 2004; Kainer et al., 2014; Schnitzer and Carson, 2010). When they reach high abundances, for example after logging activity, lianas ultimately stunt forest recovery (Marshall et al., 2020; Tymen et al., 2016). Moreover, lianas contribute less to forest carbon stocks than trees since they invest more in foliage than carbon-dense stems, meaning that lianas also reduce the recovery of carbon in logged forests (van der Heijden et al., 2015). Removing lianas, therefore, has potential as a restoration technique in logged tropical forests by enhancing tree growth, timber volume, and carbon stocks towards pre-disturbed level.

Aside from the ecological effects of liana removal, this method has strong restoration potential since cutting lianas is low-cost, does not require specialised skills or equipment, and is already implemented as part of reduced impact logging (RIL), albeit inconsistently (Marshall et al., 2016; Mills et al., 2019; Philipson et al., 2020; Sist, 2000). Hence, liana removal has the potential to be applied quickly to restore timber and carbon stocks across logged tropical forests, and is being trialled and implemented in various countries (Mills et al., 2019; Peña-Claros et al., 2008a; Sabah Forestry Department, 2020). However, refinement of liana removal treatment, consideration of potential negative consequences of removing lianas, and development of methods to monitor the impact of liana removal on carbon sequestration and timber growth are required before this method can be operationalized to restore large expanses of tropical forests.

Refining liana removal

Lianas are undoubtably problematic to adult trees, but they are also a critical component of the highly complex tropical forest system: something that should be considered when implementing liana removal. Lianas produce large quantities of leaves and fruits that are an important food source, for example, whilst their leaves can be used as nesting resources (Arroyo-Rodriguez et al., 2015; Odegaard, 2000; Putz et al., 2001). The woody stems of lianas also assist animal locomotion and, in the context of degraded forests, could connect faunal species to less disturbed forests (Arroyo-Rodriguez et al., 2015). Lianas also help to maintain a closed canopy, buffering against extreme temperatures that is crucial for species that rely on a dark and cool understory microclimate (Meunier et al., 2021c; O'Brien et al., 2019; Rodríguez-Ronderos et al., 2016; Scheffers et al., 2014). Moreover, lianas are a highly biodiverse group of plants, contributing around 20% of the rich biodiversity of tropical woody plants (Schnitzer and Bongers, 2002). The complete removal of lianas, therefore, could have catastrophic consequences for biodiversity in the tropical forest biome that is crucial for conservation. The only direct evidence of the effect of liana removal on faunal diversity found that insectivorous birds had lower abundance for 20-months post-liana removal, and some studies surmise that lianas could aid restoration through reducing edge effects or aiding faunal dispersal (Campbell et al., 2015; Magnago et al., 2017). To ensure that liana removal synergistically restores carbon and timber stocks and benefits biodiversity (a crucial step for policy as discussed in Pettorelli et al., 2021), the technique needs to be refined to ensure that an ecologically functional subset of lianas remain in the forest.

The adoption of liana removal as a restoration technique is also partly limited by variability in the efficacy of the technique. A recent review found that liana removal increased the growth of adult trees by between a quarter and three times (Estrada-Villegas and Schnitzer, 2018), while a study in Malaysian Borneo showed a decrease in tree growth (O'Brien et al., 2019). This variability is perhaps unsurprising given the complex ecology of lianas, for example that the competition between lianas and trees can vary with climate, season, and species (Schnitzer and van der Heijden, 2019; Venegas-Gonzalez et al., 2020), but there is currently no consensus as to what drives the variation in removal efficacy. Individual studies that are conducted in different field sites using an array of liana removal methods will struggle to ascertain the causes of variation in liana removal efficacy, but it is crucial to know the circumstances under which liana removal is most (or not) effective.

Remote sensing and liana removal

Another challenge facing the wide-spread adoption of liana removal, and ecosystem restoration in general, is how to validate and monitor actions applied over large scales (Murcia et al., 2016). Here, I define “validation” as a process to verify that liana removal, or other methods, has been applied, and “monitoring” as a process to assess the impact of liana removal on the canopy, carbon sequestration, and timber growth. While commitments to restore hundreds of millions of hectares of forests (Strassburg et al., 2020) and providing payments for the resulting additional carbon storage (GOFC-GOLD, 2016) are highly commendable, accessing these payments, tracking progress towards restoration targets, and determining restoration efficacy all require validation and monitoring across large areas (Deluca et al., 2010; Holl and Cairns, 2010; Murcia et al., 2016). Studies have developed optimal field sampling designs to monitor restoration projects (Londe et al., 2022; Viani et al., 2018) but this only monitors a sample of the total treated area and, given the vast expanse of degraded tropical forests that are targeted for restoration, conventional field methods still present logistical challenges.

Remote sensing, defined as the acquisition of data via non-contact recording (Wang et al., 2010), has become increasingly linked with conservation over the past few decades (Pettorelli et al., 2014; Turner et al., 2003), and recent advances show its potential for restoration monitoring (de Almeida et al., 2020). For example, satellite data, which can be acquired across the entire globe up to several times per day at 3 m resolution (Shendryk et al., 2019), has been used to detect selective logging (Hethcoat et al., 2020), track land-use change (Vancutsem et al., 2021), and assess the impact of logged forest restoration (Wu et al., 2020), evidencing that forest management can be monitored using such data.

There have been simultaneous advances in the use of remote sensing to study lianas, largely based on the light reflectance properties of this plant group (Castro-Esau et al., 2004; Chandler et al., 2021b; Meunier et al., 2021c; Waite et al., 2019), and a recent review calls for further use of remote sensing to study lianas (van der Heijden et al., 2022). Consequently, there is clear potential for satellite imagery to facilitate low-cost, temporally and spatially detailed monitoring of large-scale restoration of logged tropical forests through the removal of over-abundant lianas. Validating the large-scale application of liana removal would be particularly useful, facilitating the inclusion of liana removal projects in schemes that provide payments for restoration, such as REDD+ (GOFC-GOLD, 2016). The enhanced monitoring capacity provided by remote sensing could also further evidence the beneficial impacts of liana removal and improve the efficacy and application of the treatment, ultimately encouraging more land managers to adopt liana removal as a sustainable logging and restoration practice. Remote sensing could also reveal information about liana ecology, addressing some of the many unanswered questions about lianas that we need to consider before large-scale liana removal can be rolled out widely.

Thesis aims and objectives

The overarching aim of this thesis is to improve our understanding, the application, and the monitoring of liana removal in logged tropical forests, demonstrating the potential for liana removal in sustainable logging and the restoration of carbon storage and timber growth in huge expanses of critically important logged tropical forests. Specifically, I aim to a) quantify the overall efficacy and variability of liana removal for enhancing timber and aboveground biomass growth, b) assess a novel liana removal method that could minimise the detrimental impacts on biodiversity, and c) develop a remote-sensing method that can detect and monitor large-scale liana removal.

Chapter 2 – Removing climbers more than doubles tree growth and biomass in degraded tropical forests

Huge expanses of tropical forests have been degraded and the recovery of these forests can be inhibited by super-abundant woody climbing plants, also known as lianas or climbers. While experimental evidence shows that climber removal largely increases tree and aboveground biomass growth, there is substantial variation in the efficacy of this method. This chapter uses meta-analytic techniques to determine the magnitude and variation in the efficacy of climber removal. I quantify the overall effect of climber removal on enhancing tree growth and biomass accumulation, estimate the potential contribution of climber removal to global carbon sequestration, and explore the drivers of variation in climber removal efficacy.

Chapter 3 – Monitoring lianas from space: Using Sentinel-2 imagery to detect liana removal in logged tropical forests

Ecosystem restoration is recognised as a key global priority and logged tropical forests represent a huge area with restoration potential. Liana removal could be introduced widely to restore tree growth and carbon sequestration in such forests but validating and monitoring large-scale restoration is difficult and liana removal could have negative consequences for biodiversity. In this chapter I combined a novel field experiment, removing varying intensities of lianas, with Sentinel-2-derived vegetation indices to determine whether varying intensities of liana removal can be detected with satellite data and to ascertain whether partial removal minimises the impact of this technique on the canopy.

Chapter 4 – Commercial-scale liana removal detected using satellite data

Improving the sustainability of logging has the potential to address both the climate and biodiversity crises. Enhancing timber recovery through the removal of lianas presents one opportunity to achieve this, but monitoring application over vast areas is a challenge. This chapter builds on the results of Chapter 3, aiming to determine whether commercial-scale liana removal can be detected using satellite data and to identify any drivers of variation in this signal.

CHAPTER 2: Removing climbers more than doubles tree growth and biomass in degraded tropical forests

This thesis chapter has been published as:

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A correction to this chapter has been published at https://doi.org/10.1002/ece3.9917 and is copied below. Please consider this when reading the original chapter as I have not amended the original chapter to avoid discrepancies with the published article. In particular, please note the updated interpretation of the SMD effect size, the new log response ratio method used to calculate the percentage effect of liana removal on carbon sequestration, and the resulting updated global potential of liana removal for biomass accumulation.

CORRECTION:

In the article by Finlayson et al. (2022), titled “Removing climbers more than doubles tree growth and biomass in degraded tropical forests”, the authors note an error. The paper misinterprets the standardized mean difference (SMD) summary effect size, resulting in an overestimation of the effect of climber removal on tree growth and biomass and an overestimation of the global carbon sequestration potential of climber removal. This correction finds that climber removal almost doubles aboveground biomass (AGB) accumulation in degraded tropical forests rather than tripling AGB.

Throughout the article, SMD should be interpreted as the number of standard deviations difference between the tree growth and AGB in treated and control forest plots rather than the proportional effect of climber removal. For example, the summary SMD for tree growth of 1.56 means tree growth is 1.56 standard deviations higher in treated than control plots, rather than a 2.56-fold (or 256%) increase in tree growth, and the summary SMD for AGB of 2.09 means AGB is 2.09 standard deviations higher in treated than control plots, rather than a 3.09-fold (or 309%) increase.

In this correction, we use the log response ratio (logRR) between climber removal treated and control plots rather than the SMD to calculate the proportional effect of climber removal on biomass accumulation and the global carbon sequestration potential of climber removal. We calculate a summary log response ratio (logRR) of 0.63, equating to an 88% increase in biomass accumulation (95% CI = 40%–145%) in climber removal relative to control plots once logRR is transformed back to a normal scale. Extrapolating this proportional effect to timber production and secondary forests across the tropics, we find that climber removal could sequester an additional 7.4 Gigatons of CO2 over a decade (4.1 in production forest and 3.3 in secondary forest) at a cost of US$0.59 and US$0.08 per Mg (metric ton) of CO2 sequestered over 10 years, respectively (range: US$0.01–US$ 1.19).

The overall conclusion of the paper remains the same: There is a significant and substantial positive effect of climber removal on tree growth and aboveground biomass compared with untreated forest stands.

Abstract

Huge areas of tropical forests are degraded, reducing their biodiversity, carbon, and timber value. The recovery of these degraded forests can be significantly inhibited by climbing plants such as lianas. Removal of super-abundant climbers thus represents a restoration action with huge potential for application across the tropics. While experimental studies largely report positive impacts of climber removal on tree growth and biomass accumulation, the efficacy of climber removal varies widely, with high uncertainty as to where and how to apply the technique. Using meta-analytic techniques, we synthesise results from 26 studies to quantify the efficacy of climber removal for promoting tree growth and biomass accumulation. We find that climber removal increases tree growth by 156% and biomass accumulation by 209% compared to untreated forest, and that efficacy remains for at least 19 years. Extrapolating from these results, climber removal could sequester an additional 32 Gigatons of CO_2 over 10 years, at low cost, across regrowth and production forests. Our analysis also revealed that climber removal studies are concentrated in the Neotropics (N=22), relative to Africa (N=2) and Asia (N=2), preventing our study from assessing the influence of region on removal efficacy. While we found some evidence that enhancement of tree growth and AGB accumulation varies across disturbance context and removal method, but not across climate, the number and geographical distribution of studies limits the strength of these conclusions. Climber removal could contribute significantly to reducing global carbon emissions and enhancing the timber and biomass stocks of degraded forests, ultimately protecting them from conversion. However, we urgently need to assess the efficacy of removal outside the Neotropics, and consider the potential negative consequences of climber removal under drought conditions and for biodiversity.

Introduction

Around 300 million hectares of moist tropical forest were deforested or degraded between 1990-2020 (Vancutsem et al., 2021). Both forms of disturbance threaten biodiversity, erode carbon stocks in a biome that contributes 55% of the global forest carbon sink, and reduce future timber yield, the main economic incentive for maintaining managed forests (Fisher et al., 2011b; Gibson et al., 2011; Pan et al., 2011; Putz et al., 2012b). While the protection of pristine ecosystems remains vital (Gibson et al., 2011), the enduring biological value of degraded forests emphasises the critical role of restoration in conserving biodiversity, reducing atmospheric CO_2, and supporting livelihoods (Edwards et al., 2014; Moomaw et al., 2019; Strassburg et al., 2020).

Various global initiatives, including the UN Decade on Ecosystem Restoration, the Bonn Challenge, and REDD+, recognise the benefits of restoration, with ambitions to restore hundreds of millions of hectares of degraded land (Cerullo and Edwards, 2019; Strassburg et al., 2020). However, 'restoration' encompasses different strategies with varying potential, from converting agricultural land back to forest, to enhancing the state of degraded forests, such as those produced by selective logging (Moomaw et al., 2019; Strassburg et al., 2020). While restoring forests to currently non-forested land has huge potential (Strassburg et al., 2020), this is unlikely to yield the carbon sequestration required in the immediate future to meet global goals. Reforestation can also compete with food production and urban expansion (Moomaw et al., 2019). Alternatively, restoring degraded tropical forests to help them achieve their full ecological potential could remove approximately 350 $PgCO_2$ from the atmosphere (Erb et al., 2018), recover timber stocks that prevents the expansion of 'boom-and-bust' timber harvesting into pristine forests (Burivalova et al., 2020), and reduce the risk of degraded land being converted to more lucrative, but lower carbon and biodiversity value agricultural plantations (Cerullo and Edwards, 2019).

A key remaining question is how best to restore degraded forests (Coleman et al., 2019), and how much climate mitigation potential can be delivered, given large uncertainty in existing estimates (Griscom et al., 2017). A variety of methods have been developed for overall restoration of biodiversity and productivity in degraded forests, from ‘natural restoration’ where human activity is simply removed, to enrichment planting where trees are planted to enhance natural restoration (Cerullo and Edwards, 2019). However, especially for enrichment planting, success and carbon gains can be limited, and interventions expensive (Burivalova et al., 2020; Philipson et al., 2020). An alternative solution is climber cutting. This method targets climbing plants such as lianas (woody, climbing plants) bamboo, and rattan that limit forest recovery. It is already widely recommended as part of reduced impact logging (RIL) practices, and is legally required but poorly implemented post-logging in Indonesia and other countries (Griscom et al., 2014; Putz et al., 2008; Ruslandi et al., 2017). Furthermore, climber cutting is relatively affordable (~ \$8.64 ha^{-1} across Africa and the Americas [see additional data] compared to enrichment planting (~\$1500-\$2500 ha^{-1} in Malaysian Borneo (Philipson et al., 2020)), requires limited expertise, can be easily integrated with forest inventories, and has potential to enhance forest restoration and carbon sequestration on a faster timescale (Cerullo and Edwards, 2019).

Climbing plants tend to proliferate extensively after disturbance and compete strongly with trees for light, water, and other resources, limiting tree growth, survival, recruitment and aboveground biomass sequestration (Meunier et al., 2021b; Schnitzer and Bongers, 2002). Estrada-Villegas and Schnitzer (2018) conclude that lianas have a negative impact on all metrics of tree performance, and it has been estimated that removing climbers in tropical forests enhances tree growth up to 372%, timber yield by 1.51 m^3 per tree over 40 years, and aboveground biomass by ~76% per year compared to untreated forest (Estrada-Villegas and Schnitzer, 2018; Mills et al., 2019; van der Heijden et al., 2015). However, these are site and region-specific studies that report varying climber cutting efficacy.

Compared to untreated controls, the efficacy of climber cutting ranges from reducing tree growth by 20-90%, depending on size class (O’Brien et al., 2019), to more than doubling it (Gerwing, 2001; Grauel and Putz, 2004), with little consensus on what drives this variation. Marshall *et al* (2017) noted that, across continents, tree growth after climber removal was enhanced by between 41-122% compared to control forest, but there is conflicting evidence regarding whether the outcome of climber removal on tree growth and carbon sequestration are influenced by region and climate. For example, two studies in SE Asia and Central America conclude that efficacy of cutting varies with total annual rainfall and between wet and dry seasons, while other studies find similar efficacy in wet and dry seasons (Álvarez-Cansino et al., 2015; O'Brien et al., 2019; van der Heijden et al., 2019; Venegas-Gonzalez et al., 2020).

Climber removal is also applied in various intensities and across different forest types, spanning old growth, selectively logged, and secondary forests of various ages, with no 'best-practice' procedures yet defined. In some cases, climber removal is applied just once to selected focal trees (Grogan and Landis, 2009), while in others removal is applied to the entire stand with repeated treatments (van der Heijden et al., 2019). Again, results are conflicting: some studies find a greater impact of climber removal on tree growth in younger forest, in earlier successional species, and on larger trees as climber load tends to be greater in these contexts (De Lombaerde et al., 2021; Duncan and Chapman, 2003; Estrada-Villegas et al., 2020). Conversely, a recent study found no effect of liana removal on AGB accumulation across varying successional ages and tree sizes in a tropical dry forest (Estrada-Villegas et al., 2021).

Due to the range in efficacy, breadth of climber removal contexts, and limited systematic attempt to understand drivers of variation in treatment efficacy, it is difficult to anticipate the outcome of climber removal with accuracy. Not only is this problematic for land managers, but it also limits our ability to estimate the contribution that climber removal could have to global restoration and carbon sequestration goals.

In this study, we use meta-analytic techniques to determine the overall magnitude of climber removal efficacy in tropical forests, and to understand the potential drivers of variation in efficacy. We focus on tree growth and AGB accumulation as they contribute substantially to forest commercial value and productivity. We first synthesise existing experimental climber removal studies to quantify the effect of climber removal on enhancing tree growth and AGB accumulation, taking study context into account (Objective 1). We use this to estimate the potential contribution of climber removal to global carbon sequestration through restoration of degraded forests. Second, we exploit the breadth of study contexts to investigate whether region, climate, and forest disturbance context influence the efficacy of removal, to determine the best method of application, and to assess the longevity of treatment efficacy (Objective 2). Overall, this study determines whether climber removal can be applied to enhance aboveground biomass and timber stocks globally and, ultimately, restore function and economic value to degraded tropical forests.

Methods

1. Literature search and screening

We conducted literature searches in Web of Science (WoS), SCOPUS and Google Scholar, the latest search completed in March 2021. Author C.F. ran two search strings in each database: to find all studies that applied climber removal in tropical forests with any type of disturbance (none, regrowth after deforestation, and selectively logged), and to find studies that applied climber removal before disturbance (Table S4). We also conducted searches in the E-Theses online Service (EThOS) database, contacted academics known to work on climber removal, and contacted organisations including national forestry departments and the Centre for International Forestry Research (CIFOR). This yielded a further 8 studies. Due to the high number of irrelevant results returned by Google Scholar, we screened results for relevance against inclusion criteria set a priori (Table S5) directly from the webpage. We stopped searching Google Scholar when we reached 100 consecutive irrelevant results. All WoS and SCOPUS search results were screened.

The WoS, SCOPUS, relevant Google Scholar results, and the eight studies from other sources, yielded 5304 unique results. These were screened against the inclusion criteria, resulting in 65 studies (Figure S12). We then excluded 13 results that combined climber removal with another vegetation management, seven results that reused data from another publication, and six results that did not have a relevant tree growth or biomass metric (Table S6). A further 13 were excluded because mean tree growth, aboveground biomass (AGB), or control data were unavailable; authors were contacted for missing data before being excluded from the dataset. This resulted in 26 controlled experimental studies that assess the impact of climber removal on tree growth (Figure S12 and Table S7). For the

AGB analysis, we only included a subset of the 26 studies which measured the effect of climber removal on trees >= 5 cm dbh, resulting in 12 studies. To quantify removal efficacy we require treatment and control results for each study, contrasting to Estrada-Villegas and Schnitzer (2018) that qualitatively summarises 64 studies including non-controlled studies and other responses to climber removal, such as tree mortality and canopy openness.

2. Data extraction

Author C.F. recorded data to calculate effect size (mean tree growth or AGB accumulation across all trees measured in treatment and control plots, variation around the mean, sample size [number of treatment and control plots], and tree growth response metric), study details (e.g., sampling effort and experimental design), and explanatory variables relating to region and climate, forest disturbance context, and method of removal that could influence climber removal. C.F. verified data at the time of extraction for accuracy. See Supplementary Information, Appendix B for details of tree growth and AGB response data collection, and details of how missing data were handled, and Table S7 and our published additional data for metadata of each study included in the analyses.

3. Meta-analysis

3.1 Calculating individual effect sizes

We calculated the individual effect sizes (ES) (and variance) for each study using the standardised mean difference (SMD; Hedges *g*) in RGR or AGB between treatment and control sites using the *metafor* and *compute.es* R packages (Del Re, 2013; Viechtbauer, 2010). Multiple effect sizes were calculated per study if there were treatment vs control comparisons measured at more than one timepoint, or on different size classes of trees. SMD is less biased by small sample sizes than mean difference (MD) and there was no difference in the results using either method (Figure S17). See Del Re (2015) for equations to calculate SMD and variance.

A value of SMD greater than zero indicates greater growth or biomass accumulation in trees in plots that had climbers removed compared to trees in control plots: the larger the positive number the greater the impact of climber removal. A value of SMD not significantly different from zero indicates equal tree growth or biomass accumulation in treated and control plots, meaning that climber removal has no significant effect.

3.2 Assessing the magnitude of climber removal efficacy

To assess the magnitude of the effect of climber removal on promoting growth or biomass accumulation of trees (Objective 1), we fitted mixed-effects linear models (using *lme4* and *lmerTest* R packages: (Bates et al., 2015; Kuznetsova et al., 2017). One model was fitted to the 103 individual effect sizes from the 26 studies in the analysis of tree growth, and another to the 69 individual effect sizes from 12 studies in the analysis of biomass (Table S10). The models were run on each of the 10 datasets generated from imputing missing variances for growth and biomass (see Supplementary Information, Appendix B 'Missing data' for details). The model results presented in the manuscript are the average parameter coefficients (including intercept), standard error of the coefficient, degrees of freedom, coefficient confidence intervals, and p-values (based on these averaged values) from the 10 models. The models were weighted by the inverse SMD variance.

A unique study identifier was included as a random effect in both models to account for non-independence when there were multiple effect sizes from each study. Time of measurement after treatment, number of species measured in mean growth rate, and study quality were included as fixed effects to capture known sources of variation between effect sizes or studies (Spake et al., 2020). Study quality is an ordinal scale ("high", "medium", or "low" quality), assigned based on study design, sample size, sampling effort (sampling area or number of trees measured), whether the tree growth was relative (RGR), how far the treatment site was from control plot, and whether there were any disturbance differences between treatment and control forests (Table S8). Study quality was included as a fixed effect as it only has three categories, and allows us to account for the variation between studies in terms of their design and rigour. The 'number of species' variable accounts for variation caused by different studies measuring a different number of species, see Supplementary Information, Appendix B for more details.

We assessed the level of variation (heterogeneity) in the efficacy of climber removal using Q statistics and I^2 values. A significant Q statistic indicates significant heterogeneity, meaning that effect sizes from different studies vary more than would be expected by chance (Del Re, 2015; Harrison, 2011). The I^2 value indicates the extent of the heterogeneity, with 25% considered low, 50% considered moderate and 75% considered a high amount of heterogeneity (Del Re, 2015).

3.3 Assessing drivers of variation in climber removal efficacy

To determine whether region and climate, forest disturbance, or removal method were causing variation in climber removal efficacy (Objective 2), we added explanatory variables to the two models described previously. For the tree growth analysis, we included variables with the greatest theoretical impact on the outcome of climber removal (Table S10). The direction and size of the coefficient for each variable indicated its influence on climber removal efficacy. Several parameters could not be assessed (Table S9), or were assessed in supplementary models (Table S11), due to data constraints.

For the analysis of AGB, we were only able to assess the influence of a few parameters relating to removal method and disturbance context due to data constraints, and used three separate models to do so. We present all three models in the main text (see details in Table S10). Objective 2 models for tree growth and biomass accumulation were run for all imputed datasets (see Supplementary Information, Appendix B 'Missing data' for details), and model results herein show the average parameter coefficients, standard error of the coefficient, degrees of freedom, coefficient confidence intervals, and p-values (based on these averaged values). We assessed the heterogeneity of the objective 2 models using Q and I^2 statistics.

4. Sensitivity analysis and assessing publication bias

We tested for publication bias in several ways. Firstly, we analysed the relationship between publication year and effect size to infer whether datasets with results opposing that of the first published paper remain unpublished. Secondly, we tested for asymmetry in funnel plots with Eggers test, using the *metafor* R package (Viechtbauer, 2010).

To test the robustness of the results, we calculated fail-safe numbers following the Rosenthal, Rosenberg and Orwin methods, using the *metafor* R package (Viechtbauer, 2010). These indicate how many studies with null results would need to be added to the analysis to reduce the significance level of the summary effect size so that it was no longer significant, or to reduce the effect size by half. Larger numbers indicate the effect size is robust.

5. Global carbon sequestration potential

To determine the potential contribution of climber removal to global carbon sequestration, we extrapolated the effect of climber removal on AGB accumulation (intercept of model for objective 1.2) to an assumed maximum scenario. This includes: a) the area of natural tropical forest managed for selective timber harvest with a valid concession license (FAO, 2020), and b) the area of moist tropical forest regrowing >3 years after deforestation (Vancutsem et al., 2021). We calculated the difference between the baseline AGB growth rate for the forest type and the climber removal enhanced AGB growth rate. We then subtracted the AGB lost in removed climbing plants and their annual biomass growth, and converted the final difference in AGB to tons of CO_2 (IPCC, 2003). See Table 3 and additional published data for full details.

All analyses were conducted in R (R Core Team, 2020) and figures produced using the R package *ggplot* (Wickham, 2016).

Results

Global distribution and details of study sites

The 26 studies included in the analysis of tree growth are distributed across eight countries in the tropics, plus one in subtropical Argentina (-26 degrees latitude) (Figure 1). While there is good representation in Central and South America (22 studies), there were limited studies from Asia (2) and Africa (2). The 12 studies in the biomass analysis are from five countries, mainly in Central and South America (11 studies), plus Asia (1), with none in Africa.

The studies cover a range of elevations (range: 13-776 m.a.s.l), and gradients of precipitation (1144-2964 mm year^{-1}), temperature (21.2-27.7°C), and dry season length (0-7 months). There were three studies in sites without any disturbance, 13 had been selectively logged, seven were forests regrowing after being cleared (secondary forest), and three were forests regrowing after being cleared that had also been selectively logged. Cutting was applied 1-720 months after disturbance in studies removing climbers post disturbance, and 1-12 months before removal for studies applying climber removal pre disturbance. Study monitoring duration ranged from 12-228 months post treatment. Studies repeated climber removal between 0-27 times, and across entire plots or just on focal trees. See Table S7 and additional published data for full study metadata.

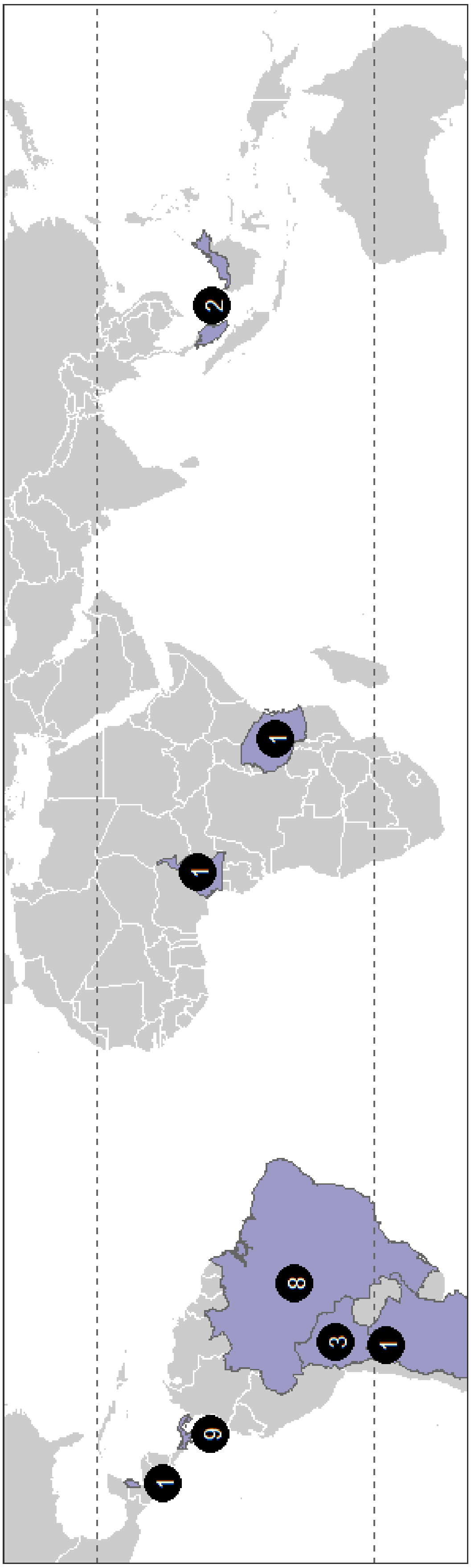

Figure 1: Geographical distribution of the 26 studies across the tropics included in the meta-analysis literature search. A subset of these is included in the biomass analysis. Black circles indicate number of studies in each country. Dashed horizontal lines indicate the Tropic of Cancer (23° N) and the Tropic of Capricorn at (23° S).

Effect of climber removal on tree growth

We find that the results of our meta-analysis are robust, even though there is some evidence of publication bias (see Figures S13-S16 and Supplementary Information, Appendix D for details). Trees in plots from which climbers were removed experienced a 2.56-fold increase in growth (summary effect size 156%; 95% CI = 109-203%) compared to those in untreated control plots (Figure 2, Table 1) across all tree size classes and various growth metrics combined. This represents the tree growth enhancement resulting from climber removal at the stand level. There was substantial variation in the effect on tree growth: the lowest individual effect size across studies showed a -36% decrease in tree growth, whereas the highest showed a 409% increase in growth. African studies had effect sizes of -36% and 12%, and Asian studies had effect sizes of 56% and 179% compared to untreated controls (Lussetti et al., 2016; Marshall et al., 2016; O'Brien et al., 2019; Parren, 2003), respectively; Figure 2). The median effect size outside the Neotropics (29%) is much lower than the overall tree growth effect size (156%), but we could not directly assess the influence of region due to insufficient studies located in Asia and Africa (see Methods section 3.3).

Q statistics and I^2 values indicate that the magnitude of the positive effect of climber removal on enhancing tree growth is expected to vary, but only by a small amount ($Q = 164$, 95 CI = [121-218], p-val <0.001; I^2= 38%, 95% CI = [16-53%]). Model results did not differ substantially if we excluded imputed data, if we calculated effect sizes using MD rather than SMD (Figure S17), nor if we removed van der Heijden et al (2015) that had an effect size almost double those of the other studies (Table S12).

The efficacy of climber removal for enhancing tree growth varied with quality of study: efficacy was 122% greater (95% CI = [44, 201]) in high- than low-quality studies, and 118% greater (95% CI = [88, 149%]) in high- than medium-quality studies (Table 1). We observed that the efficacy of climber removal for enhancing tree growth did not vary with the number of species in the mean growth rate (Table 1).

Effect of climber removal on AGB accumulation

Climber removal increased total aboveground biomass storage of all trees in treated plots by 3.09 times (summary effect size 209%; 95% CI = [60, 359 %]) compared to untreated controls. This represents the increased AGB accumulation resulting from climber removal at the stand level. Again, there was substantial variation, with the individual effect size sizes across studies ranging from -42 to 466% (Figure 2, Table 1). The only study outside the Neotropics (in Malaysia) experienced 51% increase in tree growth compared to untreated controls. The effect size was much lower and the credible intervals cross zero when imputed data is not included (N=9) (Figure S18), but only one study of nine had a negative effect of climber removal on biomass, confirming the overall positive effect of climber removal on biomass accumulation. Q statistics and I^2 values indicate that, while we expect a positive effect of climber removal, the magnitude of the effect of climber removal on AGB accumulation is likely to vary substantially (Q = 257, 95 CI = [150, 371], p-val <0.001; (I^2= 74%, 95% CI = [55, 82 %]).

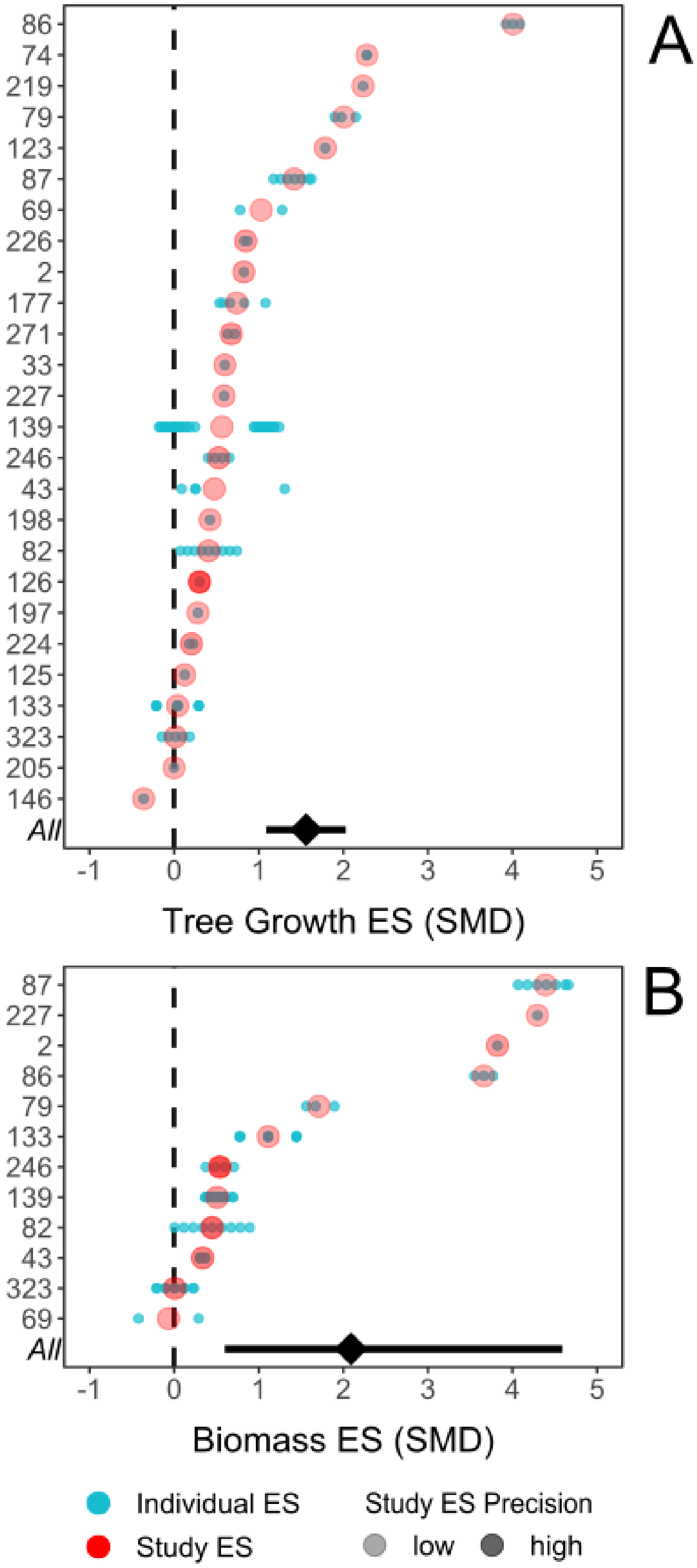

Figure 2: Overall, individual, and study average effect sizes (ES) of climber removal for promoting tree growth (Panel A) and AGB accumulation (Panel B). Numbers on the y-axis represent study ID, as given in Table S7, and metadata spreadsheet in our published additional data. Blue dots are individual

effect sizes within a study, predicted from the models for Objective 1.1 and 1.2 and averaged for all imputed datasets. Red circles are the study ES (the average of the individual ES for each study); the shade of the circle represents precision of the study ES and is proportional to the inverse of the variance of the individual effect sizes, averaged by study. The black diamond at the bottom of each figure is the overall summary effect size of climber removal for promoting tree growth and biomass, taken from the intercept of the models for Objective 1.1 and 1.2 when continuous covariates are at their mean value and study quality reference level is "high"; error bar shows 95% credible intervals.

Table 1: Magnitude and direction of climber removal efficacy on tree growth and biomass accumulation. Results of models for Objective 1.1 (tree growth) and Objective 1.2 (AGB). 'Tree growth Effect Size (ES)' and 'AGB Effect Size (ES)' are the intercept of each model and show the number of times greater tree growth or biomass accumulation with climber removal versus untreated control plots. Results are the average of 10 Linear Mixed Models using 10 datasets imputed using linear regression, including the study with just post-treatment data (Tree growth N=26 studies, Biomass N=12 studies). See Supplementary Information, Appendix C for full description of models. Bolded effect sizes indicate level of significance at either 0.05,0.01, or 0.001.

Objective	***Fixed effect***	***Estimate (SE)***	***Degrees of Freedom***
Objective 1.1: Tree growth	***Tree growth ES***	**1.56 (0.23)*****	32
	Study quality High:Low	**-1.22 (0.40)****	81
	Study quality High:Med	**-1.18 (0.15)*****	86
	Number of species	0.00 (0.00)	89
	Time elapsed since removal	**0.01 (0.00)*****	90
	AGB ES	**2.09 (0.67)***	11

Objective 1.2: AGB accumulation	*Study quality High:Low*	-1.97 (1.76)	7
	Study quality High:Med	-0.23 (0.41)	61
	Number of species	-0.00 (0.01)	8
	Time elapsed since removal	**0.01 (0.00) ***	54

** < 0.05, ** < 0.01, *** < 0.001*

Drivers of variation in efficacy for tree growth

Explanatory variables relating to climate, region, and forest disturbance did not influence the efficacy of climber removal for enhancing the growth of trees (Figure 3, Table 2). However, efficacy did increase, marginally, per month since treatment (1% greater effect on tree growth per month in objective 1.1 and 2.1 models (95% CI = [0, 1%]); Table 1 and 2). This shows that climber removal enhances tree growth for at least the maximum study monitoring period of studies in this analysis: 19 years. The model for objective 2.1 found that studies which repeated removal had 41% less tree growth enhancement compared to studies which did not repeat removal (95 % CI = [1, 82 %]; Table 2). However, the confidence intervals are very close to zero and the supplementary models suggest that repeating removal does not significantly influence the efficacy of climber removal for enhancing tree growth (Table S13). Supplementary models also found no effect of latitude, time between disturbance and removal, and dry season temperature and precipitation on the efficacy of climber removal for promoting tree growth.

As with objective 1.1, the Q statistics and I^2 values indicate that the positive effect of climber removal on tree growth is still likely to vary by a small amount, even when accounting for variation due to parameters included in the model for objective 2.1 (Q = 177, 95 CI = [132, 232], all p-values <0.001; I^2 42%, 95% CI = [23, 56%]).

Drivers of variation in efficacy for AGB accumulation

The AGB accumulated in treated plots relative to untreated plots increased with the time elapsed since removal, the number of times the treatment was applied, and the amount of time between disturbance and initial application of removal (Table 1 and 2, Figure 3). The efficacy of climber removal for enhancing AGB increased 0.1% (95% CI = [0.0, 1.2%]) with each month elapsed since removal. This shows that climber removal enhances AGB for at least 10 years: the maximum study monitoring period of studies in the biomass analysis. We also found that removal more greatly enhanced biomass accumulation in older secondary forest and forests logged longer ago: efficacy increased by 115.9% (95% CI = [29.7, 202.0%]) with each additional year between disturbance and treatment (maximum 60 years between disturbance and treatment). Efficacy also increased by 18% with each removal repetition (95% CI = [9, 28%]).

According to the Q statistics and I^2 values, the positive effect of climber removal on AGB accumulation is still expected to vary substantially, even when accounting for variation due to parameters included in the models for objective 2.2 (Q = 239-269, 95 CI = [132-383], p-val <0.001; I^2= 65-68%, 95% CI = [41, 84%]; across objective 2.2 a, b, and c models).

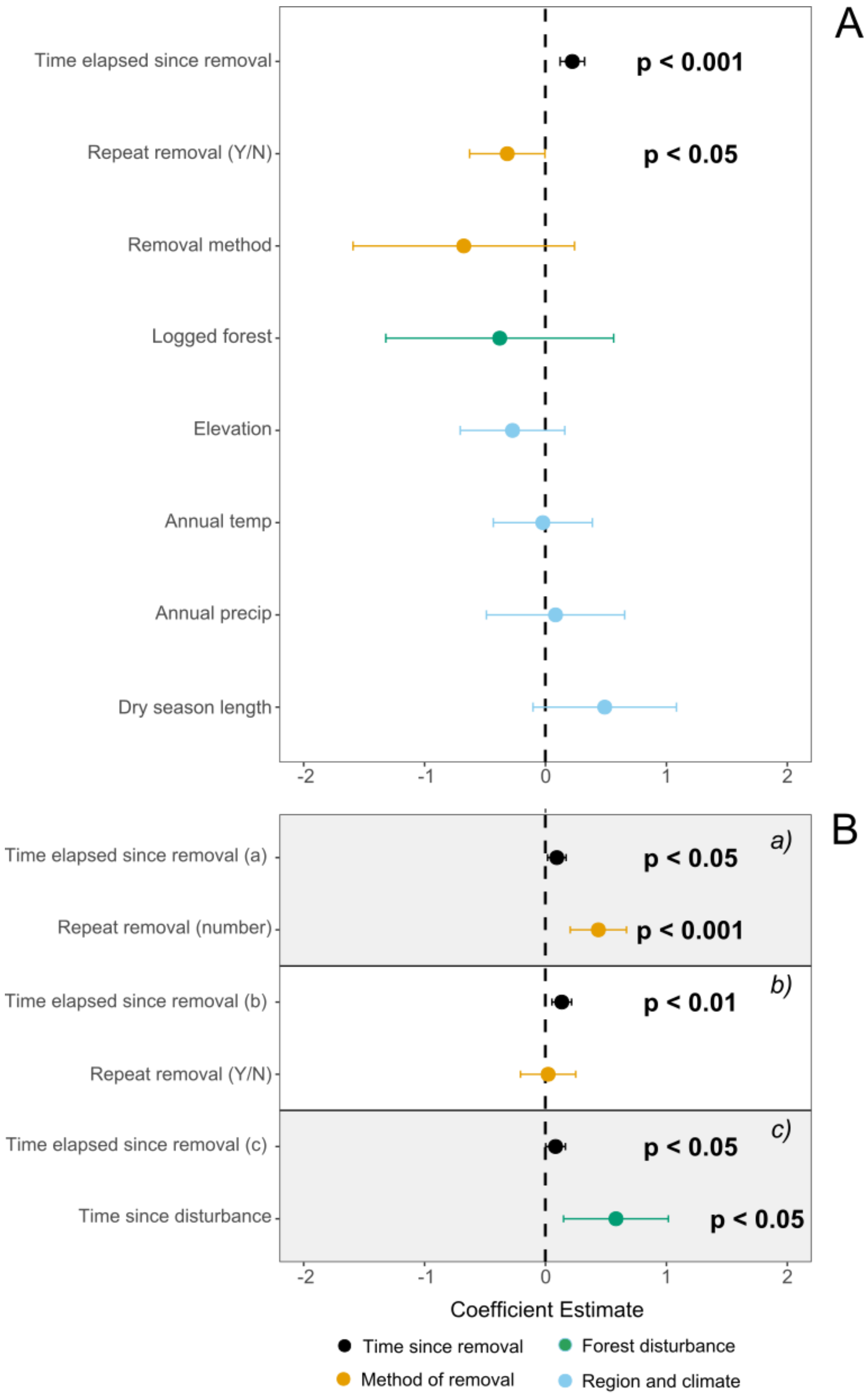

Figure 3: Influence of region and climate, disturbance context, and method of removal (whole plot vs focal tree removal and whether removal was repeated) on the efficacy of climber removal for promoting tree growth and AGB accumulation. Panel A shows coefficient estimates for the objective

2.1 (tree growth) model and Panel B shows estimates for the objective 2.2 (AGB) models a), b) and c). The coefficient for the repeat removal (number) is excluded from model 2.2 c) as it was no different from model a). Centred and scaled parameter estimates are shown for continuous variables with error bars indicating 95% CI. For categorical variables, the figure shows the fitted mean value with 95% CI between the reference level and the other categorical level. The reference level for the ‘Logged forest’ variable is ‘logged’, ‘Repeat removal (Y/N)’ variable is ‘no repeated removal’, and ‘Removal method’ variable is the whole plot removal method. Significant parameter estimates are shown with p-values. Colour indicates the parameter category.

Table 2: Drivers of variation in the efficacy of climber removal for tree growth and AGB accumulation. Results for objective 2.1 and 2.2 models, averaged from 10 Linear Mixed Models using 10 imputed datasets (imputed using linear regression), and including one study with just post-treatment data (tree growth N=26 studies, biomass N=12). Response variable is tree growth for Objective 2.1 and AGB change for Objective 2.2, see full model details in Supplementary Information, Appendix C. Bolded explanatory parameters indicate level of significance at either 0.05,0.01, or 0.001.

Objective	***Explanatory parameter***	***Estimate (SE)***	***Degrees of Freedom***
Objective 2.1 (Tree growth)	***Time elapsed since removal***	**0.01 (0.00)*****	86
	Repeat removal (Y/N)	**-0.41 (0.20)***	91
	Removal method (whole plot / focal tree)	-0.88 (0.57)	21
	Logged forest	-0.49 (0.58)	17
	Dry season length	0.30 (0.17)	17
	Annual precipitation	0.00 (0.00)	16
	Annual temperature	-0.02 (0.19)	19
	Elevation	-0.00 (0.00)	23

Objective 2.2 (AGB accumulation)	a)	***Time elapsed since removal***	**0.01 (0.00)***	54
		Repeat removal (number)	**0.18 (0.05)*****	62
	b)	***Time elapsed since removal***	**0.01 (0.00)****	54
		Repeat removal (Y/N)	0.04 (0.27)	56
	c)	***Time elapsed since removal***	**0.01 (0.00)***	48
		Repeat removal (number)[a]	**0.17 (0.05)*****	51
		Time since disturbance	**1.16 (0.40)***	13

**p<0.05, **p<0.01. ***p<0.001*

[a] Excluded from Figure 3 as the same result as model a).

Global carbon sequestration potential

Extrapolating the 209% increase in AGB accumulation resulting from climber removal to our assumed maximum application scenario (timber production and secondary forest), we find that climber removal could sequester an additional 32 Gigatons of CO_2 over a decade (22.9 in production forest and 9.2 in secondary forest; Table 3). With the mean reported cost of climber removal as US$8.64 ha^{-1} (see additional published data), we calculate the cost of climber removal as US$0.11 and US$0.03 per Mg (metric ton) of CO_2 sequestered over 10 years for selectively logged and secondary forests, respectively (range: US$0.003-US$0.22; Table 3).

Table 3: Global carbon sequestration potential of climber removal. Extrapolating the enhancement of AGB accumulation through climber removal (intercept of the model for Objective 1.2) to calculate the carbon sequestration that could be provided by climber removal in production and secondary tropical forests. See published additional data for full calculation.

Forest Classification	**$AGBg_0$ ($Mg\ C\ ha^{-1}\ yr^{-1}$) [a]**	**$AGBg_{CR}$ ($Mg\ C\ ha^{-1}\ yr^{-1}$) [b]**	**Area of forest ($ha\ yr^{-1}$) [c]**	**Additional carbon sequestration with climber removal over 10 years ($Mg\ C0_2$) [d]**	**Cost of climber removal per CO_2 sequestered over a decade (US$ $Mg\ C0_2^{-1}$) (min/max) [e]**
Production forests	1.49	4.61	282,879,090	22,862,805,785.40	0.11 (0.01,0.22)
Secondary Forests >3 years since deforestation	4.49	13.87	29,500,000	9,163,581,171.56	0.03 (0.003,0.06)
		Total	***312,379,090.00***	***32,026,386,956.95***	

[a] *$AGBg_0$* is the baseline biomass growth (in metric tons [Mg] of carbon per hectare per year): for production forest this is the mean biomass growth rate from (Butarbutar et al., 2019; Gourlet-Fleury et al., 2013; Rutishauser et al., 2015); for secondary forest this is the mean from Cook-Patton *et al* (2020).

[b] *$AGBg_{CR}$* is the climber removal enhanced biomass growth rate:

$AGBg_0 + (AGBg_0 * 2.09)$ [effect of climber removal on AGB accumulation; intercept of model for objective 1.2].

[c] Forest area classified as production forests with a valid concession license (designated management objective or production forests;(FAO, 2020); Area of moist tropical forest that is classified as regrowing >3 years post deforestation event (Vancutsem et al., 2021).

[d] Difference in baseline and enhanced AGB growth for 10 years in each area of forest, accounting for biomass lost from removing climbers ($AGB_{climber}$) and converted to CO_2 as per IPCC 2003 guidelines: $\left((AGBg_{CR} * Area * 10) - (AGBg_0 * Area * 10) - (AGB_{climber} * Area * 10)\right) * 44/12$.

[e] Cost of climber removal per each additional metric ton of CO_2 sequestered in a decade:

$$Cost\ climber\ removal\ for\ area \Big/ Additional\ CO_2\ sequestered\ over\ 10\ years$$

See additional published data for total cost of climber removal in production and secondary forests.

Discussion

Quantifying the benefits of climber removal for tree growth and AGB is crucial for deciding whether removal should be implemented. We find that climber removal more than doubles tree growth and roughly triples AGB accumulation compared to untreated forests, and that efficacy is sustained for at least 19 years. We also quantify the potential of implementing climber removal for global carbon sequestration and provide recommendations for applying climber removal in certain regions, but note the lack of evidence outside the Neotropics and highlight urgent areas for research.

Climber removal substantially enhances tree growth and AGB accumulation

Our results confirm the findings of individual studies that climber removal has an overall positive effect on tree growth and AGB accumulation (Estrada-Villegas and Schnitzer, 2018), plus emphasise the dramatic role of climbers in tropical forest growth dynamics, carbon sequestration, and forest management. Our approach builds on the largely qualitative Estrada-Villegas *et al* (2018) review by calculating the size of the effect of climber removal and uncertainty in efficacy, whilst methodically accounting for study context. We also estimate the potential contribution of climber removal to global carbon sequestration: sequestering 32 Gigatons of CO_2 in a decade at relatively low cost if applied to secondary forests and production forests across the tropics.

The global carbon sequestration potential is not surprising given the unrealized carbon potential of degraded tropical forests (350 Gigatons CO_2) identified by Erb *et al*. (2018). However, our extrapolation may be influenced by (i) selection bias for studies occurring in locations with high climber density, (ii) our inclusion of a few studies that only measure the efficacy of removal on trees directly infested with climbing plants (rather than all trees in a given plot), and (iii) extrapolating to the total area of secondary forests >3 years old while our analysis only included studies conducted in secondary forests 20-60 years old. On the other hand, climbers do influence entire plots, not just the tree they infest (van der Heijden et al., 2015), and climber infestation in degraded forests tends to be high (Schnitzer and Bongers, 2002); up to 80% trees infested in selectively logged forest in Malaysian Borneo [unpublished data]). Moreover, secondary forests only contribute a third of the total calculated sequestration potential of climber removal, and we do not account for the reduced tree mortality and enhanced seedling recruitment associated with climber removal (O'Brien et al., 2019; Philipson et al., 2020; van der Heijden et al., 2015). For these reasons we anticipate that any over-estimate of the climate mitigation potential of climber removal is limited. Nevertheless, more research and more detailed data, such as climber abundance and individual tree-level data, are needed to further refine our global estimates of the stand-level impact of climber removal on tree carbon sequestration rates.

This study demonstrates how to extrapolate our results to the extent of tropical forest in two scenarios. Our estimate assumes that the maximum extent where climber removal is appropriate is 312.4 million ha (tropical timber production natural forest and secondary forest). While it will not be feasible in every hectare in these landscapes, and many logging concessions are not yet logged nor will see the benefit of climber removal for some time, we consider this a conservative estimate. We restrict our tropical timber production forest to areas under valid timber concession licences (282.9 million ha), while noting there is a larger area reported as production forest (~400 million ha according to FAO (2020)). Further, (Potapov et al., 2017) estimate ~1.4 billion ha is non-intact tropical forest, indicating considerably larger maximum extent for implementing climber removal. Using our study as an example, extrapolations could be made for alternative forest extents, at scales relevant to individual countries or landowners, and regarding timber rather than carbon stocks.

Influence of region and climate remains unclear

Though our results give no indication that the efficacy of climber removal on tree growth and biomass accumulation is influenced by elevation, latitude, presence and length of dry season, precipitation, and temperature, the poor distribution of study sites means there is insufficient evidence to conclude that region and climate have no effect. There are very few studies outside the Neotropics, none in the montane tropics and forests with the highest annual rainfall (e.g., the Chocó, Colombia), and very few studies considered the influence of drought, despite their increasing frequency and concerns that climber removal may be detrimental in drought conditions (Berenguer et al., 2021; IPCC, 2021; O'Brien et al., 2019). The scarcity of climber removal studies outside the Neotropics represents a major gap in our knowledge: particularly troubling as climber removal is increasingly prescribed as a restoration intervention (Cerullo and Edwards, 2019; Philipson et al., 2020).

Climber removal, nonetheless, remains an important potential restoration action, especially in Africa and Asia where forest disturbance is widespread and climber abundance is high (DeWalt et al., 2015; Hansen et al., 2013). Removal studies in these regions and across wider climatic gradients are urgently required so that evidence-based climber removal can be rolled out pan-tropically. Beyond the tropics, and outside the scope of this meta-analysis, climber removal could also be important in temperate regions, where competing vegetation and climber abundance can hinder growth and biomass accumulation (De Lombaerde et al., 2021; Smith, 1984).

Efficacy of climber removal is similar across disturbance history and methods of removal

Overall, we find limited evidence that the efficacy of removal is influenced by forest disturbance context or method of removal. Climber removal enhances tree growth to a similar extent in selectively logged and secondary forests disturbed up to 60 years previously. This confirms that climber competitive advantage is similar in both selectively logged and secondary forests, and sustained long after disturbance (Schnitzer and Bongers, 2002). Furthermore, our results suggest that sufficient climbers are removed to enhance tree growth with a single intervention and when limited to focal trees. The number of removal interventions and intensity of removal (focal tree or whole plot removal) are key considerations when applying climber removal (Gerwing, 2001; Grogan and Landis, 2009; Mills et al., 2019).

While our biomass analysis found that AGB accumulation was more enhanced by climber removal in forests disturbed longer ago and when removal is repeated, the strength of our conclusions is limited by the number of studies (N=12). However, given that the abundance of larger trees increases with age of forest, and that only trees > 5cm dbh were included in the biomass analysis, this result could indicate that larger trees benefit more from climber removal, potentially due to higher climber loads (Estrada-Villegas et al., 2020). Moreover, the four studies with higher biomass effect sizes in Fig 2 all experienced disturbance at least 55 years ago, or were undisturbed, highlighting the need to corroborate the influence of time since disturbance on removal efficacy.

Recommendations for application and conclusions

We identify two key climber removal scenarios for timber and carbon benefits in the Neotropics. Firstly, in timber production forests, forestry personnel could apply removal to just focal trees, during pre-harvest inventory and timber cruising for greatest efficiency. This is especially significant considering the huge area of production forests (FAO, 2020). Secondly, a single application of ‘whole-plot’ climber removal could be conducted by unskilled labour in degraded forests (regrowing or already selectively logged). Edges of forests could be specifically targeted as they have low value and are easy to access (Ordway and Asner, 2020; Poor et al., 2019), though the important role of climbing plants in edge forests should not be jeopardised (Magnago et al., 2017). Moreover, prioritising removal in older regrowth forests would yield the highest AGB accumulation rates as regrowing forests have higher baseline sequestration rates than selectively logged forests (Butarbutar et al., 2019; Cook-Patton et al., 2020; Gourlet-Fleury et al., 2013; Rutishauser et al., 2015).

The expected gains in growth rates in these scenarios will ultimately contribute to climate mitigation, enhance sustainable timber yields, potentially limit the expansion of timber harvesting into primary forest (Burivalova et al., 2020), and enhance the economic value and function of degraded forests that may prevent their conversion (Cerullo and Edwards, 2019). However, while preventing degraded forests from conversion could protect biodiversity, this study only considers the impact of climber removal on tree and AGB growth. Climbing plants have various functions in tropical forests and their removal can have negative consequences for biodiversity by reducing the species richness of climbing plants, removing food and locomotion resources, and influencing the microclimate (Addo-Fordjour et al., 2020; Arroyo-Rodriguez et al., 2015; Campbell et al., 2015; Cosset and Edwards, 2017; Magnago et al., 2017; Putz et al., 2001; Schnitzer et al., 2020), though see (Cerullo et al., 2019). Our study finds that applying removal just to focal trees and not repeating treatment yield growth benefits while giving climbers greater chance to recover, but this will not be enough to prevent biodiversity losses from climber removal. Additional best-practice guidelines, such as leaving areas of forest untreated and avoiding certain climber species, are critical to safeguard the functional role of climbing plants and minimise negative impacts on biodiversity.

While it may not be feasible, nor advisable, to apply climber removal across the entire tropics, this action clearly presents a major climate mitigation opportunity: one that has not been accounted for in prior estimates of natural climate solutions (Griscom et al., 2020, 2017; Roe et al., 2021). We recommend that climber removal is implemented to some extent as part of restoration and carbon sequestration programmes in the Neotropics, specifically as part of forest management in logging concessions, pre- and post-harvest, and in already degraded forests. However, further studies are urgently required to confirm treatment efficacy in Africa and Asia, and to minimise negative biodiversity implications of climber removal. With climber removal, we have the potential to greatly improve the value of degraded tropical forests, and the future of global biodiversity and carbon

CHAPTER 3: Monitoring lianas from space: Using Sentinel-2 imagery to detect liana removal in logged tropical forests

Abstract

Liana removal – the cutting of over-abundant woody climbing plants (lianas) and bamboos – has the potential to substantially increase tree growth and biomass accumulation across millions of hectares of degraded tropical forest. Satellite imagery could provide data capable of detecting the effect of liana removal on the forest canopy, enabling the large-scale monitoring and validation of liana removal, which remains a key hurdle to its widespread implementation. Using a 320-ha liana removal experiment in Sabah, Malaysian Borneo, we tested whether a time series of Sentinel-2 imagery could detect the canopy signature of liana removal. Calculating a range of metrics derived from the Normalized Burn Ratio – a vegetation index based on spectral reflectance that differentiates leaf from non-leaf – we quantified satellite derived canopy disturbance and fragmentation across a range of liana removal intensities and examined how canopy effects changed in the 12-months following removal treatments. We find that liana removal significantly increases canopy disturbance and fragmentation one month after removal, with partial removal having a smaller effect than complete removal. The impact of liana removal on the canopy declined over time, with measures of canopy disturbance and fragmentation largely indistinguishable from control plots within 12-months of treatment. Our findings provide the first evidence that freely available satellite imagery can effectively detect and monitor large-scale liana removal at a range of intensities. Additionally, we find evidence that partial liana removal could be used to significantly reduce initial canopy disturbance during forest restoration programs.

Introduction

Logging has a profound impact on tropical forests globally. Over 400 million ha of the world's forest are currently designated as timber production forests (FAO, 2020) and global timber demand is only increasing (Malhi et al., 2014). While logging threatens biodiversity (Gibson et al., 2011), alters forest structure (Gatti et al., 2014), and reduces carbon stocks (Pan et al., 2011), logged forests are still instrumental in biodiversity conservation (Edwards et al., 2011; Fisher et al., 2011b; Gilroy et al., 2014), carbon sequestration (Erb et al., 2018; Putz et al., 2012b), and for local economies (Edwards et al., 2021). Protection of logged forests from conversion to non-forest uses is therefore a global priority (Edwards et al., 2014, 2011).

One option to protect logged forests from conversion is to enhance forest function and value (Cerullo and Edwards, 2019). This can include restoring tree composition, timber volumes, or carbon stocks in logged forest towards that of primary forests (Putz et al., 2023; Toledo-Aceves et al., 2021). Such restoration methods include enrichment planting, which aims to replenish tree seedling stocks, and interventions that enable the passive recovery of forests (Cerullo and Edwards, 2019). Large-scale implementation of planting initiatives, however, can be costly, requiring significant increases in global carbon payments to off-set such initial costs (Philipson et al., 2020), and the success of passive restoration depends on particular environmental conditions and protection of recovering forest from human activities (Zahawi et al., 2014).

An alternative solution is the removal of woody, climbing plants (called lianas) that proliferate in logged forests and limit their recovery. Lianas compete intensely with trees and are sometimes referred to as "structural parasites", climbing the stems of trees to reach the canopy rather than investing in their own supportive trunk. Liana removal, therefore, accelerates forest recovery (César et al., 2016; Marshall et al., 2016) by substantially enhancing tree growth, carbon stocks (Estrada-Villegas et al., 2022; Finlayson et al., 2022), and other tree-based metrics including tree reproduction and survival (Estrada-Villegas and Schnitzer, 2018). Restoration in this study is therefore focussed on restoring tree growth and carbon stocks. Liana removal also has substantial potential as a natural climate solution, with one study finding that liana removal could sequester up to 7.4 Gt CO_2 per decade across the tropics at comparatively low cost (Finlayson et al., 2022). However, this is an emerging technique and there are several barriers to its widespread implementation.

Liana removal is already applied over large swathes of logged forest in Malaysian Borneo (Sabah Forestry Department, 2020) and is poised to be rolled out across millions more hectares globally (Finlayson et al., 2022; Putz et al., 2023). Verifying removal extent and monitoring forest responses to liana removal are vital for land managers to accurately track treatment application and efficacy, quantify carbon or tree growth responses, and secure carbon credits and payments from initiatives such as REDD+ and Verra (GOFC-GOLD, 2016). However, monitoring such responses over large and often remote areas of forest using traditional field-based methods requires a lot of labour hours and is logistically problematic (Camarretta et al., 2020; Murcia et al., 2016). Remotely sensed data, which can now be accessed freely at high spatial and temporal resolutions, could be the solution to large scale restoration monitoring and may be particularly relevant to liana management (de Almeida et al., 2020; van der Heijden et al., 2022).

Previous studies have already demonstrated the ability of remote sensing products to differentiate between tree crowns and over-topping lianas based on distinct spectral reflectance (Chandler et al., 2021b; Meunier et al., 2021c; van der Heijden et al., 2022), and to detect decreases in canopy vegetation one year after combinations of enrichment planting and liana removal (Wu et al., 2020). These studies evidence the utility of remote sensing imagery to observe lianas, but do not determine the satellite signal of purely liana removal, nor the spatial or temporal nuances in this signal that could help develop remote sensing tools to monitor treatment. For example, a time-series of satellite images could detect the initial loss and browning of canopy leaves after liana removal and track the recovery of the canopy (Martínez-Izquierdo et al., 2016; Perez-Salicrup, 2001). Moreover, assessing the spatial pattern of changes in the canopy, which is expected due to the variable abundance of lianas within a forest (Campanello et al., 2007; Campbell et al., 2018), could quantify the extent of canopy disturbance and fragmentation (here defined as the process by which a closed canopy becomes disturbed, resulting in smaller patches of contiguous closed canopy) caused by liana removal. As well as aiding with detection and monitoring, temporal and spatial analyses of remote sensing imagery could reveal information about the role of lianas in canopy structure and dynamics.

Remote sensing data could also help to refine the application of liana removal. Lianas are a key component of tropical forest systems: constituting 20% of the woody plant diversity in tropical forests, providing food and nesting resources, facilitating arboreal animal locomotion, and buffering the understory from extreme temperatures (Arroyo-Rodriguez et al., 2015; Magnago et al., 2017; O'Brien et al., 2019; Putz et al., 2001; Schnitzer and Bongers, 2002). Hence, there are serious concerns about the unintended negative consequences of large-scale liana removal, with many recommending that a proportion of lianas should be retained in a target area (here-in termed "partial removal") (Estrada-Villegas and Schnitzer, 2018; Finlayson et al., 2022). However, the trade-offs of partial removal between carbon and timber recovery and wider biodiversity and ecosystem functioning have yet to be experimentally tested. Satellite data could be used to compare the extent of canopy disturbance and fragmentation after partial and complete removal, evidencing whether partial liana removal minimises damage to the forest, and potentially has fewer negative consequences. A less fragmented canopy after partial removal, for example, could indicate that the movement of arboreal animals will be less restricted by this form of removal treatment.

Different satellite signals for partial compared to complete removal would also suggest that satellite data could detect areas where liana removal has missed some liana individuals. This issue has been identified in commercial liana removal – for example in Belize where Mills *et al* (2019) found that 30% of climbers were missed during commercial liana removal – and could reduce the tree growth and carbon sequestration achieved by liana removal. Consequently, satellite data could be used to identify where liana removal crews need to re-visit, or to adjust the expected outcomes of removal treatment.

Here, we experimentally applied varying intensities of liana removal to 320 ha of logged forest in Malaysian Borneo and used a time series of satellite images to determine whether Sentinel-2 can monitor and detect this emerging restoration activity. Specifically, we test: (1) whether satellite imagery can be used to detect canopy degradation and fragmentation caused by liana removal; (2) whether the effects of liana removal differ between varying intensities of removal; and (3) how long the signal of liana removal remained detectable post-treatment.

Methods

Study area

We set up the liana removal experiment in the Ulu Segama-Malua Forest Reserve (USMFR), within the Yayasan Sabah (YS) logging concession, Sabah, north-eastern Malaysian Borneo (Figure 4). The study site is defined as an aseasonal lowland dipterocarp forest, with mean annual rainfall of 2651 mm year^{-1} and a mean maximum temperature was 29.1°C between 2018-2020 (SEARRP, 2020).

The USMFR forest reserve was selectively logged twice using a modern uniform system, employing tractors and high-lead cable extraction techniques. The area was first logged between 1976 and 1991 when ~120 m^3 ha^{-1} of timber was extracted, and then again between 2001 and 2007 when an additional 15-72 m^3 ha^{-1} of timber was extracted (Edwards et al., 2011). Tree basal area at the site averaged 4.85 m^3 ha^{-1} (±1.56) with tree composition dominated by fast-growing, early successional species. Liana infestation was high, with 82% of adult trees infested with lianas, and lianas covering an average of 50% of infested trees' crowns (Cannon et al., 2023).

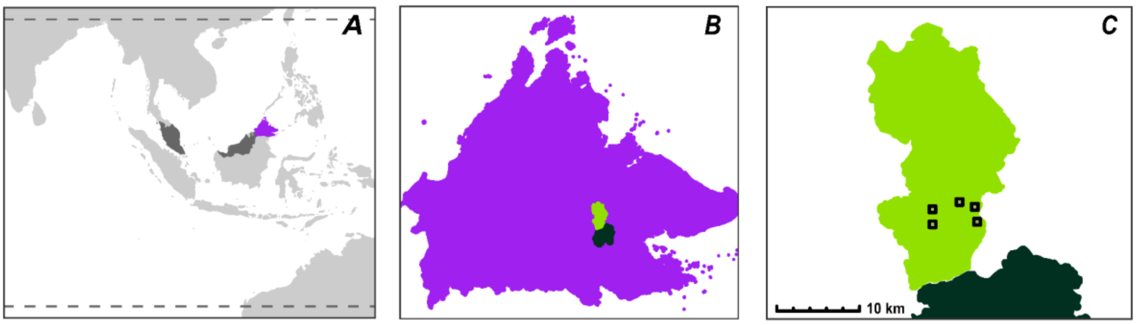

Figure 4: Study site location. Map of SE Asia with Malaysia highlighted in dark grey and Sabah in purple (A), map of Sabah with Danum Valley Conservation Area in dark green and Malua Forest

Reserve in light green (B), locations of the five 800 x 800 m experimental sites within the Malua Forest Reserve (C).

Liana removal experiment and field data

In 2019, we established five independent 800 x 800 m sites at least 1 km apart and 100 m from the nearest logging road (Figure 4C). We divided each site into sixteen 200 x 200 m treatment blocks (80 blocks in total) (Figure 5A). Between September and November 2019, we applied one of three liana removal treatments or a control to each treatment block. The treatments represent varying intensities of liana removal, achieved by leaving different proportions of the block with uncut lianas: 0% area treated (control), 60% area treated (two 40 m strips uncut), 80% area treated (two 20 m strips uncut), and 100% area treated (complete removal across whole block). We kept the number of uncut strips consistent between intermediate treatments, thus limiting difference in the amount of uncut edge between blocks. Cutting in strips aligned with the methods used by commercial liana removal teams in the region. We arranged treatments in a 4 x 4 Latin square design with all four treatments represented in each row and column (Figure 5A), totalling 20 replicates of each treatment across the five sites.

A team of local contractors with experience of liana removal and forest management within USMFR carried out the liana removal treatments. Climbing plant stems (including lianas, climbing bamboo, and rattan) were cut near to the floor and at shoulder height using machetes to prevent stems from re-connecting (Putz et al., 2023). Cut climbers were not physically removed and were allowed to decompose in situ to avoid damaging tree crowns.

To account for variation in initial liana abundance, we recorded pre-treatment canopy liana load in two to five 20 x 20 m subplots randomly located in the central 100 m^2 of each treatment block (Cannon et al., 2023). Canopy liana load estimates the proportion of liana coverage in each adult tree crown (Muller-Landau and Visser, 2019), following a five-point ordinal scale (0 = no lianas in the canopy, 1 = 1-25% coverage, 2 = 26-50% coverage, 3 = 51-75% coverage, 4 = 76-100% coverage). Canopy liana load was averaged for all trees within each subplot, and then averaged across all subplots within each treatment block. Rainfall data were collected at the Malua Forest Research site, twice daily where possible (accessible at: http://www.searrp.org/scientists/available-data/).

Remote sensing data

In order to detect potentially fine-scale and temporally dynamic changes in canopy structure following liana removal, we used high spatial (10 x 10 m) and temporal (every 5 days) resolution imagery from the Sentinel-2 (S2) MultiSpectral Instrument (Level 2A data). Imagery is orthorectified and atmospherically corrected to surface reflectance. This instrument acquires reflectance data in 12 spectral bands, ranging from aerosols (443.9 nm) to short-wave infrared (2202.4 nm).

We used all S2 images acquired across our experimental sites from December 2018 (the first surface reflectance corrected images available over the study region) to November 2020, totalling 78 images spanning nearly one year before and one year after treatment (Table S14). As Borneo is among the cloudiest places on Earth (Wilson and Jetz, 2016) and to minimise noise from atmospheric effects obscuring subtle canopy disturbances, clouds, cloud shadows, and non-forest artefacts were removed from all images using the in-built S2 cloud mask, which determines presence of clouds based on several bands (European Space Agency, 2023), and fine-tuned thresholds in the aerosol, blue, red, and green bands.

Quantifying canopy disturbance and fragmentation

To quantify canopy disturbance and fragmentation resulting from liana removal, we derived the Normalized Burn Ratio (NBR) from the Sentinel-2 images in Google Earth Engine. We used NBR as it detects a loss of photosynthetically active leaves, directly quantifies canopy openness and disturbance, and has recently been used to detect small-scale canopy disturbance (Langner et al., 2018). Initial data exploration also demonstrated that liana removal treatment blocks were more clearly distinguishable using NBR than Normalized Difference Vegetation Index (NDVI), Enhanced Vegetation Index (EVI), and Greenness Index (GI) (see Supplementary Information section *'Other satellite imagery and metrics*'). The equation for NBR is as follows:

NBR = (N-SWIR2)/(N+SWIR2) (1)

Letters indicate spectral reflectance bands: N = near-infrared (835.1 nm); SWIR2 = short-wave infrared 2 (2202.4 nm).

We calculated NBR for each pixel in each S2 image one year before and one year post-treatment and summarised the NBR values in each treatment block using four metrics indicating the level of canopy disturbance and fragmentation. We excluded pixels within 5 m of the edge of each block to account for GPS error and excluded data when more than 15% of pixels in a treatment block were masked due to clouds or other artefacts. We calculate two metrics of canopy disturbance and fragmentation for each treatment block:

1. Median NBR: lower NBR suggests fewer photosynthetically active leaves in the canopy, more bare earth, or greater canopy openness.
2. Proportion of canopy disturbed: we quantified the proportion of S2 pixels in each treatment block that had > 5% reduction in NBR compared to the median NBR value for each pixel during the year pre-treatment.
3. Mean area of intact canopy patches: we classed pixels as ‘intact’ when they had < 5% reduction in NBR compared to the median value for the pixel for a year pre-treatment. We then calculated the mean area of ‘intact patches’ in each treatment block, defined as an area of adjoining intact pixels (the minimum patch size is one pixel).
4. Aggregation of intact canopy patches: we quantified how aggregated (or clumped together) intact canopy patches were, using the definition of an intact canopy patch from metric 3.

Metrics 2-4 were devised following landscape ecology theory (Hesselbarth et al., 2019; Senior et al., 2019) and calculated using *landscapemetrics* and *landscapetools* packages in R (Hesselbarth et al., 2019; Sciaini et al., 2018). The 5% change in NBR was an arbitrary threshold that aimed to differentiate between changes in NBR caused by liana removal and naturally expected variation in NBR. Since this paper aimed to test whether satellite data can detect liana removal, the ecological meaning of the 5% change is less important. To test whether metrics 2-4 were impacted by different NBR reduction thresholds, we also calculated these metrics with a 10% change in NBR threshold. To verify conclusions about the influence of liana removal on canopy disturbance based on NBR we also calculated metrics 1 and 2 using NDVI, EVI, and GI (Zeng et al., 2022). See Supplementary Information section *Other satellite imagery and metrics* for details of other satellite-derived metrics that we explored but do not present in this manuscript.

Statistical tests

To test whether liana removal caused canopy disturbance that could be detected by satellite (objective 1), we initially visualised the pixel-level NBR and percentage change in NBR compared to the year pre-treatment for one S2 scene in which one experimental site had no cloud-masked pixels. To confirm any visual signal of liana removal – suggesting a significant effect on canopy degradation and fragmentation – we analysed the difference between the median NBR in treated blocks compared to control and pre-treatment, and the difference between the proportion of pixels with decreased NBR, mean area of intact patches, and aggregation of intact patches in treated compared to control blocks. We ran statistical analyses for all images within one-month, when we expect the impact of liana removal to be largest (O'Brien et al., 2019), and all images within 12-months post-treatment to determine if there was a significant satellite signal across both time series.

The one-month time-series was analysed using linear fixed effects models, with canopy disturbance and fragmentation metrics for each treatment block as the response variables, treatment (0, 60, 80, or 100% liana removal) and experimental design (row and column of treatment blocks) as fixed effects, and rainfall and mean liana load for the treatment block as fixed effects when significant. The models were run using the *nlme* package in R (Pinheiro et al., 2018). The 12-month time series was analysed using generalized additive models (GAMs) to account for seasonality and temporal non-independence in the time-series. These models followed the same model structure as the one-month models, included a smoothing term of the image date * site interaction, and were run using the *mgcv* package in R (Wood, 2011). The reference treatment level in all models was 0% removal (control), meaning that a significant positive coefficient for 60, 80, or 100% removal treatment indicated that liana removal significantly increased the disturbance or fragmentation metric compared to control and showing that liana removal could be detected using the response variable metric. To determine whether canopy disturbance and fragmentation metrics differed significantly between removal intensities (objective 2), we also calculated the estimated marginal means for all combinations of removal intensities (i.e., 60% vs 100% removal) from the one-month and 12-month models.

To determine whether the canopy degradation and fragmentation caused by liana removal varied across a year post-treatment (objective 3), we compared the coefficients for liana removal treatments between the one-month and 12-month analyses. We also plotted the canopy disturbance and fragmentation metrics in the treated blocks relative to control blocks for each month in the year post-treatment. This showed the change in NBR metrics caused by liana removal relative to control blocks throughout the year. All analyses and figures were produced using R statistical software (R Core Team, 2020).

Results

Canopy disturbance and fragmentation detected by Sentinel-2

We found that liana removal caused canopy disturbances that were clearly detectable using Sentinel-2 (S2) imagery. Liana removal treatment blocks with any level of liana removal (60, 80, or 100%) were visually distinct from surrounding forest and control blocks when using raw NBR (Fig 5B) and the change in NBR compared to pre-treatment (Fig 5C) at one-month post-treatment. Moreover, we found that liana removal caused significant canopy disturbance when examining all NBR metrics from all five experimental sites and all S2 images within a month post-treatment. Specifically, all liana removal intensities (60, 80, and 100%) significantly reduced median NBR compared to control blocks and pre-treatment levels and increased the proportion of the canopy that was disturbed (the proportion of the canopy with decreased NBR) compared to control blocks (Fig 6A; p-values < 0.01; Table S16). On average across 12-months post-treatment there was also a significant increase in canopy disturbance according to these metrics (Fig 6B, p-values < 0.001, Table S16). Liana abundance had no influence on the level of canopy disturbance across either time series (Table S16).

Additional analyses corroborated these results. We found that liana removal affected canopy disturbance across one-month and 12-months post-treatment similarly irrespective of whether the proportion of the canopy with decreased NBR was calculated using a 5% or 10% reduction threshold (Table S17). The canopy disturbance caused by liana removal was also detected when using minimum NBR, median GI, NDVI, and EVI, and the proportion of the canopy with a decrease in these vegetation indices across 12-months post-treatment (p-values < 0.05), with one exception, and across many of these metrics when we only analysed images from one-month post-treatment (Table S18; Fig S19).

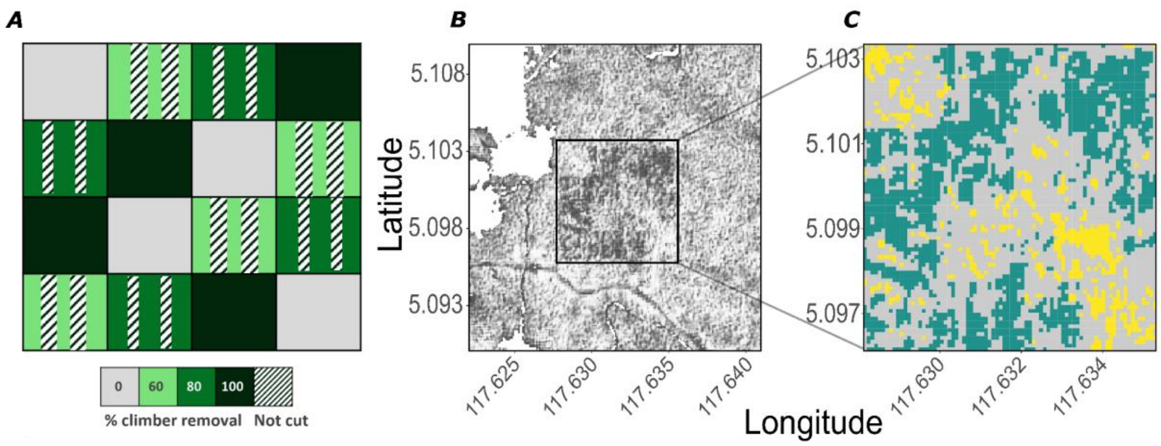

Figure 5: Impact of liana removal on Sentinel 2- derived NBR in one experimental site four weeks post-treatment. Layout of liana cutting treatments for each of the five sites using a Latin-Square design (A), raw NBR values across one experimental site (inside the black square) and the surrounding forest (B), change in pixel-level NBR values compared to median for the year pre-treatment (C). Darker pixels in (B) represent lower NBR values (minimum -1), lighter colour represents higher NBR values (maximum 1), and white patch is a cloud that has been masked. Yellow

pixels in (C) indicate those with > 5% increase in NBR compared to pixel-level median NBR for one year pre-treatment, blue indicates pixels with > 5% decrease in NBR, and grey indicate < 5% change in NBR.

We also detected a significant increase in canopy fragmentation after liana removal treatment. Intact canopy patches were significantly smaller and less aggregated in all liana removal treated blocks than control blocks during the first-month post-treatment, irrespective of removal intensity (Fig 6A, p-values <0.01, Table S16). However, only 100% removal blocks had significantly higher canopy fragmentation than control blocks across 12-months of treatment (p-values < 0.001, Fig 6B, Table S16). Liana removal increased canopy fragmentation similarly irrespective of whether intact canopy was defined as less than 5% or 10% reduction in NBR (p-values < 0.001, Table S17). Again, the impact of liana removal on forest fragmentation was not influenced by initial liana load.

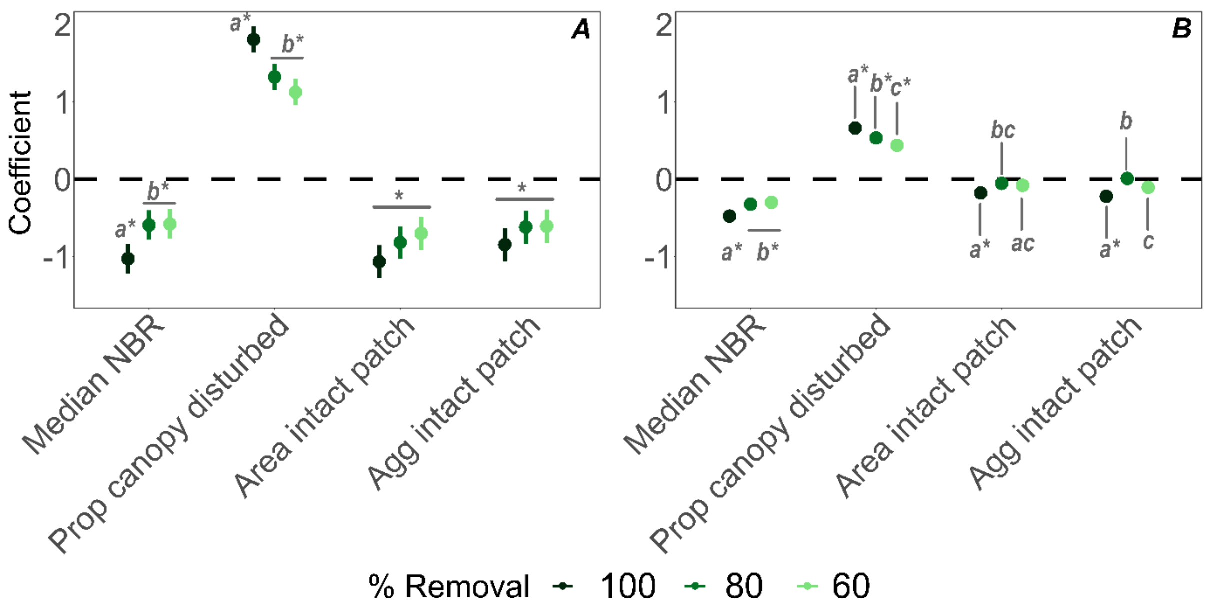

Figure 6: Effects of different intensities of liana removal (60, 80, and 100% removal) on canopy disturbance and fragmentation based on S2 images acquired during one-month (A) and 12-months (B) post-treatment. Points show coefficients of treatment intensities from linear models in

(A), and from GAMs in (B); response variables are normalized before running models. The dotted line shows control (0% removal), coefficients below the line indicate a decrease compared to control, and above the line indicate an increase compared to control. Different grey letters indicate a significant difference between percentage removal treatments, calculated using the estimates marginal means, and "*" indicates removal treatments that are significantly different from control (zero). Error bars show standard error.

Higher intensity liana removal causes greater canopy disturbance and fragmentation

We were able to differentiate between the effects of some intensities of removal (60, 80, and 100%) on the forest canopy using Sentinel-2 imagery. Blocks treated with complete (100%) removal had greater canopy disturbance (lower median NBR and greater proportion of the canopy with decreased NBR) compared to control blocks than partial (60 and 80%) removal treatments during the first month and one-year post-treatment (Fig 6, p-values < 0.05, Table S16). There was also greater canopy fragmentation (smaller and less aggregated intact patches) after complete removal than partial removal, but this was only significant when assessing all S2 images within 12-months of treatment (Fig 6B, p values <0.06, Table S16). Greater canopy disturbance and fragmentation in complete than partial removal blocks was also observed when metrics were calculated using a 10% rather than a 5% reduction in NBR threshold (p-values < 0.05, Table S17).

We were generally unable to differentiate between partial removal treatments (60 and 80% removal) using canopy disturbance and fragmentation metrics. Exceptions to this did not conclusively indicate whether 60 or 80% removal had a greater impact on the canopy. While 80% removal caused a greater proportion of the canopy to be disturbed than 60% across 12-months post-treatment, 80% removal had a smaller effect on the aggregation of intact canopy patches (Fig 6B, p-values <0.05, Table S16) and caused a smaller reduction in EVI compared than 60% (Fig S19B, p-value = 0.001, Table S18). Moreover, partial removal treatments were indistinguishable when disturbance and fragmentation metrics were calculated using 10% rather than 5% reduction in NBR (Table S17).

Overall, these results suggest there is a difference in canopy disturbance and fragmentation when decreasing the proportion of lianas removed from 100 to 80% but no difference between partial liana removal treatments.

Canopy disturbance and fragmentation decline over a year post-liana removal

The impact of liana removal on the canopy reduced over a year post-treatment – there was a greater average effect of liana removal in the one-month time-series compared to 12-months for all metrics (represented by larger coefficients in Fig 6A than 6B). By month 12 post-treatment, minimum NBR, proportion of canopy disturbed, and fragmentation metrics had returned to being similar in treated and control blocks (Figure 7), indicating substantial canopy recovery within a year of liana removal. These results are consistent across additional canopy metrics (GI, NDVI, EVI, minimum NBR indices), and when using a 10% threshold for calculating the proportion of disturbed canopy and fragmentation metrics (Tables S17-S18 & Fig S19). Also, the large drop in median NBR in treated compared to control blocks at month five (Fig 7A) results from a sharp increase in the median NBR in control blocks (Fig S20), likely caused by an artefact in the imagery and fewer images in this month due to cloud cover.

The proportion of the canopy with decreased NBR had the clearest recovery across 12-months (Fig 7B), with all intensities of removal increasing the proportion of disturbed canopy compared to control blocks until 8-months post-treatment. Conversely, in terms of median NBR, treated blocks became indistinguishable from control blocks by month six (Fig 7A) and much sooner in terms of the area and aggregation of intact pixels (Fig 7C & D). Supplementary analyses showed that median and minimum NBR had similar trends over 12-months post-treatment (Fig S21), but the trend is more variable and treated blocks become indistinguishable from control sooner with metrics based on GI, NDVI, and EVI (Fig S22). In general, complete removal had a larger influence on canopy disturbance and fragmentation than partial removal throughout 12-months post-treatment (Fig 7), but all intensities of removal became indistinguishable from control at a similar time for each metric.

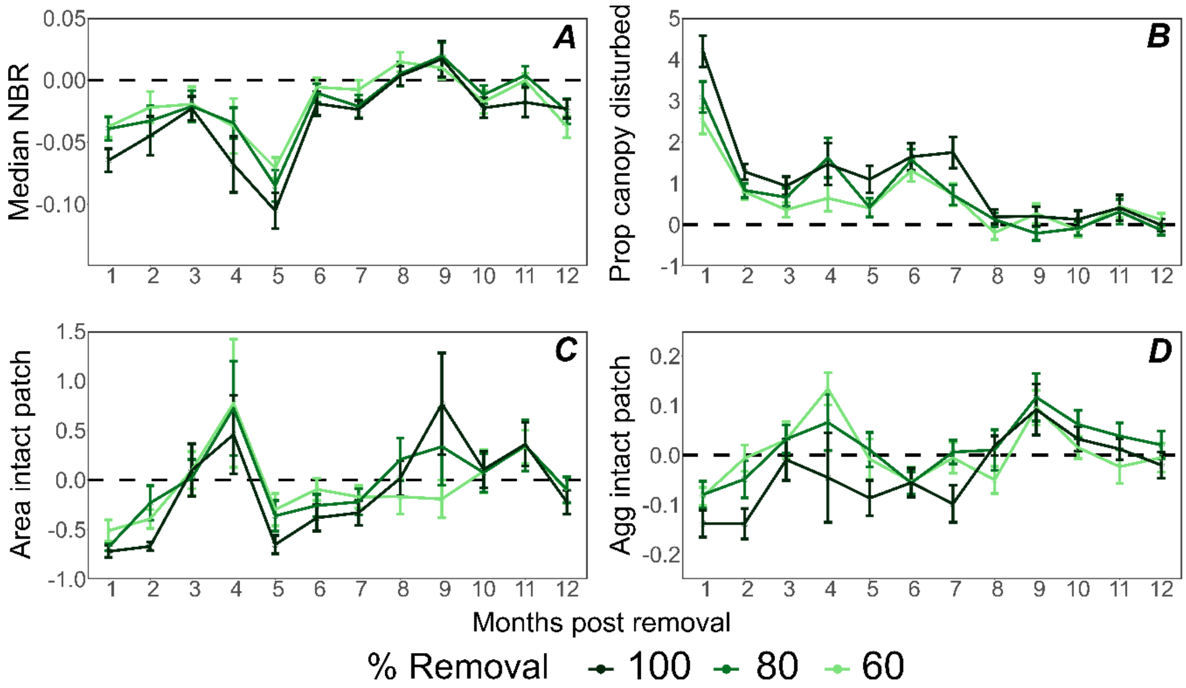

Figure 7: Effect of liana removal on canopy degradation and fragmentation over 12-months post-treatment. Lines in shades of green represent the mean degradation and fragmentation metric value at each month post-treatment for each treatment intensity, relative to the mean control value at each month (dotted black line at zero). Values above the dotted line indicate that in that month, the metric was higher in treated than control blocks, and vice versa. (A) Shows median NBR, (B) shows the level of canopy degradation, defined as the proportion of the canopy with more than a 5% decreased NBR, (C) shows the mean area of undisturbed canopy patches, and (D) shows the aggregation of undisturbed canopy patches. Error bars show standard error.

Discussion

The ability to detect and quantify the effects of liana removal via remote sensing is key to its large-scale monitoring and validation. We show, for the first time, that remote sensing data can detect the impact of varying intensities of liana removal on the canopy over large spatial and temporal scales. Specifically, we find that: liana removal fragments and disturbs the canopy, these impacts are minimised with partial removal, and the canopy largely recovers within a year of treatment. Below, we discuss what our work reveals about liana ecology and tropical forests and outline the implications of our results for enhancing tree growth and biomass accumulation in logging concessions.

Liana removal and canopy dynamics

Studying the satellite signal of liana removal provides insight into the influence of lianas on the canopy and their role in tropical forests. Consistent with other studies, our results indicate that liana removal increases canopy browning or openness (O'Brien et al., 2019; Perez-Salicrup, 2001; Wu et al., 2020). Detecting these canopy changes using 10 m resolution imagery evidences that lianas are a substantial component of the canopy, supporting literature showing that lianas maintain cool, low-light, and low-wind understory conditions (Meunier et al., 2021c). Hence, our results emphasise that liana removal could reduce the survival of shade-tolerant tree species (O'Brien et al., 2019) and subject fauna and flora to more extreme conditions (Scheffers et al., 2014). The fragmentation of the canopy after liana removal also exemplifies that lianas help to connect across the canopy, without which arboreal animals may become isolated and have fewer modes of movement (Benjamin J. Adams et al., 2019; Putz et al., 2001). While further field data is required to directly explore how liana removal influences these factors and the response of faunal taxa, the substantial impact of liana removal on the canopy is likely to have myriad knock-on effects, emphasising that safeguarding the functional role of lianas is critical when implementing liana removal in tropical forests.

Partial removal has been suggested as a safeguard against the potential negative consequences mentioned above (Estrada-Villegas and Schnitzer, 2018; Finlayson et al., 2022), and this study is the first experimental comparison between complete and partial removal. As anticipated, partial removal significantly reduced canopy disturbance, but, interestingly, our results suggest that 60% and 80% liana removal release trees from lianas to a similar extent. Evidently, further research is required to explicitly measure the impact of partial removal on biodiversity, forest function, and tree growth and biomass accumulation, but this is the first evidence that leaving 20-40% of the target area untreated could substantially reduce canopy openness, fragmentation and their harmful consequences. The relative impact of alternative configurations of partial removal on tree and carbon benefits and biodiversity should also be tested. For example, could treating a proportion of future crop trees, as proposed by Putz *et al* 2023, preserve some of the connectivity and resource functions of lianas throughout a treated area?

The recovery of the canopy within 12-months of treatment, consistent with field-based data from the same region (O'Brien et al., 2019), shows that the canopy is highly dynamic, but it is difficult to determine whether this recovery is driven by trees or lianas. While other studies quantify liana abundance using airborne hyper-spectral and trained satellite data (Chandler et al., 2021), Sentinel-2 imagery alone is too coarse resolution so we cannot use this data to determine the relative proportion of trees and lianas in the canopy (van der Heijden et al., 2022). Lianas are known to recover after liana removal treatment (Alvira et al., 2004; Campanello et al., 2012) and are generally thought to have faster growth rates than trees due to lower investment in woody stems (Phillips et al., 2005; Schnitzer et al., 2014), suggesting that canopy recovery could be driven but lianas, but a recent study found that leaf turnover in aseasonal forests is similar between lianas and trees (Medina-Vega et al., 2021). There was a significant positive correlation between pre-treatment liana load and median NBR, but the relationship was relatively weak (Fig S23; R^2 = 5%) so we cannot use NBR values post-treatment to accurately estimate the liana load. Ground data and higher resolution imagery, such as from Unmanned Aerial Vehicles (UAVs) (van der Heijden et al., 2022; Waite et al., 2019), are required to elucidate whether canopy closure in our study is due to tree or liana growth. In either case, the closure of canopy gaps caused by liana removal implies that the microclimate buffer recovers within a year, benefitting shade-tolerant tree species and understory fauna. If further research finds that canopy closure is driven by lianas, this could suggest that the negative impacts of liana removal on food, nesting, and locomotion resources are temporary.

Detecting and monitoring liana removal

This study presents a method for using Sentinel-2 imagery – a freely available remote sensing product – to detect liana removal. We build on work by Wu *et al* (2020), showing that liana removal increased overall canopy disturbance, the proportion of the canopy that is disturbed, and the mean area and aggregation of intact patches. Combining these four metrics may differentiate liana removal from other disturbances that can be detected with Normalized Burn Ratio (Langner et al., 2018), but further work is needed to compare the signal of liana removal to other disturbances. Our study shows that, once operationalised, quantifying NBR within a few months post-treatment could evidence that liana removal activities have taken place, helping land managers to earn payments from schemes such as REDD+ (Sirro et al., 2018). Compared to collecting similar verification evidence from the ground, using S2 imagery will be faster, cheaper, and be able to cover the entire area treated rather than a subsample of the area (Camarretta et al., 2020; Murcia et al., 2016; Zahawi et al., 2015).

Our results also support the use of Sentinel-2 imagery to monitor application efficacy and forest changes following liana removal treatments over large areas. The clear recovery of the canopy within 12-months suggests that NBR metrics could be used to determine when the effect of liana removal for enhancing tree growth or carbon accumulation has diminished, potentially indicating when removal should be repeated, if desired. Moreover, since we were able to differentiate between partial and complete removal, Sentinel-2 imagery could be used to identify areas where lower intensity liana removal has been applied. This could direct further removal applications to solve the issue of incomplete liana removal in large-scale implementation (as seen in Mills et al., 2019) that reduces the tree growth and carbon sequestration enhancement that can be achieved.

Operationalising large-scale monitoring of liana removal

While our study takes the first steps towards using Sentinel-2 imagery for verifying and monitoring large-scale liana removal, further work is necessary to check the generalisability of our results and for the method to be operational. Firstly, the satellite signals of liana removal that we have identified should be tested across larger, non-experimental areas of liana removal, such as logging concessions in Malaysian Borneo (Sabah Forestry Department, 2020). Commercial treatments may use different methods and intensities of removal that could influence the magnitude and spatial arrangement of the satellite signal, such as the tree-centred approach used in Belize that achieved 70% removal (Mills et al., 2019).

Secondly, forest structure may influence the detectability of liana removal. Since liana abundance had limited effect on the NBR metrics in this study, and this study was conducted in the Asian tropics that are thought to have lower liana abundance than the American and African tropics (DeWalt et al., 2015), we anticipate that liana removal will cause some level of canopy changes detectable by satellite in all tropical regions. However, it may still be harder to detect liana removal with Sentinel-2 imagery when there is lower liana abundance, such as areas with less intense or less recent disturbance (Schnitzer et al., 2014; Yorke et al., 2013). Moreover, the canopy fragmentation found in this study may be specific to forests of the Asian tropics that are dominated by Dipterocarps (Brearley et al., 2016). This family of tree species tends to be less infested with lianas than other tree species, potentially giving rise to the patchy influence of liana removal on the canopy (Wright et al., 2015). Expanding our work to other global regions and forest types may identify common liana removal signals or indicate that calibrating the signal is required in each site.

Finally, while the high spatial and temporal resolution and free access to Sentinel 2 are huge benefits of this imagery source, it is worth exploring whether other remote sensing tools and products, such as GEDI (Dubayah et al., 2022), Planet (Roy et al., 2021), or drone imagery (Waite et al., 2019), find alternative signals of liana removal that could be used to detect and monitor the application of this restoration technique. Notably, GEDI is also freely available and has been used to quantify carbon stored in forests (Ngo et al., 2023; Potapov et al., 2021), so it is worth exploring whether GEDI data can calculate the additional carbon storage achieved by liana removal, facilitating access to Verra carbon credits (GOFC-GOLD, 2016) without extensive ground data collection.

Conclusion

Liana removal causes disturbances to the canopy that can be detected using Sentinel-2-derived NBR. Further work is required to determine whether partial liana removal reduces the negative impacts of the technique, but we recommend leaving at least 20% of target forests untreated to safeguard the various roles lianas have for faunal communities and forest function. Once operationalized, satellite-based detection of liana removal could be employed by land managers to validate and monitor the efficacy of liana removal, assisting the widespread application of the technique to restore tree growth and carbon sequestration in logged tropical forests.

CHAPTER 4: Commercial-scale liana removal detected using satellite data

Abstract

There is growing need for logging practices to become more sustainable to reduce the negative impacts on biodiversity, carbon stocks, and local livelihoods. Lianas (woody climbing plants) grow extensively after logging, becoming a barrier to forest recovery and logging sustainability. While removing lianas significantly enhances timber recovery and is a powerful restoration tool, monitoring the effectiveness of liana removal over vast areas is a challenge. Local-scale liana removal can be detected using satellite data, but it is not known whether this approach can be deployed at greater spatial scales. This study aimed to determine whether commercial-scale liana removal – applied across 17,000 ha of selectively logged forest in Malaysian Borneo – could be detected using satellite-derived data. We also aimed to assess the drivers of variation in the satellite signal of commercial liana removal. We analysed two metrics based on Sentinel-2-derived Normalized Burn Ratio (NBR): minimum NBR and the proportion of the canopy with decreased NBR compared to pre-treatment. These were calculated for one year pre- and post-treatment in logging compartments in which liana removal had been applied, and reference compartments. We ran generalized additive models and mixed effects linear models to determine the effect of commercial liana removal on these metrics. In the year post-treatment, commercial liana removal significantly increased minimum NBR. There was also a negative impact of removal on the proportion of the canopy with decreased NBR that became more negative at higher daily rainfall. The signal of liana removal on NBR metrics was also influenced by terrain and distance from roads. Overall, our study shows the potential for using remote sensing to monitor commercial liana removal and variation in removal intensity, reducing the need for costly on-the-ground quality assessments.

Introduction

Logging is one of the biggest causes of tropical forest loss and degradation globally (Hosonuma et al., 2012), with myriad detrimental impacts on biodiversity, carbon, and timber value (Gibson et al., 2011; Pan et al., 2011; Putz et al., 2012b). Although the expanse of logging is concerning, with recent estimates suggesting that 25% of forests across the globe are subjected to selective logging (Putz et al., 2022), forest management has become increasingly sustainable over the past several decades in many regions (Putz et al., 2008). Sustainable forestry practices are essential to managing increasing timber demand (Malhi et al., 2014), have the potential to reduce the damage to forest ecosystems (Putz et al., 2008), and are being adopted to address biodiversity and climate crises (Betts et al., 2021; Griscom et al., 2020). However, for the most part, logging practices still deplete timber yields over time and truly sustainable logging practices are still lacking (Putz et al., 2022).

One way to improve the sustainability of selective logging is to enhance timber recovery. This can be achieved in various ways, from removing trees that are competing with those of commercial value (Peña-Claros et al., 2008a) to re-planting commercially viable trees (Philipson et al., 2020). Timber enhancement also increases carbon storage in previously-logged areas (Erb et al., 2018), which enhances the economic value of the forest while combating global carbon emissions, and reduces the expansion of timber harvesting into undisturbed areas of forest (Cerullo and Edwards, 2019), which ultimately protects pristine forests that are critical for biodiversity (Gibson et al., 2011). However, timber enhancement can be intensive and expensive (Finlayson et al., 2022).

An emerging, relatively inexpensive method to increase timber recovery is the removal of woody climbing plants such as lianas and bamboos (Finlayson et al., 2022). Lianas become problematic after logging as they grow rapidly in the increased light conditions (Schnitzer and Bongers, 2002), and their removal substantially enhances tree growth and nearly doubles aboveground carbon storage (Finlayson et al., 2022), amongst other benefits for trees (Estrada-Villegas and Schnitzer, 2018). Consequently, liana removal (LR) has been advocated to restore timber and carbon stocks in huge expanses of logged tropical forests and is already being implemented and trialled in countries including Malaysia, Belize, and Bolivia (Mills et al., 2019; Peña-Claros et al., 2008b; Reynolds et al., 2011; Sabah Forestry Department, 2020). However, monitoring the implementation of liana removal over vast areas and validating its application to access carbon sequestration payment schemes, such as REDD+, are critical to deriving the greatest benefit from liana removal.

Satellite-derived vegetation indices have been used to detect small-scale liana removal (Finlayson et al., 2022; Wu et al., 2020), suggesting that remote sensing data could be used to validate and monitor the intervention. Using a field experiment, Finlayson and Hethcoat *et al* (2022) found that Normalized Burn Ratio (NBR) can be used to detect changes in canopy greenness and gaps in forest plots that were subject to liana removal. However, LR was applied at a small spatial scale in this experiment (in 200 x 200 m forest blocks) compared to the several thousand hectares over which LR has been applied in commercial logging sites (Sabah Forestry Department, 2020). A key step towards operationalising the use of remote sensing for large-scale monitoring, therefore, is to determine whether commercial-scale liana removal can be similarly detected using satellite-derived indices.

One issue with liana removal that may influence its detectability over large areas is that achieving complete removal is difficult. For example, 30% of climbing plants were missed during commercial LR in Belize (Mills et al., 2019) and 13% were missed during reduced impact logging practices across 1400 ha in Malaysian Borneo (Pinard and Putz, 1997). Incomplete removal of lianas could reduce disturbance to the canopy and thus make this intervention harder to detect by satellite than the careful experimental cutting of lianas across 300 ha in Finlayson et al (2022). Consequently, this could make validating treatment to acquire carbon credits using remote sensing more difficult. Finlayson and Hethcoat *et al* 2022 also found that NBR could differentiate between low intensity and complete LR, suggesting that remote sensing could help to identify areas where a proportion of lianas have been missed, directing where crews need to revisit, or quantifying the completeness of removal to generate better estimates of the resulting timber growth and carbon sequestration enhancement.

Variation in liana removal completeness could be due to crew fatigue. Mills et al (2019) found that most lianas were missed due to distance from tree targeted for liana removal, and difficulty finding all lianas entering a tree’s canopy, but crews missed treating 6% of trees entirely. While there is no direct evidence that crew fatigue reduces liana removal efficacy, liana removal is physically demanding. Hence, we posit that the accuracy of removal crews (and signal of liana removal) may diminish with larger treatment areas or with increased crew exertion due to further distance walked from the access road and steeper terrain. Building on the work by Finlayson and Hethcoat *et al* (2022), in addition to quantifying and detecting incomplete removal, satellite-derived NBR could be used to determine what drives variation in LR completeness, indicated by the strength of satellite signal.

The signal of LR, in terms of changes in NBR, may also differ at the commercial compared to the experimental scale due to the variable distribution of lianas and competition with trees. For example, studies have shown that competition between lianas and trees varies with precipitation, as does the relative growth of the two groups (O’Brien et al., 2019; Schnitzer and van der Heijden, 2019; Venegas-Gonzalez et al., 2020). Consequently, the impact of LR on NBR, or the ‘detectability’ of LR, may be greater with lower precipitation – when lianas would be growing fastest relative to tree growth and potentially be at higher abundance. Moreover, the effect of liana removal on the canopy is likely to vary with pre-treatment liana abundance. While Finlayson and Hethcoat *et al* (2022 did not find a consistent effect of pre-treatment liana abundance on the canopy disturbance caused by liana removal, liana abundance is likely to vary to a greater extent across larger areas, altering the signal of liana removal across space and potentially revealing information about the ecology of lianas. Areas of a forest with higher liana abundance, due to factors such as higher timber extraction (Addo-Fordjour and Rahmad, 2015a; Putz et al., 2019; Schnitzer and Bongers, 2002), greater time since logging (Yorke et al., 2013), or shallower terrain (Addo-Fordjour et al., 2014; Dalling et al., 2012), may see a greater reduction in NBR after liana removal, for example. Yet, variation in the impact and detectability of liana removal in satellite data remains unexplored as the only other remote sensing LR studies applied removal to a small area of forest within a few months of a single year (Finlayson et al., 2022; Wu et al., 2020).

In this study, we extend our previous experimental work to ask whether commercial-scale liana removal can be monitored using satellite-derived data, and to investigate factors that could influence the detectability or efficacy of the treatment. We explore two core objectives: 1) determine whether commercial-scale LR can be detected using satellite-derived NBR; and 2) assess whether the impact of liana removal on NBR metrics varies due to factors relating to crew fatigue or liana abundance (treatment year, precipitation, size of compartment, terrain, or distance from the road).

Methods

Study area and commercial liana removal

Commercial liana removal was applied in three forest reserves in Sabah, Malaysian Borneo (Fig 8) between 2007 and 2021: Ulu-Segama (USFR), Malua (MFR), and Bukit Piton (BPFR) Forest Reserves (Fig 8B and C). The reserves are dipterocarp forests that tend to have higher rainfall from December to March (Ancrenaz et al., 2010). From 2017 to 2021, the mean annual rainfall was 2908 mm year^{-1} and the mean maximum temperature was 31.6°C (SEARRP, 2022). Annual rainfall was lowest in 2020 (2373 mm year^{-1}, respectively), with a peak in annual rainfall in 2017 (2992 mm year^{-1}). There was a very strong El Nino in 2015-2016 and a weak El Nino in 2018-2019. All reserves were selectively logged twice between 1960 and 2013, using a variety of harvest methods including conventional logging, logging using a log-fisher or crawler tractor, and heli-logging (Sabah Forestry Department, 2020). The volume of extracted timber was 117 m^3 ha^{-1} in the first round of logging and 34.70 m^3 ha^{-1} in the second round of logging (Fisher et al., 2011a).

We analysed satellite imagery across 34 compartments, ranging between 200 ha and 840 ha in size, that had greater than 90% of their area treated with liana removal within 12 months between 2017 and 2021 (Fig 8D). In 2017, 13 compartments were treated in USFR; in 2019, three compartments were treated in MFR and six in BPFR; in 2020, nine compartments were treated in MFR; and in 2021, three compartments were treated in MFR. A team of contractors cut liana stems in each compartment using machetes and allowed them to decompose in situ. Untreated compartments from these three reserves were used as reference forests (Fig 8D).

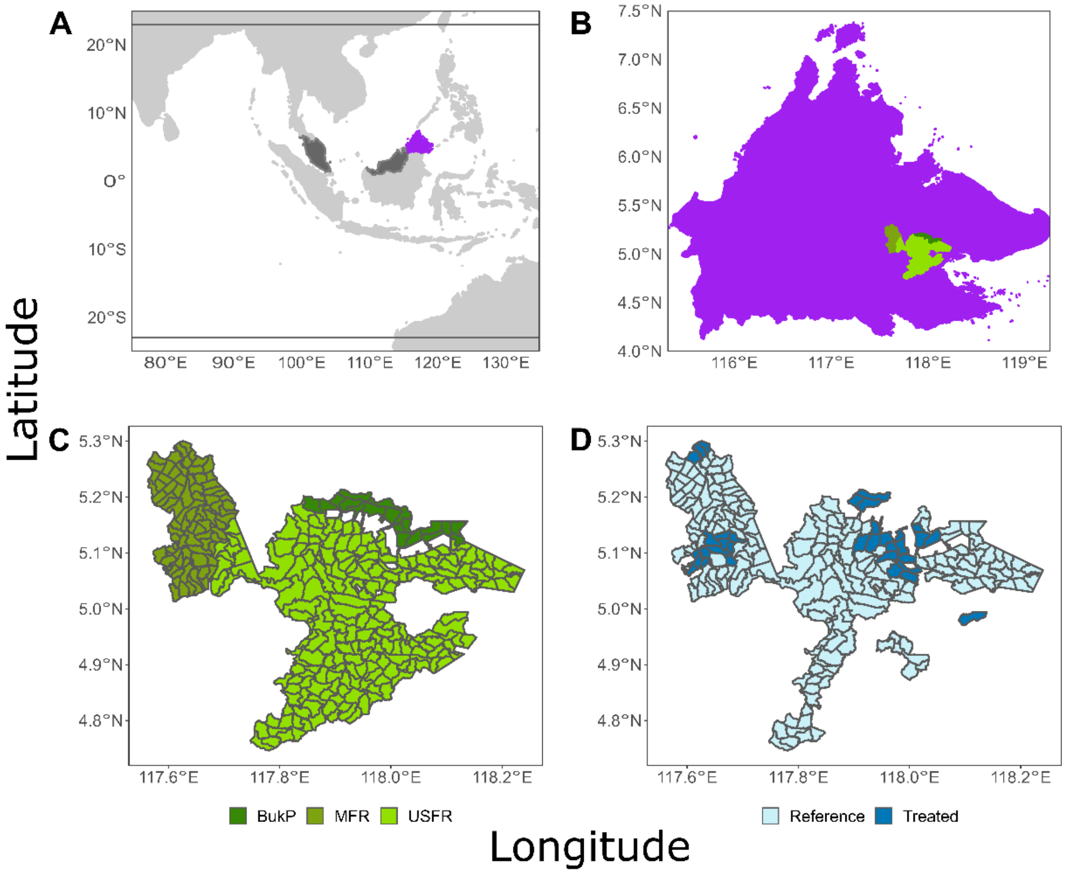

Figure 8: Map of South East Asia showing Malaysia in dark grey and Sabah in purple (A); the location of the three forest reserves in Sabah (B); the three forest reserves in which liana removal was applied between 2017 and 2021: BukP = Bukit Piton Forest Reserve, MFR = Malua Forest Reserve, USFR = Ulu Segama Forest Reserve (C); the logging compartments in which more than 90% of the area was treated with liana removal within 12 months, and compartments used as reference (D).

Remote-sensing data and NBR metrics

We used Level 2A data from the Sentinel-2 (S2) MultiSpectral Instrument, which is orthorectified and atmospherically corrected to surface reflectance (S2-SR), to detect commercial liana removal in compartments treated between 2020 and 2021, following Finlayson and Hethcoat *et al* (2022). Level 2A data (S2-SR) was only available from December 2018 for our study region, so, to detect commercial liana removal treatments conducted between 2017 and 2019, we used Level 1C data from S2, which records top-of-atmosphere reflectance (S2-TOA). We used the in-built S2 quality band and fine-tuned thresholds of aerosol, blue, red, and green bands to remove cloud-affected and non-forest pixels, as per Finlayson and Hethcoat *et al* (2022).

We calculated the Normalized Burn Ratio (NBR) vegetation index (see Equation 1) for each S2 image one-year pre and one-year post-liana removal (from the first day of the first annual quarter in which liana removal was applied to the last day of the last annual quarter in which removal was applied). We used NBR because it is calculated from wavelengths that relate to leaf pigments and can differentiate between leaf and non-leaf material (such as bare ground or wood) (Langner et al., 2018). Finlayson and Hethcoat *et al* (2022) also found that NBR showed the strongest response to experimental liana removal compared to a suite of other vegetation indices.

$$NBR = \frac{N-SWIR}{N+SWIR} \qquad (1)$$

Letters indicate spectral reflectance bands: N = near-infrared (835.1 nm); SWIR2 = short-wave infrared 2 (2202.4 nm).

From the raw NBR values, we derived two summary metrics for each treated and reference compartment at each S2 image date, following Finlayson and Hethcoat *et al* (2022):

1. Minimum NBR: We extracted the minimum NBR value across each compartment, indicating the overall change in canopy greenness.
2. Proportion of the canopy with decreased NBR: We calculated the percentage difference between the median NBR for each pixel one-year pre-treatment and the NBR value for each pixel in each S2 image post-treatment. We summarised this as the proportion of the pixels in each compartment

that had greater than 5% decrease in NBR compared to pre-treatment, hereafter referred to as the proportion of the canopy with decreased NBR. This metric showed the proportion of the canopy in which greenness had decreased in treated and reference compartments.

Statistical analyses

To answer our first objective, asking whether commercial-scale LR can be detected by assessing changes in minimum NBR or the proportion of the canopy with decreased NBR, we used the two NBR metrics as response variables in generalized additive models (GAMs). GAMs were used to account for seasonality and temporal non-independence in the time-series data (Simpson, 2018). These models included LR treatment (treated or reference) as a fixed effect to determine if LR could be differentiated from reference compartments. Coefficients for LR treatment significantly above zero indicated that commercial LR significantly increased the minimum NBR value or the proportion of the canopy with a decrease in NBR compared to reference logging compartments, and vice versa. We ran additional models to account for error in the treatment dates as we only knew the annual quarter in which liana removal treatment started and ended: firstly, we ran the above models with compartments in which liana removal was completed in the fewest months (within 3 months in 2017, 2019, and 2020, and within 6 months for 2021), and, secondly, excluded S2 images acquired during treatment applications.

To reduce the noise in the data that could be masking the effect of LR on NBR metrics, this model also included compartment size, daily rainfall, and treatment year as fixed effects, compartment ID and forest reserve as random effects, and month as a smooth term with 12 knots. We also included the interaction between LR treatment and compartment size, rainfall, treatment year, and forest reserve, where possible, to answer part of our second objective: examining the causes of variation in liana removal signal at the compartment level, potentially due to crew fatigue or liana abundance. A significant compartment ID term indicated that there was significant variation in the NBR metrics between compartments, and significant interactions between the LR treatment term and compartment size, rainfall, treatment year, or forest reserve indicated that these variables influenced the signal of liana removal on NBR metrics.

To further assess the causes of variation in the signal of liana removal, this time within compartments, we split treated and reference compartments into 200 m x 200 m sub-compartments. We calculated the minimum NBR and proportion of pixels with a decrease in NBR for each sub-compartment in each S2 image. Sub-compartment models included NBR metrics as the response variables, LR treatment, month, daily rainfall, forest reserve, and treatment year as fixed effects, and compartment ID as a random effect. To test whether NBR metrics varied substantially *within* compartments we compared the conditional R^2 values between models with and without sub-compartment ID as a random effect. A higher R^2 in the model with sub-compartment ID indicated that there was variation in the NBR metrics within compartments. We used mixed effects models instead of GAMs due to the size of the sub-compartment dataset and computational limitations.

Lastly, to assess whether terrain steepness or distance from road (factors that could influence crew fatigue and completeness of removal or liana abundance) influenced the signal of LR on NBR metrics, we ran the above sub-compartment models with the interaction between liana removal treatment and maximum terrain steepness and minimum distance from main roads or roads used for some activities (hereafter called “secondary roads”). Terrain steepness was obtained for each sub-compartment using the 90 m resolution digital elevation dataset from the Centre for Tropical Agriculture (CIAT) (Jarvis et al., 2008), and the minimum distance of each sub-compartment from roads was calculate using a road network provided by the Sabah Forestry Department. Significant coefficients for these interaction terms would suggest that the impact of liana removal on minimum NBR or the proportion of the canopy with decreased NBR were influenced by terrain and distance from roads, potentially indicating variation in completeness of liana removal or liana abundance.

We used a subset of 51 reference logging compartments (representing all three forest reserves) for the sub-compartment analyses due to excessive time to extract the minimum NBR per sub-compartment in Google Earth Engine. All analyses based on the proportion of the canopy with decreased NBR excluded S2 data when more than 15% of the compartment was affected by cloud cover. GAMs were run using the *mgcv* R package (Wood, 2011) and linear mixed effects models were run using the *lme4* R package (Bates et al., 2015). All analyses and figures were produced using R statistical software (R Core Team, 2020).

Results

Effect of commercial liana removal on NBR

Commercial liana removal caused changes to the forest canopy that were visible with satellite-derived NBR. There was a significant increase in minimum NBR in treated compared to untreated compartments across a year post-treatment (coefficient = 0.08, $p < 0.01$, Figure 9A, Table S19), meaning that liana removal increased canopy greenness. Supplementary analyses also found a significant positive effect of commercial liana removal on minimum NBR when the analysis was constrained to compartments treated within a shorter time-frame, but the positive effect was no longer significant when the analyses only included Sentinel-2 images that were captured after the annual quarter in which treatment was completed (Table S19).

The signal of commercial liana removal on the proportion of the canopy with decreased NBR was less clear. Commercial liana removal reduced this NBR metric compared to reference compartments within a year of treatment (9B), but this was non-significant potentially due to the interaction of treatment with daily rainfall (coefficient = -0.04, $p < 0.001$, Figs 9C & 10B, Table S19). Below 5 mm of rain per day similar proportions of the canopy had decreased NBR in reference and treated compartments, but as rainfall increased, treated compartments had an increasingly smaller proportion of the canopy with decreased NBR compared to reference compartments (Fig 9C). These results suggest that commercial liana removal causes a slight reduction in the proportion of the canopy which is open or dominated by non-photosynthetic material. Results were similar in supplementary analyses constrained to compartments with shorter treatment lengths and excluding S2 images during treatment (Table S19).

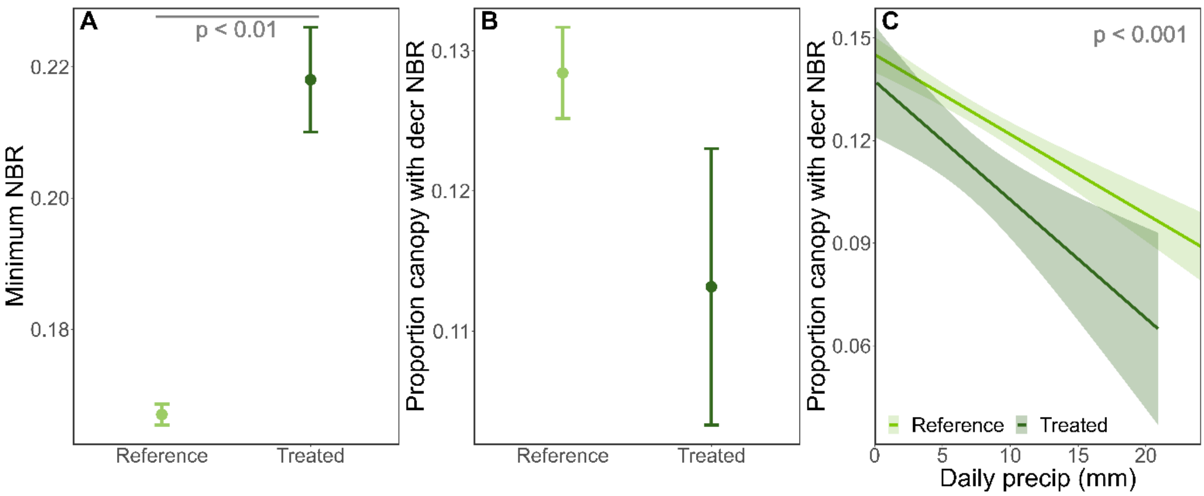

Figure 9: Effect of commercial liana removal on (A) minimum NBR, (B) the proportion of canopy with a decrease in NBR compared to pre-treatment, and (C) the interaction between the effect of commercial liana removal on the proportion of the canopy with decreased NBR and daily precipitation. Points in (A) and (B) show average NBR metrics for reference compartments and all compartments pre-liana removal ("Reference"), and average NBR metrics in treated compartments during 12 months post-liana removal ("Treated"). P-value in (A) is the level of statistical difference in minimum NBR between treated and reference, taken from the GAMs, and error bars indicate the 95% confidence interval. Lines in (C) show the relationship between the proportion of the canopy with decreased NBR and daily precipitation, separated by treated and reference compartment. The lighter band around each line shows the SE of the relationship and the p-value in (C) is taken from the GAM, indicating the significant effect that daily rainfall had on impact of treatment on the proportion of the canopy with decreased NBR.

Variation in the signal of liana removal on NBR

There was substantial variation in NBR metrics between and within compartments. This is shown by the significant compartment ID smooth terms in the GAMs (EDF: 123-200, p-values < 0.001, Table S19) and the roughly doubling of conditional R^2 values when including sub-compartment ID in the linear mixed effect models (increasing from 0.256 to 0.419 and from 0.077 to 0.143 in the minimum NBR and proportion of canopy with decreased NBR models, respectively).

We also found several variables that caused significant variation in the impact of commercial liana removal on NBR metrics (i.e., the signal of liana removal), potentially indicating that crew fatigue or liana abundance influenced the outcome of commercial liana removal. There were significant interactions between LR treatment and precipitation, terrain steepness, and distance from main and secondary roads (Fig 9C, 10 & 11, Tables S19 & S20). Distance from main and secondary roads had the largest effect, reducing the proportion of the canopy with decreased NBR in treated compared to reference compartments by 14-18% as distance from road increased (Fig 10B, 11D & F).

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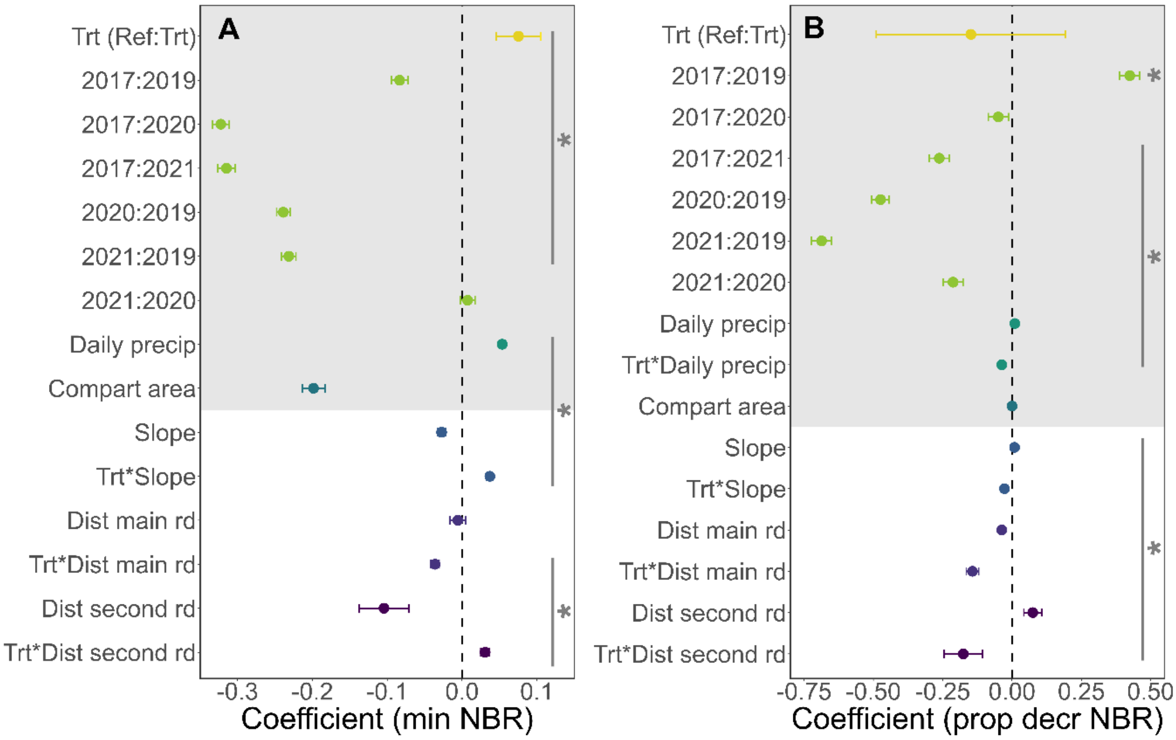


Figure 10: Effect of covariates on minimum NBR (A) and the proportion of the canopy with decreased NBR (B), and the interaction between liana removal treatment and covariates. Points represent the coefficient for different covariates or interactions, taken from whole compartment GAMs for the coefficients in the grey panel and the sub-compartment linear mixed effect models for the coefficients in the white panel. Slope is terrain steepness in degrees. Continuous covariates are scaled in the models and the part of the name before the colon shows the reference level in categorical covariates. Coefficients that interact with treatment are indicated by "Trt*" in the covariate name. Dotted lines at zero indicate where there is no effect of the covariate on the NBR metric and a grey star indicates covariates that significantly increase or decrease the NBR metrics (actual p-values in Table S19 and S20). Error bars indicate standard error.

The significant interactions of covariates with the effect of commercial LR treatment on NBR metrics reveal the conditions in which the signal of liana removal is likely to be largest, potentially indicating where removal crews removed the greatest proportion of lianas, or where liana abundance was highest. For example, the difference in minimum NBR between treated and reference compartments was greatest in steeper sub-compartments and those further from main roads (Fig 11A&C), but the difference diminished in sub-compartments further from secondary roads (Fig 11E). In fact, the positive effect of liana removal on minimum NBR was reversed at the furthest distances from secondary roads, with liana removal reducing minimum NBR compared to reference compartments.

The strength of the signal of commercial liana removal in terms of the proportion of the canopy with decreased NBR was also context dependent. Firstly, as shown in Fig 9C, the reduction in the proportion of the canopy with decreased NBR in treated compared to reference compartments was most pronounced with higher rainfall. This lower proportion of canopy with reduced NBR after treatment was also clearest further from main and secondary roads (Fig 11D & F), and the proportion of the canopy with reduced NBR was actually higher in liana removal than reference compartments when close to secondary roads. There was also a very small influence of terrain steepness on the difference between the proportion of the canopy with reduced NBR in treated and reference compartments (Fig 11B).

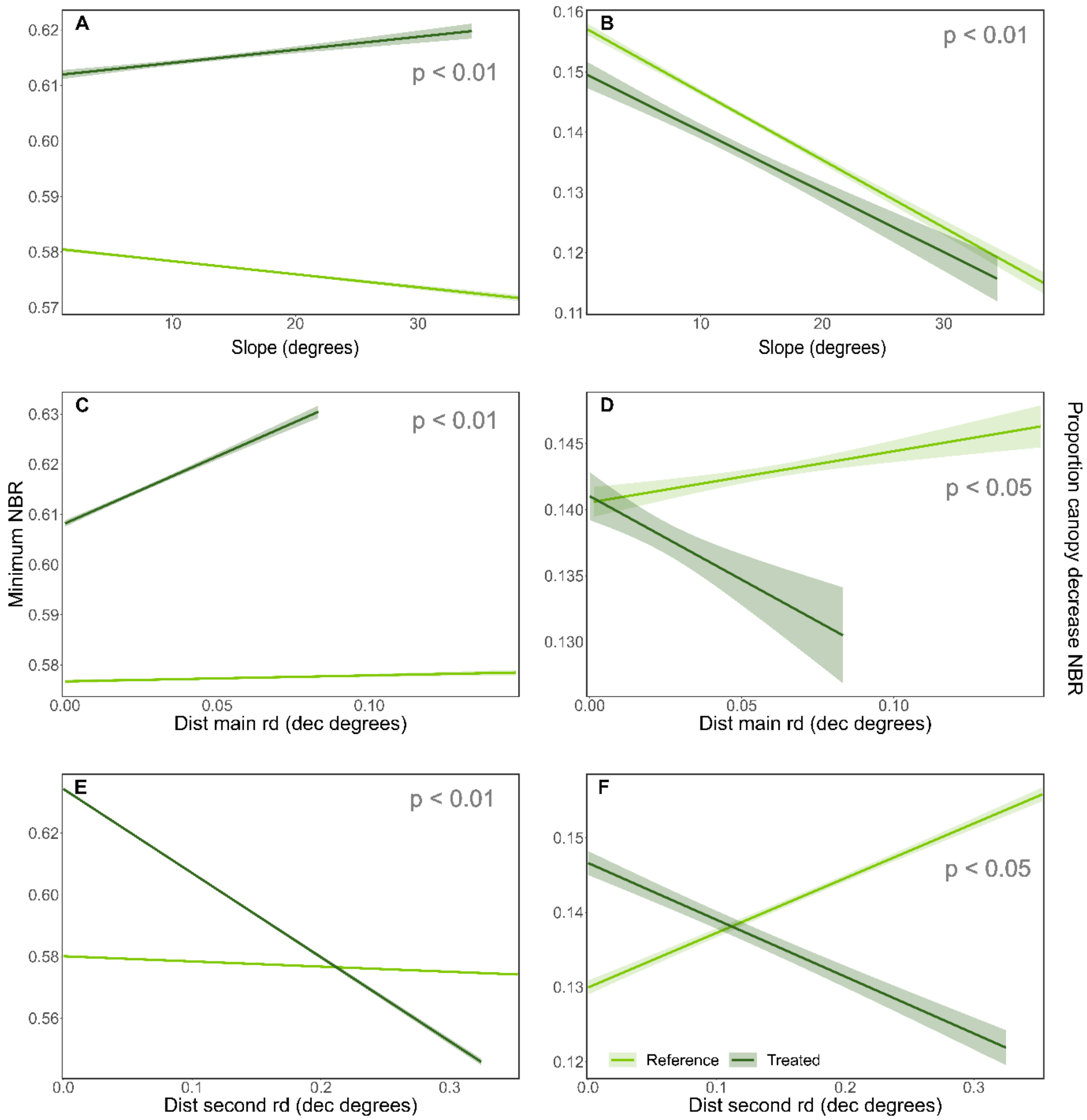


Figure 11: Variation in the effect of commercial liana removal on NBR metrics. Panels A, C, and E show the interaction between steepness of terrain (slope), distance from main and secondary roads and the effect of liana removal on minimum NBR, while panels B, D, and F show the interaction of these variables with the effect of liana removal on the proportion of the canopy with decreased NBR. Lines show the linear relationship for treated and reference compartments separately and the lighter band

around each line shows the SE of the relationship. P-values indicate the level of significance for the coefficient of the interaction, taken from the mixed effects models.

Discussion

This study finds that satellite imagery can detect commercial scale liana removal, but that the signal of commercial liana removal is context dependent. While we detect significant variation in the signal of commercial liana removal, our analyses are unable to determine if liana abundance and crew fatigue are driving this variation. We propose further work to operationalise the use of satellite imagery to validate and monitor this emerging method to restore timber and carbon stocks in large areas of logged tropical forests.

Commercial liana removal can be detected using Sentinel-2 data

Our study provides the first evidence that logged tropical forests treated with liana removal at the commercial scale can be distinguished from untreated forests using satellite-derived data. This supports the conclusion of Finlayson and Hethcoat *et al* (2022) that remote sensing could be an important tool for validating the application of liana removal and monitoring the outcome of the intervention. To detect commercial liana removal, we recommend comparing minimum NBR in treated and reference forest for the year post-treatment. While there was variation in minimum NBR between compartments, likely due to differences in liana abundance, forest structure, and other factors (Finlayson et al., 2022; Putz et al., 2019; Schnitzer and van der Heijden, 2019; Venegas-Gonzalez et al., 2020), the signal of liana removal on minimum NBR was consistent across daily rainfall. Our results also show that liana removal can be detected using the proportion of the canopy with decreased NBR, but this may not be visible when there is less than 5 mm of rainfall daily. These results are a key step in operationalizing the use of satellite data to validate the proposed widespread implementation of liana removal (Finlayson et al., 2022), and could facilitate access to schemes that pay for restoration, enhanced logging sustainability, and carbon sequestration (GOFC-GOLD, 2016).

While our results agree with studies that found a change in the canopy after climber removal (César et al., 2016; Finlayson et al., 2022; O'Brien et al., 2019; Wu et al., 2020), contrary to previous studies we found that commercial liana removal *increases* rather than decreases canopy greenness. The difference in these results could arise from this study assessing a longer period post-removal than in Finlayson and Hethcoat *et al* (2022). Since we only know the annual quarter in which removal occurred, the post-treatment period analysed in this study could be up to 18 months post-treatment if liana removal started at the beginning of a quarter and ended at the end of a quarter. However, an additional analysis of the liana removal experiment in Finlayson and Hethcoat *et al* (2022) shows no increase in canopy greenness in treated compared to untreated forest between 12 to 24-months of treatment (Fig S24).

Alternatively, the increase in canopy greenness could indicate that commercial liana removal achieved lower removal intensity than the 60-100% removal in Finlayson and Hethcoat *et al* (2022). Rather than creating canopy gaps, less intense removal could have promoted quicker re-infestation of lianas or greater growth of new leaves in liberated canopy trees, both of which have high green light reflectance and could influence NBR (Chandler et al., 2021a; Wu et al., 2017). Increased greenness could also occur if fewer canopy gaps arise post liana removal due to a lower abundance of lianas resulting in reduced damage to trees (Estrada-Villegas and Schnitzer, 2018; Garrido-Pérez et al., 2008). Equally, the removal applied to the forests in this study could have promoted the growth of plant species that have higher green reflectance (Taddeo et al., 2019), but such changes to composition and gap formation would take several years to manifest, longer that the study period in this paper. With more precise treatment dates we could observe the temporal changes in canopy greenness post commercial liana removal, determining whether the expected dip in greenness after treatment did occur. Higher resolution imagery, such as from UAVs, could also allow us to determine canopy plant composition (Waite et al., 2019), helping to determine the precise impact of commercial liana removal on the forest canopy.

Satellite imagery detects variation in commercial liana removal signal

We identified several factors that drive variation in the satellite signal of commercial liana removal. Consequently, this study shows that remote sensing could be used to monitor *how* liana removal is applied across large scales such as entire logging concessions, agreeing with the results of Finlayson and Hethcoat *et al* (2022) who found that NBR could differentiate between liana removal that was applied at different intensities. We found that the signal of liana removal can vary at the compartment (200-840 ha) and sub-compartment (4 ha) scale, showing the utility of satellite data to observe the impact and application of liana removal across spatial scales that would be hard to achieve with field surveys (de Almeida et al., 2020; Deluca et al., 2010). Detecting variation in liana removal signal may be particularly useful to identify areas where the removal of all lianas has not been achieved, something which may be important for forest managers seeking to maximise the timber and carbon stocks of logged forests (Finlayson et al., 2022) or to better estimate the expected outcome of the intervention.

While we observed a variable signal of liana removal, it is unclear whether this supports our hypothesis that crew fatigue or liana abundance influence the signal of liana removal. We anticipated that larger compartments, steeper terrain, and greater distance from road would increase crew fatigue and reduce removal completeness (Mills et al., 2019), leading to a smaller signal of liana removal. However, the results tell a mixed story: the most convincing results suggested that steeper terrain and distance from main roads increased the signal of liana removal, potentially indicating higher removal completeness and lower crew fatigue in these contexts, while distance from secondary roads appeared to reduce the liana removal signal. Combined with the fact that compartment size did not influence liana removal signal, this suggests that crew fatigue is not a main driver of variation in removal signal and intensity. Alternatively, since we cannot directly link crew activity to terrain and distance to main or secondary roads, these may be poor proxies of crew fatigue and therefore tell us little about the impact of crew fatigue on liana removal signal.

We also proposed that variation in the satellite signal of commercial liana removal could be due to variation in liana abundance, but we again find minimal support for this. While the lack of impact of liana abundance agrees with Finlayson and Hethcoat *et al* (2022), we expected liana abundance to vary to a greater extent in this larger study area due to greater variation in terrain, edge effects, logging activity, and composition of tree and liana species that influence liana abundance (Addo-Fordjour and Rahmad, 2015a; Campbell et al., 2018; Putz et al., 2019; Schnitzer and Bongers, 2002). However, we found that there was a smaller signal of liana removal on flatter terrain (where some studies report higher liana abundance (Addo-Fordjour and Rahmad, 2015b)) and further from main roads (which create edges that tend to have higher liana abundance (Campbell et al., 2018)). Without direct liana abundance data from the logging compartments in this study, we cannot fully test the influence of liana abundance on the strength of the liana removal signal.

We recommend further studies with detailed information on removal crew activity and liana abundance to determine the causes of variation in satellite-derived-NBR after liana removal. Ultimately, this would reveal the utility of satellite data for detecting commercial liana removal in varying circumstances and for determining removal completeness. Commercial liana removal is poised to enhance selective logging sustainability and restore logged tropical forests globally and our work shows that large-scale monitoring of such interventions could be assisted by remote sensing tools.

CHAPTER 5: General Discussion

Summary

My thesis focusses on developing our understanding, the application, and the monitoring of liana (or climber) removal – an emerging technique that can restore the tree growth and carbon stocks of logged forests towards pre-disturbance levels and enhance the sustainability of selective logging. Conducting a meta-analysis of global liana removal papers (Chapter 2), I found that liana removal vastly enhances timber recovery and carbon sequestration in logged tropical forests – potentially sequestering 7.4 Gigatons of CO_2 over 10 years at less than \$1 $MgCO_2^{-1}$ if applied to logged and secondary forests globally. However, there was poor representation of studies outside the Neotropics. Following this, Chapters 3 and 4 present the first steps for using freely available satellite data to detect and monitor large-scale liana removal in logged tropical forests. I find that both experimental and commercial-scale liana removal cause changes to the canopy that are detectable with satellite data, and that this method can monitor the intensity of liana removal. In Chapter 3, I also provide the first experimental evidence of partial liana removal, showing that leaving 20 or 40% of the forest untreated reduced the canopy disturbance caused by liana removal, and tentatively suggesting that partial removal could reduce the harmful impacts on biodiversity. Ultimately, I conclude that liana removal could be a cost-effective and powerful tool to restore the carbon and timber value of logged tropical forests and find that it can be detected and monitored using a simple remote sensing method based on freely available data. However, serious caution should be taken to preserve the biodiversity value of lianas and forest functions that they provide.

In the following sections I will discuss how findings from this thesis contribute to our understanding of liana ecology, outline the issue of regional bias in tropical ecology research, and detail the next steps for monitoring forest restoration and liana removal research. I will also outline some of the important concerns about liana removal and provide some recommendations for using liana removal to protect and enhance logged tropical forests.

Liana ecology

In addition to developing our understanding of liana removal, my analyses reveal information about the impact of lianas on tropical forests, both confirming the intense burden that lianas have on trees (Schnitzer and Bongers, 2002), but also evidencing the beneficial roles that lianas have. The overwhelmingly positive impact of liana removal on tree growth in Chapter 2 agrees with another meta-analysis on the topic (Estrada-Villegas et al., 2022), and evidences the substantial influence of lianas in terms of timber recovery and aboveground carbon dynamics. Chapter 3 and 4 further demonstrate the impact of lianas, showing that, while liana removal may be thought of as causing relatively minimal canopy disturbance compared to logging, the removal of lianas has an impact on the canopy that is visible from space. This emphasises the high occupancy of lianas in the canopy of logged tropical forests, but also the critical part that lianas play in forest structure and light interception (Meunier et al., 2021c) and the impact that liana removal could have on the survival of fauna and flora that rely on the microclimate buffering provided by a closed canopy (O'Brien et al., 2019; Scheffers et al., 2014). This thesis clearly evidences the substantial impact that lianas have on tropical forests, supporting recent calls to stop this plant group from being overlooked in field and modelling studies (di Porcia e Brugnera et al., 2019), and to retain a proportion of lianas in restoration treatments (Putz et al., 2023).

While experimental and commercial liana removal lead to changes in canopy greenness, the remote sensing methods that I used in this thesis were unable to determine the precise causes of this. I posit that the recovery of the canopy in Chapter 3 could be due to lianas given their documented re-growth after disturbance (Addo-Fordjour et al., 2016; Rocha et al., 2020) and faster growth rate than trees (Schnitzer and Bongers, 2002). However, the growth rate of trees and lianas may be more similar in aseasonal forests (Medina-Vega et al., 2021), such as the forests in Chapter 3 and 4, so it is hard to infer what was driving the recovery of the canopy post liana removal. Better inference about the post-treatment canopy composition using Sentinel-2 imagery could be made by including NDVI, GI, and other spectral bands alongside NBR in models to predict liana load. However, remote sensing imagery greater than 10 m resolution, such drone imagery (van der Heijden et al., 2022; Waite et al., 2019), may be required to draw detailed conclusions about forest dynamics and help to ascertain the nuanced impacts of the restoration technique. For example, individual tree crown liana abundance could be monitored by combining airborne or satellite data with machine learning and some field data on liana infestation (Chandler et al., 2021b) and hyper-spectral data could quantify canopy plant diversity (Clark et al., 2005; Zhao et al., 2018).

In this thesis I also demonstrate that the impacts of lianas and their removal varies across different spatial scales, from small scale (4 ha) treatment blocks (Chapter 3) to field sites in different countries (Chapter 2). Much of the variation in all three chapters could be due to differences in liana abundance resulting from the patchy intensity in timber harvesting that is characteristic of selective logging (Putz et al., 2019). However, less than half of the studies in the meta-analysis reported liana abundance, and this data was not available for the commercial logging compartments in Chapter 4, preventing me from investigating the impact of liana abundance fully in this thesis.

Variation in the impact of liana removal is also likely influenced by the intensity (or "completeness") of removal, but, to my knowledge, there are only two studies that report this data (Mills et al., 2019; Pinard and Putz, 1996). The provision of liana abundance and removal completeness data would make it possible to determine whether there is a critical level of liana abundance or removal intensity under which trees are not released from lianas sufficiently to enhance timber and carbon stocks – information that is critical for the effective restoration of logged forests using liana removal. In cases where collecting liana abundance data in the field is not feasible (Londe et al., 2022), remote sensing methods that detect liana abundance could be employed, as discussed above (Chandler et al., 2021b).

This thesis has provided some new insights into liana ecology in terms of carbon and canopy dynamics, but there is still a lot that we do not yet know about this understudied plant group (van der Heijden et al., 2023). While many studies have investigate the impact of removal on aboveground carbon, for example, there is a dearth of studies considering the impact on belowground carbon pools (Meunier et al., 2021a). Moreover, while lianas are represented as a group of plants that share traits – fast-growing, flexible stems and competing extensively with trees – there is substantial variation between liana species (Coppieters et al., 2022), suggesting that the response of a forest to liana removal may depend heavily on liana species composition. Liana abundance may also determine the outcome of liana removal, but only a few drivers of liana abundance across the tropics, such as precipitation patterns and disturbance, are well-studied (DeWalt et al., 2015; Schnitzer et al., 2014). Finally, lianas provide myriad functions for faunal taxa, but very few studies consider this interaction (Arroyo-Rodriguez et al., 2015), and even fewer the impact of liana removal on faunal species (Schnitzer et al., 2020). Overall, this highlights the need for caution when promoting liana removal to restore the growth of timber trees and carbon stocks in degraded forests – we could be removing lianas from an area of forest that they have not yet been studied and without knowing the full range of impacts that their removal could have. This issue is discussed further in *The unknown impact of forest restoration on biodiversity* section, later in this chapter.

Regional bias in tropical research

Variation in the efficacy of liana removal in my meta-analysis (Chapter 2) is likely to be partly explained by regional or climatic differences. While a meta-analysis by Estrada-Villegas *et al* (2022), which is analogous to mine, concluded no effect of rainfall on liana removal efficacy, the concentration of liana removal studies in the Neotropics, particularly in Panama, could prevent the impact of climate (and region) from being fully assessed in both studies.

The bias of scientific studies towards the Neotropics has been noted in tropical ecology over the past 40 years (Clark, 1985; Deikumah et al., 2014), preventing the generalisability of findings, and causing a concerning lack of understanding of tropical ecology in poorly studied regions. This paucity of research is particularly troubling regarding forest management research, considering that Southeast Asia has some of the highest rates of forest disturbance whilst hosting several biodiversity hotspots (Fisher et al., 2011a), and deforestation has the slowest deceleration rates in the African tropics

(Deikumah et al., 2014). Results in my thesis emphasise the need for greater study of tropical phenomena and forest systems in the Asian and African tropics.

In Chapters 3 and 4 I develop a remote sensing method to detect liana removal based on Sabah, Malaysian Borneo. Since lianas have similar ecology across tropical forests (Schnitzer and Bongers, 2002), I anticipate that liana removal in other regions will also generate a satellite signal based on changes in canopy greenness. However, the magnitude and temporal trend in canopy greenness post-removal may vary due to liana abundance (DeWalt et al., 2015), tree and liana species composition (Schnitzer, 2018; Venegas-Gonzalez et al., 2020), and timber extraction methods in different regions (see Box S1 from (Marshall et al., 2020)). Establishing a network of standardised experiments and commercial liana removal sites across different tropical countries, such as the system of Center for Tropical Forest Science (CTFS) – Forest Global Earth Observatory (ForestGEO) plots that monitor the response of tropical forests to global change (Anderson-teixeira et al., 2014), would be the gold standard to robustly quantify the efficacy and remote-sensing signal of this promising restoration technique. Such a research network would facilitate fast and effective introduction of liana removal across the tropics. I recommend testing the remote sensing signal of liana removal with the numerous existing removal experiments and areas where liana removal has already been applied.

The unknown impact of forest restoration on biodiversity

A critical element of tropical forests that is often overlooked when implementing logged forest restoration is the impact on biodiversity (Cerullo and Edwards, 2019). Pettorelli *et al* (2021) discuss how nature-based solutions – methods that work with nature to solve environmental issues (Seddon et al., 2021) – risk having overall negative impacts on the environment when their focus is too narrow, such as only considering the impact of restoration on carbon. While there are synergies between carbon stocks and biodiversity in many cases, enhancing forest carbon does not universally enhance biodiversity (Strassburg et al., 2010). In the case of liana removal, there are only five studies that have investigated the response of biodiversity compared to more than 60 that focus on the tree response (Estrada-Villegas and Schnitzer, 2018), and the results show that the impact varies depending on taxa, functional group, and tropical region (Benjamin J Adams et al., 2019; Cerullo et al., 2019; Cosset and Edwards, 2017; Edwards et al., 2009; Schnitzer et al., 2020). Therefore, to ensure that restoration actions result in fully functioning, biodiverse tropical ecosystems, and to prevent our attempts to mitigate the climate crisis from accelerating biodiversity loss, it is imperative to assess the influence of restoration methods on biodiversity. This should be a priority for further research into liana removal.

While the results in this thesis show that partial removal has less of an impact on the canopy than complete removal, remote sensing data alone could not determine whether this translates into a smaller impact on biodiversity. A crucial extension to the liana removal experiment in Chapter 3 is measuring changes to faunal communities post-removal and between removal intensities. Baseline data of dung beetle communities and soundscapes has already been collected from this experiment for this purpose. Post-treatment data on faunal communities would explicitly show the impact of liana removal on faunal diversity and whether partial removal reduces biodiversity impacts. Moreover, concurrently measuring tree growth in the experiment, for which baseline data has also been collected, could determine whether there is a trade-off between biodiversity and timber and carbon enhancement. Such trade-offs are inevitable in restoration (Edwards et al., 2021), necessitating data on both the desired and unintended consequences of restoration actions.

In addition to extending my experimental work, research assessing the impact of liana removal on biodiversity should consider other configurations of partial removal. This could include cutting lianas only on trees of commercial interest, such as in Mills *et al* (2019) or Putz *et al* (2023), to enhance growth in valuable timber trees while allowing lianas to persist in the remainder of the forest. Alternatively, liana removal could avoid liana species that are most ecologically important in terms of the volume of food or nesting resources they provide (Addo-Fordjour et al., 2016), for example, or only cut the more common liana species in an attempt to preserve liana diversity. Such nuanced liana removal could also reduce the cost of removal by limiting unnecessary cutting. These methods, however, require substantial knowledge about lianas, a plant group that is chronically understudied (di Porcia e Brugnera et al., 2019). Further research into liana function and diversity would assist with developing methods of liana removal that minimise the impact on biodiversity,

Monitoring large-scale restoration

I have shown that developing methods to monitor restoration can be achieved with freely available and easily accessible remote sensing data products. I focussed on the European Space Agency's Sentinel-2 data, but similar products, such as Planet's Dove satellites, also provide freely available data at similar temporal and spatial resolutions (Roy et al., 2021). While there are more advanced remote sensing methods for landscape monitoring (Reiche et al., 2016), the simplicity of my approach could make it easier to adopt. This is important for countries that may not have the skills or budget for sophisticated solutions, but in which the restoration of logged forests is crucial (Misiukas et al., 2021). Accessible monitoring methods are important to facilitate financial support for restoration projects through schemes such as REDD+ (GOFC-GOLD, 2016), especially if interventions are to be rolled out across the nearly 300 million ha proposed for liana removal in Chapter 2, and if progress towards restoration targets, such as those proposed by the Bonn Challenge, need to be measured (Strassburg et al., 2020).

I recommend further collaboration between researchers in the fields of remote sensing and conservation to generate new solutions to the biodiversity and climate crises, and to explore other ways in which remote sensing could benefit the application of liana removal (Pettorelli et al., 2014). For example, NASA's recently launched Global Ecosystem Dynamics Investigation (GEDI) product provides biomass estimates across the globe at 1 km resolution (Dubayah et al., 2022). Such data will greatly enhance the monitoring of restoration activities aimed at sequestering carbon, but advances are required, potentially using field data, to quantify biomass at greater spatial resolution. GEDI data could also be used to study the impacts of liana removal on forest structure. These data are generated using satellite-borne lidar sensors, providing information about canopy height and indices related to leaf cover (plant area index: PAI) and plant density that have been used to study lianas (Rodríguez-Ronderos et al., 2016; Tymen et al., 2016). For example, we would expect lower PAI in the months following liana removal and increased canopy height as tree growth is enhanced. Combining alternative sources of spectral imagery from drones and Unmanned Aerial Vehicles (UAVs) with liana removal studies could also provide higher resolution imagery that can quantify liana abundance (van der Heijden et al., 2022; Waite et al., 2019), creating more nuanced remote sensing tools for monitoring liana removal and revealing more insights about liana ecology.

Conclusions

Liana removal is an affordable method of forest restoration that has yet untapped potential for enhancing the sustainability of logging and as a nature-based solution to the climate crisis. I find that the positive impact of liana removal is well supported in the Neotropics, making this good candidate for rolling out implementation. However, while the carbon and timber benefits of liana removal are well documented, much is yet unknown about liana ecology and the impacts of liana removal on overall biodiversity and forest function. Hence, research should urgently focus on these two fields to prevent adverse impacts on the already imperilled tropical biodiversity and we implore liana removal initiatives to leave a substantial proportion of the land untreated. Further work is required to minimise the potentially harmful impacts of liana removal, to build certainty in the efficacy of the technique in the African and Asian tropics, and to test my remote sensing monitoring and detection method in wider contexts. There is a wide and exciting scope of further research into liana ecology and removal that will make liana removal a key player during the UN's Decade on Ecosystem Restoration.

SUPPLEMENTARY INFORMATION: Chapter 2

Appendix A: Literature search and screening

Table S4: Literature search strings for Web of Science, SCOPUS and Google Scholar. Two different search strings per database.

Pre disturbance/ any	Database	Search string	Further refinement [a]
Any	Web of Science (All databases)	TOPIC: ((liana* OR vine* OR climb*) AND (remov* OR cut* OR clear* OR thin* OR liberat* OR experiment*) AND (forest*), NOT vineyard, NOT medical	- Refined by Science and Technology - Excluding patent and clinical trials and engineering research domain
Any	SCOPUS	(TITLE-ABS-KEY (liana* OR vine* OR climb*) AND TITLE-ABS-KEY (remov* OR cut* OR clear* OR thin* OR liberat* OR experiment*) AND TITLE-ABS-KEY (forest*) AND NOT TITLE-ABS-KEY (vineyard) AND NOT TITLE-ABS-KEY (medical)) AND (EXCLUDE (SUBJAREA , "ENGI"))	- Exclude Social Sciences and Engineering
Any	Google Scholar	(liana OR vine OR climb) AND (remove OR cut OR clear OR thin OR liberat OR experiment) AND (forest)	-
Pre	Web of Science (All databases)	TOPIC: ((“pre-disturb*” OR “pre-log*” OR “pre-fell*” OR “pre-harvest*” OR “pre-exploit*” OR prefell* OR “prior to log*” OR “prior to disturb*” OR “prior to fell*” OR “prior to harvest*” OR “prior to exploit*”) AND ((liana* OR vine* OR climb*) AND (remov* OR cut*	- Refine by (forest* OR concession* OR “logging operation”) and Science/Technology - Exclude Patents/News

		OR clear* OR thin* OR liberat* OR experiment*)) OR “thinning operation*”) OR TOPIC: (RIL OR "reduced-impact log*" OR “silvicultur* field experiment” OR “planned fell*” OR “planned log*” OR “planned harvest*” OR “FSC cert*”) OR TOPIC: (“silvicultur* treatment*” AND (liana* or vine* OR climb* OR RIL OR "reduced-impact log*"))	
Pre	SCOPUS	(TITLE-ABS-KEY ((("pre-disturb*" OR "pre-log*" OR "pre-fell*" OR "pre-harvest*" OR "pre-exploit*" OR prefell* OR "prior to log*" OR "prior to disturb*" OR "prior to fell*" OR "prior to harvest*" OR "prior to exploit*") AND ((liana* OR vine* OR climb*) AND (remov* OR cut* OR clear* OR thin* OR liberat* OR experiment*)) OR "thinning operation*") OR (ril OR "reduced-impact log*" OR "silvicultur* field experiment" OR "planned fell*" OR "planned log*" OR "planned harvest*" OR "FSC cert*") OR ("silvicultur* treatment*" AND (liana* OR vine* OR climb* OR ril OR "reduced-impact log*")))) AND (forest* OR concession* OR "logging operation")	- Exclude Social Sciences
Pre	Google Scholar	(("pre-logging" OR "pre-harvest" OR "prior to logging") AND (liana OR vine) AND (removal	-

		OR cut OR cutting OR liberation) OR "thinning operation") OR RIL OR "reduced impact logging" OR "planned logging" OR "FSC certified" OR "silviculture treatment"	

[a] Any further filtering applied to the search results after using the indicated search string.

Table S5: Inclusion criteria.

PICO[a]	**Inclusion criteria**
Population	Tropical (latitudes between 26°C North and South, inclusive) Selectively logged (i.e., never fully clear-cut), secondary (clear-cut and regrowth) and undisturbed forest Natural forest system, i.e., not the following: ○ Tree plantation ○ Vineyard Not mangrove forest Experimental removal of climbers; not modelling paper
Intervention	Climber removal (by any method such as cutting with machete or poisoning) explicitly applied
Comparator	Control sites in tropical forest with the same level of disturbance in which no climber removal was applied
Outcome	Tree growth (diameter, basal area, biomass, height, etc.)

[a] PICO elements are used to structure a search strategy and eligibility criteria aimed at answering a specific question (Livoreil et al., 2017).

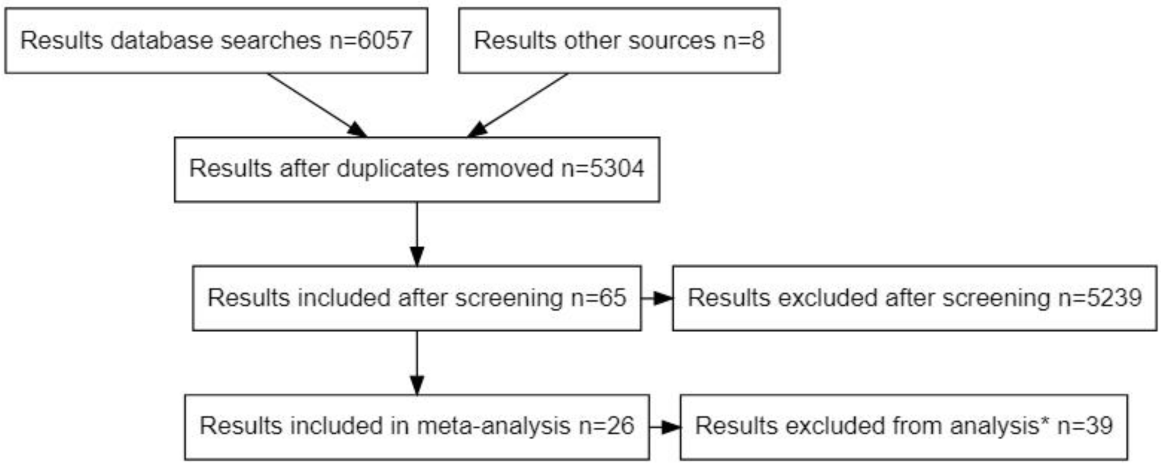


Figure S12: Flow chart illustrating literature search and screening process

*These 39 results were excluded for the following reasons: additional vegetation management, duplicated data, not useful tree growth metric, mean growth data unavailable for climber removal and/or control treatments.

Table S6: Studies not included in meta-analysis after data extraction and full text assessment.

Author and year	Excluded reason category	Reason, full
Cardoso et al 2014	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Heuberger et al 2002	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Pena-Claros et al 2002	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Putz et al 1984	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Schiotz et al 2006	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Schwartz et al 2013	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Villegas et al 2009	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Guaragiata 1999	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Butarbutar et al 2019	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Coimbra Cordeiro et al 2019	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Oliveria et al (unpublished thesis chapter)	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Minh Quang et al 2020	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Truong et al 2021	Additional vegetation removal	Additional silvicultural treatment other than just climber or understory removal (e.g., thinning of competing trees or planting of seedlings)
Schnitzer et al 2014	Duplicate data	Only data available (biomass growth) represents the same tree growth (using the same plots and treatments) as *Schnitzer et al, 2010*
Schnitzer et al 2004	Duplicate data	Duplicate data using same plots and treatments as *Parren, 2003*
Taffarel and de Carvalho et al 2014	Duplicate data	Only has growth data for individual species that contribute to overall growth in *de Souza et al, 2015*

Vatraz et al 2012		Only has growth data for individual species that contribute to overall growth in *de Souza et al, 2015*
Venturoli and Carvalho et al 2015		Duplicate data from same experimental sites in *Freitas Xavier et al, 2017*
Venturoli and Franco et al 2015		Duplicate data from same experimental sites in *Freitas Xavier et al, 2017*
Estrada-Villegas 2019		Duplicate data in *Estrada-Villegas et al, 2020*
Douglas 1996	Not useful growth metric	Only has data for post-treatment size of tallest tree
Campanello et al 2012	Not useful growth metric: Net growth	Tree growth in basal area of entire plot, could include recruitment
Forshed 2006		Net growth including recruitment and mortality of trees
Forshed et al 2008		Net growth including recruitment and mortality of trees
Mendez-Toribio et al 2019		Net growth including recruitment and mortality of trees
Philipson et al 2020		Net growth including recruitment of trees
Lussetti 2017	Missing mean growth data	-
Okali 1987		
Strugnell 1939		
Vidal et al 2016		
West et al 2014		
Do et al 2019	No control data	Not appropriate controls and sampling to assess impact of liana removal on tree growth
Inada and Widiyatno et al 2017		-
Pena-Claros and Fredericksen et al 2008		

Pena-Claros and Peters et al 2008		
Schulze, 2003		
Vidal et al 2002		
de Avila, 2016		
Roopsind et al 2008		

Table S7: List of studies included in meta-analysis and summary of study details.

Study ID	Author & Year	Country	Latitude	Longitude	Elevation (m.a.s.l)	Total rainfall (mm year^{-1})	Mean temp (°C)	Dry season length (months)	Disturbance type [a]	Removal before disturbance (months)	Disturbance before removal (months)	Removal number (max) [b]	Removal method [c]
2	Alvarez-Cansino et al 2015	Panama	9.11	-79.85	82	2866	26.3	4	S	NA	660	4	all
33	Campanello et al 2007	Argentina	-25.97	-54.22	255	1883	21.6	0	SL	NA	2	1	all
43	Cesar et al 2016	Brazil	-22.71	-47.63	555	1583	22.4	6	SL	NA	420	3	all
69	Freitas Xavier et al 2017	Brazil	-15.85	-48.96	776	1732	23.7	5	SL	NA	132	0	all
74	Gerwing 2001	Brazil	-3	-50	65	2325	27.8	5	SL	NA	96	1	all
79	Grauel et al 2004	Panama	8.12	-77.87	13	1495	26.8	5	SL	NA	120	0	all
82	Grogan et al 2009	Brazil	-7.83	-50.27	246	1878	26.5	4	SL	NA	48	1	tree centred
86	van der Heijden et al 2015	Panama	9.11	-79.85	82	2896	26.5	4	S	NA	720	13	all
87	van der Heijden et al 2019	Panama	9.11	-79.85	82	2870	26.6	4	S	NA	720	21	all
123	Lussetti et al 2016	Malaysia	4.55	117.03	533	2743	24.7	0	SL	12	NA	0	all
125	Marshall et al 2017	Tanzania	-7.82	36.98	290	1144	21.5	7	SL+S	NA	384	10	tree centred
126	Martinez-Izquierdo et al 2016	Panama	9.11	-79.85	82	2848	26.4	4	S	NA	720	12	all
133	Mills et al 2019	Belize	17.25	-89	80	2048	25.7	3	NA	1	NA	0	tree centred
139	O'Brien et al 2019	Malaysia	5.09	117.64	144	2964	25.7	0	SL	NA	240	1	all
146	Parren 2003	Cameroon	3	10	475	2393	25.4	4	SL+S	9	NA	0	all
177	Schnitzer et al 2010	Panama	9.17	-79.85	86	2943	26.3	4	S	NA	600	27	all

Table S7 continued:

Study ID	Author & Year	Country	Latitude	Longitude	Elevation (m.a.s.l)	Total rainfall (mm year^{-1})	Mean temp (°C)	Dry season length (months)	Disturbance type(s) [a]	Removal before disturbance (months)	Disturbance before removal (months)	Removal repeats (max) [b]	Removal method [c]
198	de Souza et al 2014	Brazil	-3.62	-48.62	141	2490	28	4	SL	NA	1	0	tree centred
205	Taffarel et al 2014	Brazil	-3.62	-48.62	141	2490	28	4	SL	NA	1	0	tree centred
219	Verwer et al 2008	Bolivia	-15.78	-62.92	233	1133	24.7	6	SL	NA	1	0	tree centred
224	Wright et al 2015	Panama	9.17	-79.85	86	2920	26.3	4	S	NA	720	8	all
226	Perez-Salicrup 2001	Bolivia	-14.75	-62	194	1237	25.7	6	NA	NA	NA	0	all
227	Perez-Salicrup et al 2000	Bolivia	-14.75	-62	194	1347	25.6	6	NA	NA	NA	0	tree centred
246	Estrada-Villegas et al 2020	Panama	9.2	-79.75	198	2871	26.7	4	S	NA	270	4	all
271	Venegas-Gonzalez et al 2020	Brazil	-22.71	-47.63	555	1501	22.3	4	SL	NA	384	0	all
323	Estrada-Villegas et al 2021	Panama	7.43	80.18	73	2296	27.5	2	SL+S	NA	420	4	all

[a] Type of forest disturbance: SL = selectively logged, S = secondary forest (forest regrowth after clear cutting), NA = undisturbed forest

[b] Number of times climber removal was repeated

[c] Removal of all climbers from the plot/stand (“all”), or removal of climbers from focal trees only (“tree centred”)

Appendix B: Data extraction and explanatory variables

Tree growth response data

We used relative growth rate (RGR) where possible to standardise tree growth by tree size, accounting for differences in growth rate across life stages and sizes. Tree growth was typically reported as change in diameter (*d*) at breast height (dbh) (20 studies), but also as change in biomass, basal area, or height (six studies). Some studies calculated RGR using equation (1) or (2) below, and we calculated RGR using the same equations where possible if RGR was not given. When initial size was not available, but the study used a narrow tree size class (e.g., 5-10 cm dbh), RGR was estimated using the mid-point of the size class. RGR was not available nor could be estimated for seven studies. Whether a study response variable is based on RGR is included in the study quality index (see Table 2).

$$RGR = \frac{\ln(d_1) - \ln(d_0)}{t_1 - t_0} \tag{1}$$

$$RGR = \left(\frac{1}{d_0}\right)\left(\frac{d_1 - d_0}{t_1 - t_0}\right) \tag{2}$$

d_0 initial diameter/basal area/biomass/height*;* d_1 final diameter/basal area/biomass/height*;*

t_0 initial time*;* t_1 final time.

If a study reported multiple growth rates for individual species or subsets of species, we aggregated them to obtain a single mean growth rate per study. Some studies measured the growth of the whole tree community in the experimental site, while others only measured a subset of species. We included the number of species measured in each study as a covariate. If a study did not state the number of species measured, we took the average number of species in the ‘whole community’ or ‘subset community’ from the other studies in the meta-analysis (160 and 10 species, respectively).

Biomass response data

We quantified the effect of climber removal on biomass from a subset of data that met two criteria: 1) the outcome of climber removal on biomass was available or could be calculated, and 2) data was from trees 5 cm dbh or greater as they have the greatest contribution to aboveground biomass. This resulted in 12 studies. Biomass was already reported in four of these studies, but we calculated biomass in the remaining studies from the tree growth data. In studies for which we had individual tree diameter measurements (N=5), we estimated the biomass using a pan-tropical allometry for moist tropical forests using the equation:

$$AGB = \exp(-2.024 - 0.896E + \log(W) + 2.795\log(D) - 0.0461[\log(D)]^2) \qquad (3)$$

(Chave et al., 2014)

E is a variable that represents climatic factors for each region that constrain the height-diameter relationship when height measurements are not available; W is wood density (g cm^3); $D$is the diameter (cm). W was acquired from the global wood density database (Zanne et al., 2009). We used wood density specific to species and region where possible, followed by median wood density for genus and region, or median wood density for the dataset when species name was missing or did not match the wood density database. When individual tree level measurements were unavailable but narrow diameter classes were reported (N=3 studies), we estimated change in biomass using the mid-point of the reported diameter class.

Missing data

We contacted authors when there were missing values for growth rate, variance, sample size, or other study design data. If data were still not available but presented in a figure, we extracted values using DigitizeIT software (Bormann, 2020). Studies were excluded from analysis if mean tree growth or biomass increase was still unavailable. However, we used multiple imputation to estimate missing variances following(Kambach et al., 2020)) and using the *mice* R package (van Buuren and Groothuis-Oudshoorn, 2011). We imputed missing variance using the linear relationship between variance, mean growth and sample size because mean and sample size explained a high percentage of the variance. We ran 10 imputation iterations, generating 10 tree growth and biomass datasets.

Variables relating to the efficacy of climber removal

Climate measures were obtained for each study using high-resolution (0.5 x 0.5 degree) data from the Climate Research Unit (CRU) (Harris et al., 2020). The variables calculated were mean annual temperature, total annual precipitation, presence of dry season, dry season length (dry season defined as any month <100mm total rainfall), total dry season precipitation, and mean dry season temperature. We used the International Centre for Tropical Agriculture (CIAT) dataset to obtain elevation for each study site, using the site coordinates with a 1 km buffer (Jarvis et al., 2008). All other potential explanatory variables were extracted directly from the paper.

Table S8: Criteria for ordinal study quality score. Study quality is included in models as to account for variation due to study design.

Total score and quality category	**Score***	**Criteria**
Low <6	1 1 1 (0.33 per part)	Sample size <4 Sample area <1000 / #trees <50 Design - just post-treatment + not RGR; - >10km between treatment and control plots; - different forest disturbance history in treatment and control plots
Med 6-7	2 2 2 (0.66 per part)	Sample size 4-10 Sample area 1000-10,000 / #trees 50-100 Design - before/after data without RGR (or vice-versa); - 1-10 km between treatment and control plots; - Different disturbance type/ logging type/ secondary forest age between treatment and control plots
High >=8	3 3 3 (1 per part)	Sample size >10 Sample area >10,000 / #trees >100 Design - before/after design + RGR; - <1km between treatment and control; - no differences in disturbance history between treatment and control

***** score of .99 is rounded up. E.g., 7.99 rounds up to 8

Table S9: Explanatory parameters that could not be included in the tree growth models for Objective 2 (main nor supplementary models).

Driver of variation	Parameter	Reason not assessed in Objective 2
Region and climate	Region	Too few studies in each category level / correlated with other variable
	Continent	Too few studies in each category level / correlated with other variable
	KPG Climate Classification	Too few studies in each category level / correlated with other variable
	Seasonality	Too few studies in each category level / correlated with other variable
Forest type and disturbance	Forest disturbance context	Too few studies in each category level / correlated with other variable
	Secondary forest age	Too few studies in secondary forest
	Liana abundance	Too few studies reporting data
	Tree species or functional group	Too few studies reporting data
Liana removal method	LR pre or post disturbance	Too few studies in each category level / correlated with other variable
	Time LR pre disturbance	Too few studies

Appendix C: Model specifications

Table S10: Model specification for quantifying the magnitude of climber removal efficacy (objective 1) and for assessing drivers of variation in efficacy (objective 2). All models used SMD (standardised mean difference) effect size as response variable, were weighted by 1/SMD variance, and included study as random effect. ‘Nuisance’ variables of study quality, number of species used to calculate mean growth, and time elapsed between removal and measurement were also included as fixed effects in all models.

Objective number	***Objective***	***Response variable***	***Explanatory variable***
Objective 1.1	**Quantify magnitude of, and variance in, efficacy of CR to promote tree growth**	• Tree growth	• *None*
Objective 1.2	**Quantify magnitude of, and variance in, efficacy of CR to promote aboveground biomass accumulation**	• AGB change	• *None*
Objective 2.1	**Assess the drivers of variation in**	• Tree growth	• Logged forest (Y/N), • Repeat removal (Y/N), • Elevation,

	efficacy of CR to promote tree growth		• Dry season length, • Annual precipitation, • Average temperature • Removal method (remove climbers on focal tree / climbers from entire area)
Objective 2.2a	**Assess the drivers of variation in efficacy of CR to promote aboveground biomass accumulation**	• AGB change	• Repeat removal (number of times repeated)
Objective 2.2b			• Repeat removal (Y/N)
Objective 2.2c			• Repeat removal (number of times repeated) • Time since disturbance (time between disturbance and treatment)

Table S11: Supplementary models assessing additional drivers of variation in climber removal efficacy for tree growth (objective 2.1) which could not be included in the main model. Each model includes an additional explanatory variable that could not be assessed in the objective 2 model in the main text, highlighted in bold. All models used SMD (standardised mean difference) as response variable, were weighted by 1/SMD variance and included study as random effect. ‘Nuisance’ variables study quality, number of species used to calculate mean growth and time elapsed between removal and measurement were also included as fixed effects in all models.

Additional variable assessed	***Explanatory variable***
Latitude	• Logged forest (Y/N), • Repeat removal (Y/N),

	• Elevation, • Dry season length, • Annual precipitation, • Average temperature • **Latitude**
Number of times removal repeated	• Logged forest (Y/N), • Elevation, • Dry season length, • Annual precipitation, • Average temperature, • **Repeat removal (number of times repeated)**
Time since disturbance **(post-treatment studies only)**	• Logged forest (Y/N), • Repeat removal (Y/N), • Elevation, • Average temperature, • **Time since disturbance (time between disturbance and treatment)**
Dry season climate **(dry season studies only)**	• Logged forest (Y/N), • Repeat removal (Y/N), • Elevation, • Dry season length, • **Dry season precipitation,** • **Dry season temperature**

Appendix D: Additional analyses

Sensitivity analysis and publication bias

We found some evidence for publication bias in our meta-analysis. While there was no relationship between publication year and effect size for tree growth and biomass analyses (Appendix D, Figure 1, 2), the funnel plots of effect size against variance were asymmetric (Appendix D, Figure 3, 4), and the Eggers test indicates a significant relationship between effect size and variance ($p < 0.01$ and $p < 0.0001$ for tree growth and biomass, respectively). However, fail-safe numbers indicate that the meta-analysis results are robust. According to the Rosenberg and Rosenthal methods, there would need to be between 310-560 additional studies with null results to reduce the significance level of the tree growth summary effect size to above alpha = 0.05, and 118-294 for the biomass effect size. Alternatively, according to the Orwin method, there would need to be 26 and 12 further studies with null results to reduce the tree growth and biomass summary effect sizes by half, respectively.

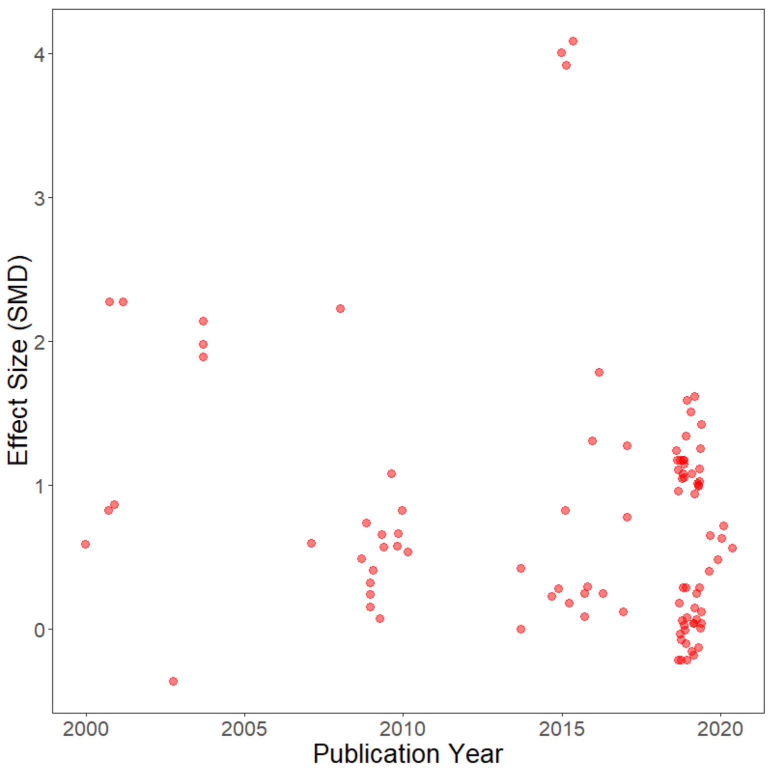


Figure S13: Relationship between publication year and tree growth effect size. Publication year is plotted against average study tree growth effect size (average of individual effect sizes in each study), predicted from growth summary ES model (objective 1.1), to assess publication bias.

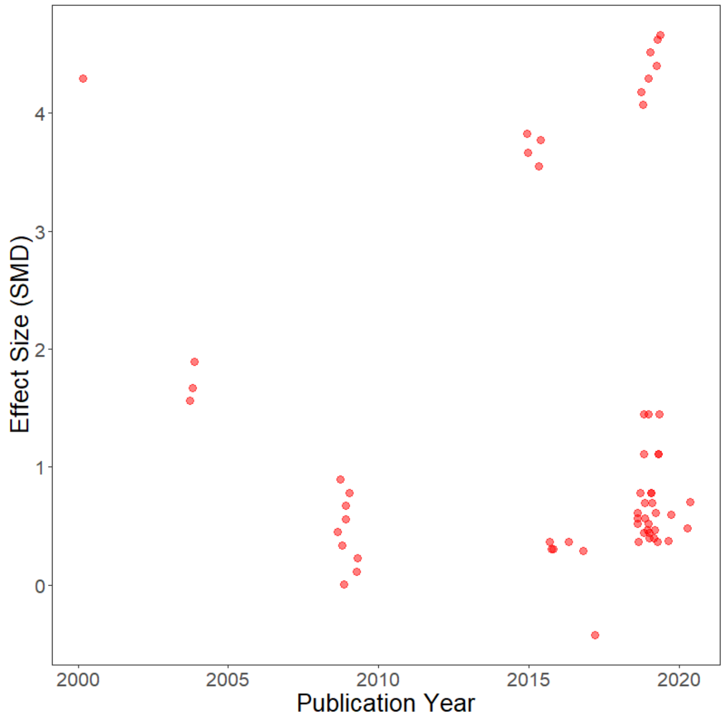


Figure S14: Relationship between publication year and AGB effect size. Publication year is plotted against average study AGB effect size (average of individual effect sizes in each study), predicted from growth summary ES model (objective 1.2), to assess publication bias.

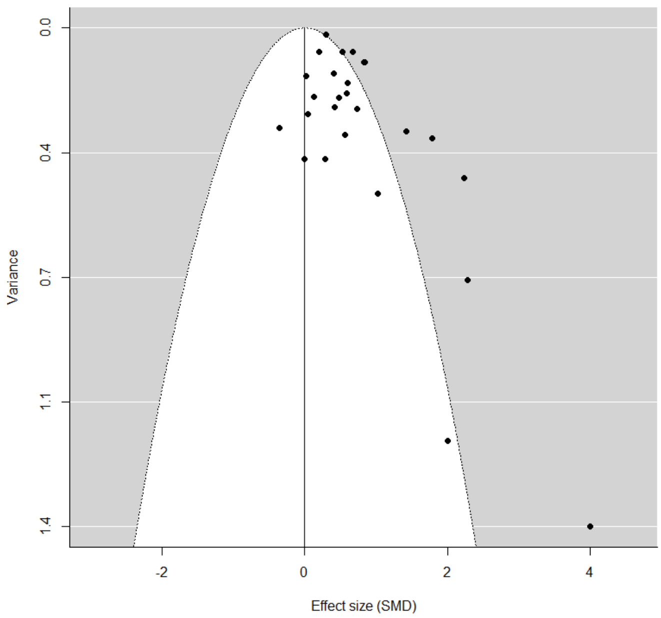


Figure S15: Funnel plot of study tree growth effect size (SMD) (average of individual effect sizes in each study predicted from objective 1.1 model) against effect size variance.

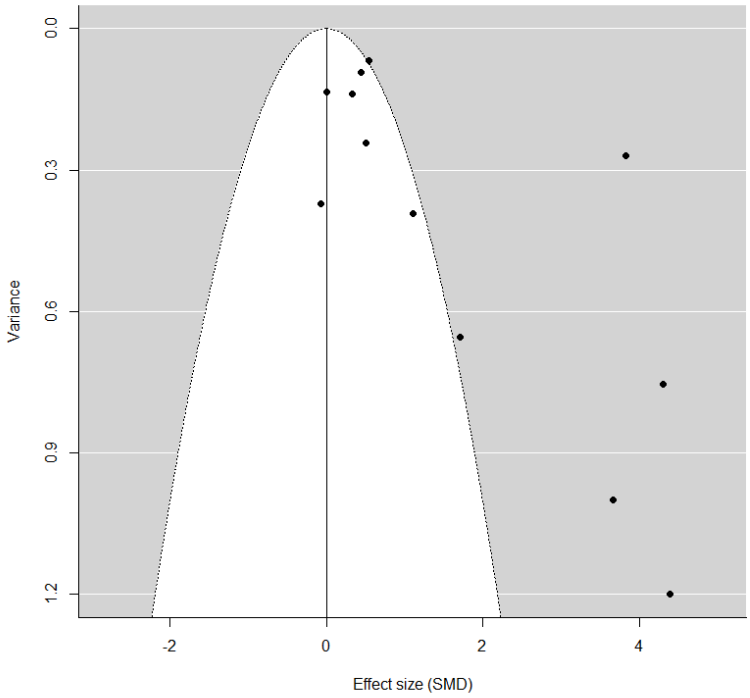


Figure S16: Funnel plot of biomass study effect size (SMD) (average of individual effect sizes in each study predicted from objective 1.2 model) against effect size variance.

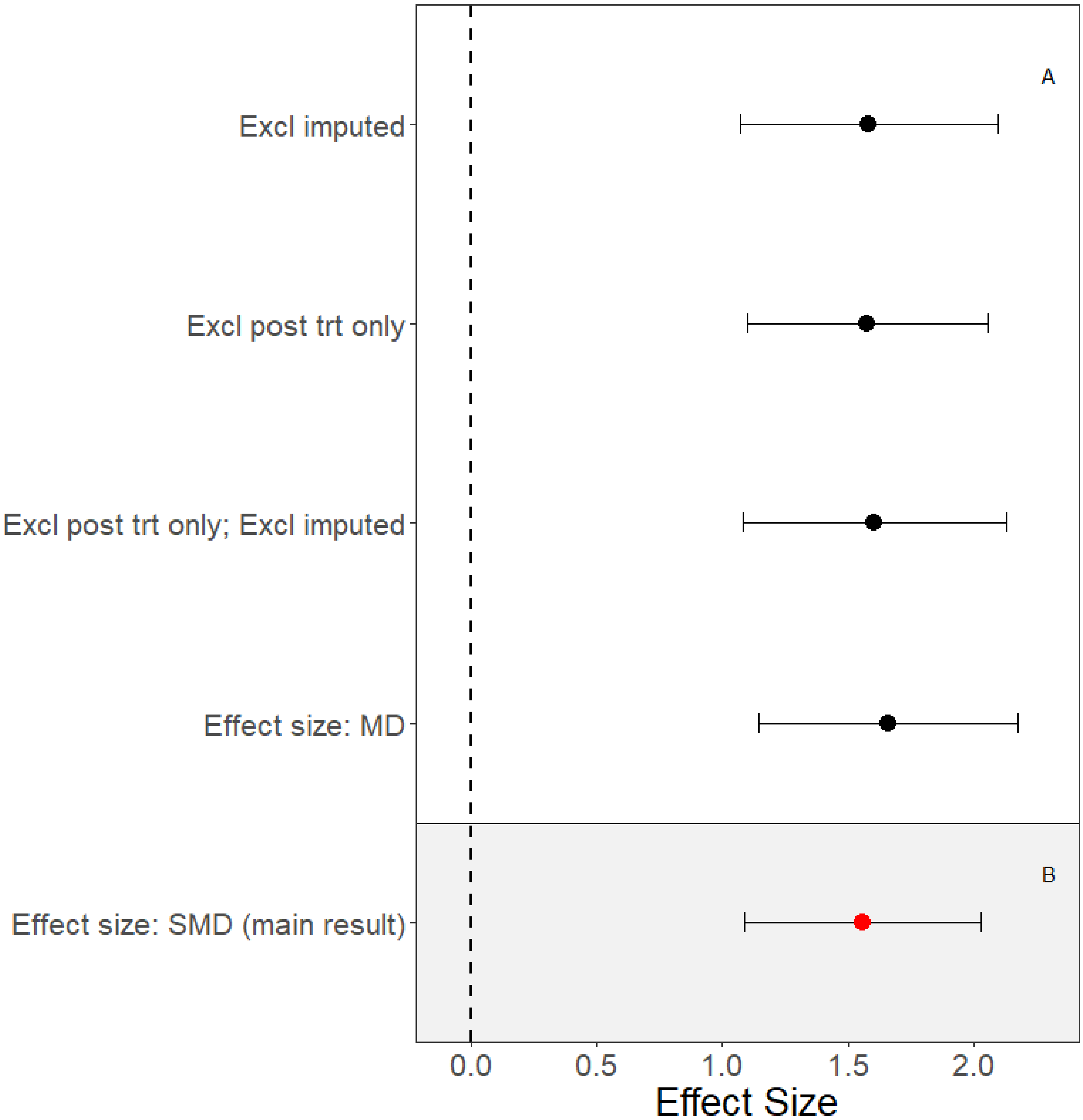


Figure S17: Efficacy of climber removal for enhancing tree growth depending on data type and effect size, calculated using model for objective 1.1 (response variable tree growth) with different combinations of data: a) excluding imputed data, excluding studies with just post-treatment data, excluding both data, using MD as effect size rather than SMD, b) final tree growth summary effect size used in the manuscript: including imputed data calculated using linear regression, studies with just post-treatment data and using SMD.

Table S12: Magnitude and direction of climber removal efficacy on tree growth without van der Heijden et al (2015) outlier. Results of models for Objective 1.1 (tree growth) without outlier. ‘Tree growth Effect Size (ES)’ are the intercept of the model and show the number of times greater tree growth with climber removal versus untreated control plots. Results are the average of 10 Linear Mixed Models using 10 datasets imputed using linear regression, including the study with just post-treatment data (N=25 studies). See Supplementary Information, Appendix C for full description of models. Bolded effect sizes indicate level of significance at either 0.05,0.01, or 0.001.

Objective	***Fixed effect***	***Estimate (SE)***	***Confidence Intervals***	***Degrees of Freedom***
Objective 1.1: Tree growth	***<u>Tree growth ES</u>***	**1.38 (0.19)*** **	0.98 – 1.78	27
	Study quality High:Low	**-1.12 (0.34)** **	-1.81 – -0.43	64
	Study quality High:Med	**-1.11 (0.14)*** **	-1.39 – -0.83	86
	Number of species	0.00 (0.00)	0.00 – 0.00	89
	Time elapsed since removal	**0.01 (0.00)*** **	0.00 – 0.01	89

**p<0.05, **p<0.01. ***p<0.001*

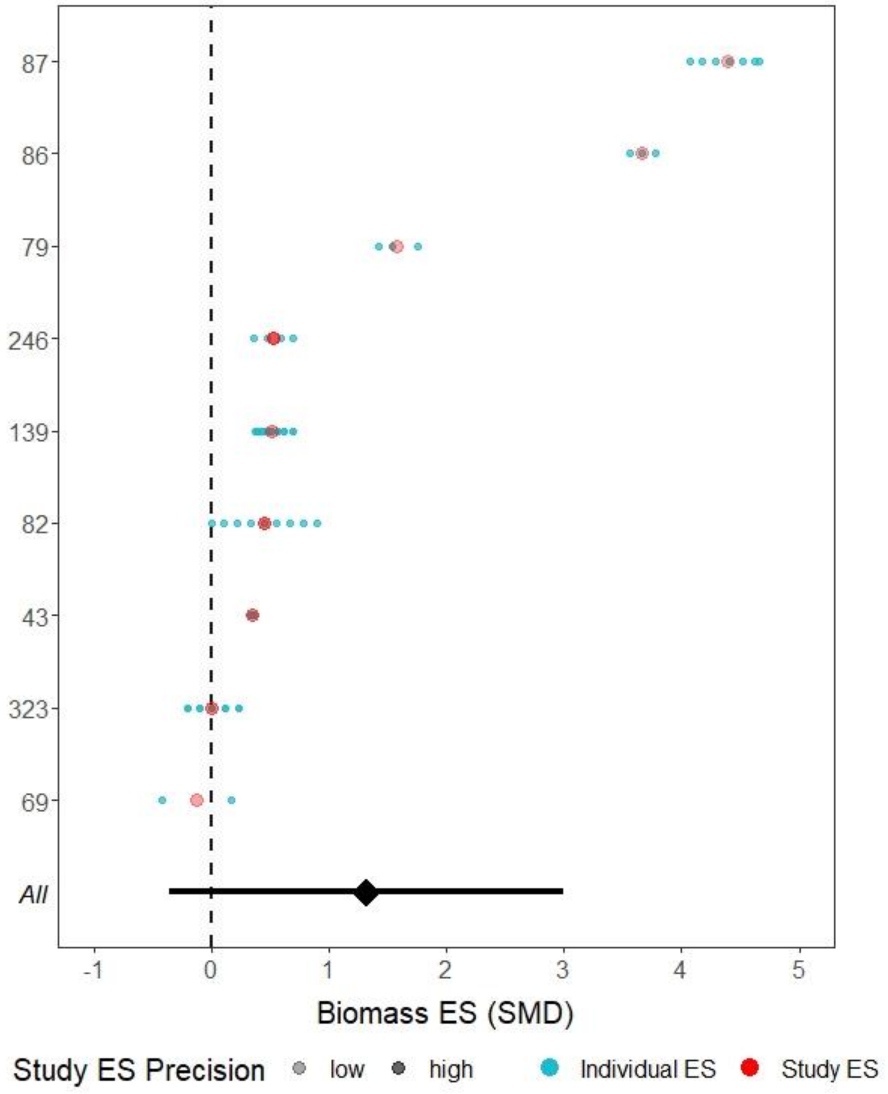


Figure S18: Overall, individual, and study average effect sizes (ES) of climber removal for promoting AGB accumulation, without including imputed data. Blue dots are individual effect sizes within a study, predicted from the models for Objective 1.2 and averaged for all imputed datasets. Red circles are the study ES (the average of the individual ES); the size of the circle represents precision of the study ES and is proportional to the inverse of the variance of the individual effect sizes, averaged by study. The black diamond at the bottom of each figure is the overall summary effect size of climber removal for promoting tree growth and biomass, taken from the intercept of the models for Objective 1.2 when continuous covariates are at their mean value and study quality is set to high; error bar shows 95% credible intervals.

Table S13: Additional drivers of variation in the efficacy of climber removal for tree growth. Table shows results of supplementary models for objective 2.1, averaged from 10 Linear Mixed Models using 10 imputed datasets (imputed using linear regression) and including one study with just post-treatment data (N=26 studies). Response variable is tree growth, see full model details in Supplementary Information, Appendix C.

Model	*Explanatory parameter*	*Estimate (SE)*	*Degrees of Freedom*
a	***Time elapsed since removal***	**0.28 (0.07)*** **	85
	Repeat removal (Y/N)	-0.34 (0.20)	91
	Logged forest	-0.30 (0.61)	16
	Dry season length	0.56 (0.38)	16
	Annual precip	0.26 (0.40)	17
	Annual temp	-0.7 (0.27)	18
	Elevation	-0.26 (0.29)	22
	Latitude	-0.03 (0.34)	19
b	***Time elapsed since removal***	**0.21 (0.07)** **	84
	Repeat removal (number)	0.12 (0.13)	87
	Logged forest	-0.32 (0.60)	17
	Dry season length	0.39 (0.40)	20
	Annual temp	0.15 (0.27)	21
	Annual precip	0.02 (0.37)	20
	Elevation	-0.04 (0.28)	22
c	***Time elapsed since removal***	**0.31 (0.09)** **	60
	Repeat removal (Y/N)	-0.37 (0.23)	68
	Time since disturbance	-0.01 (0.22)	15
	Dry season length	0.53 (0.36)	13
	Elevation	-0.29 (0.32)	14

	Annual temp	-0.01 (0.24)	15
d	***Time elapsed since removal***	**0.17 (0.07)***	43
	Repeat removal (Y/N)	0.04 (0.22)	53
	Logged forest	0.03 (0.54)	14
	Dry season length	0.95 (0.58)	14
	Elevation	-0.29 (0.31)	15
	Dry season annual temp	0.02 (0.34)	15
	Dry season annual precip	-0.15 (0.29)	14

**p<0.05, **p<0.01. ***p<0.001*

SUPPLEMENTARY INFORMATION Chapter 3

Table S14: List of dates of Sentinel-2 images used within the study. Representing the first image acquisition in study region through until one year after the final block was treated with liana removal

Month Yr	S2 image dates	No. images
Dec 2018	17/12/2018; 22/12/2018	2
Jan 2019	06/01/2019; 11/01/2019; 21/01/2019	3
Feb 2019	10/02/2019; 15/02/2019; 20/02/2019; 25/02/2019	4
Mar 2019	02/03/2019; 07/03/2019; 12/03/2019; 17/03/2019; 22/03/2019; 27/03/2019	6
Apr 2019	06/04/2019; 16/04/2019; 21/04/2019	3
May 2019	06/05/2019; 11/05/2019; 16/05/2019; 26/05/2019; 31/05/2019	5
June 2019	25/06/2019	1
July 2019	20/07/2019; 30/07/2019	2
Aug 2019	14/08/2019; 19/08/2019	2
Sep 2019	08/09/2019; 23/09/2019; 28/09/2019	3
Oct 2019	03/10/2019; 08/10/2019; 13/10/2019; 18/10/2019; 28/10/2019	5
Nov 2019	07/11/2019; 12/11/2019; 17/11/2019; 22/11/2019; 27/11/2019	5
Dec 2019	07/12/2019; 17/12/2019; 22/12/2019; 27/12/2019	4
Jan 2020	06/01/2020	1
Feb 2020	20/02/2020; 25/02/2020	2
Mar 2020	16/03/2020; 21/03/2020; 26/03/2020; 31/03/2020	4
Apr 2020	05/04/2020; 10/04/2020; 15/04/2020; 20/04/2020; 30/04/2020	5
May 2020	05/05/2020; 10/05/2020; 15/05/2020; 30/05/2020	4
June 2020	04/06/2020; 24/06/2020	2
Jul 2020	14/07/2020; 19/07/2020; 29/07/2020	3
Aug 2020	03/08/2020; 08/08/2020; 23/08/2020; 28/08/2020	4
Sept 2020	02/09/2020; 07/09/2020; 12/09/2020; 27/09/2020	4
Oct 2020	02/10/2020; 07/10/2020	2
Nov 2020	06/11/2020; 16/11/2020	2

Other satellite imagery and metrics

We calculated three additional vegetation indices to verify results in the main text that are based on NBR. The greenness index (GI) was used as it focuses solely on leaf pigments and correlates with liana infestation (Chandler et al., 2021b), normalized difference vegetation index (NDVI) includes spectral bands that are influenced by leaf pigment, indicating the "greenness" of the canopy (Huete et al., 1997), and enhanced vegetation index (EVI) is similar to NDVI but optimises the vegetation signal in high plant biomass regions (Huete et al., 1997). Lower values of GI, NDVI, and EVI within forests indicate lower concentrations of leaf pigments and signal fewer photosynthetically active leaves in the canopy, canopy gaps, or leaves with lower pigment content. We calculated GI, NDVI, and EVI using equations S1-S3, below. As remnant clouds and cloud shadows impacted the calculation of EVI, NDVI, and GI, we excluded pixels with EVI, NDVI, and GI values less than the lower 99% confidence interval pre-treatment (Table S2). We calculated the median and proportion of canopy with decrease in these three additional indices.

$$GI = G/(R+B+G) \qquad (S1)$$

$$NDVI = (N-R)/(N+R) \qquad (S2)$$

$$EVI = 2.5\,(N-R)/((N+6R-7.5B)+1) \qquad (S3)$$

Letters indicate spectral reflectance bands: G = green (560 nm); R = red (664.5 nm); B = blue (496.6 nm); N = near-infrared (835.1 nm).

We also explored different summary metrics for each vegetation index, including minimum NBR to verify the effect of liana removal on median NBR. We expected minimum NBR to have a larger response to liana removal than median NBR, but we focus the results of the manuscript on median NBR due to the lower variation in this summary statistic. This metric was calculated by creating a mosaic of the minimum NBR value per pixel across all S2 images in each month post-treatment. This was summarised per treatment block, resulting in the median minimum NBR pixel value per treatment block per month post-treatment. We also found that there was a clearer signal of liana removal when comparing the proportion of pixels with >5% change in NBR than the proportion of pixels with more than z-score change between treatment and control blocks.

There was no visual signal of liana removal in radar (Sentinel-1), so we did not pursue this remote sensing data. Liana removal was more visible with Sentinel-2 than with Landsat 8, and we thought that the higher resolution S2 data would be more suited to detecting liana removal, so we focussed on S2 images.

Table S15: Mean and confidence intervals of three vegetation indices before treatment to calculate cloud filters for Google Earth Engine. Data are shown for Surface Reflectance Sentinel-2 data (SR) [used in the study] and Top of Atmosphere Sentinel-2 data (TOA), using a 150 m buffer around cloud pixels, no buffer [used in the study], or the in-built cloud buffer for TOA.

Index S2	Mean	SE	Lower 99% CI	Upper 99% CI
EVI SR; 150m buffer	0.328695	0.003679	0.319195	0.338195
EVI SR; No buffer	0.31591	0.00259	0.309231	0.322589
EVI TOA; in-built cloud mask	0.332972	0.00168	0.328643	0.337301
NDVI SR; 150m buffer	0.787811	0.001918	0.782858	0.792764
NDVI SR; No buffer	0.715604	0.002671	0.708716	0.722492
NDVI TOA; in-built cloud mask	0.489484	0.002663	0.482621	0.496347
GI SR; 150m buffer	0.40213	0.001084	0.399331	0.40493
GI SR; No buffer	0.383332	0.000694	0.381541	0.385122
GI TOA; in-built cloud mask	0.331781	0.00016	0.33137	0.332192

Table S16: Effect of different intensities of removal (60, 80, 100%) on the canopy across 1- and 12- months post treatment in terms of median NBR, proportion of the canopy with decreased NBR ("Prop decr NBR"), intact patch size ("Area intact patch"), and aggregation ("Agg intact patch"). Table gives the coefficient for the difference between each intensity of liana removal and control, the difference between removal treatments, and, where significant, the influence of rain. 1-month coefficients are from linear mixed effects models, and 12-month coefficients are from generalized additive models. Response variables are normalized prior to running models. Row and column fixed effects are not presented but did absorb some variation.

Months post treatment	**Metric**	**Contract / fixed effect**	**Estimate**	**SE**	**P-value**
1	Median NBR	60 - 0	-0.58	0.164	**0.003**
	Median NBR	80 - 0	-0.59	0.164	**0.002**
	Median NBR	80 - 60	-0.01	0.155	> 0.999
	Median NBR	100 - 0	-1.03	0.163	**< 0.001**
	Median NBR	100 - 60	-0.45	0.155	**0.022**
	Median NBR	100 - 80	-0.43	0.154	**0.028**
	Prop decr NBR	100 - 0	1.81	0.142	**< 0.001**
	Prop decr NBR	60 - 0	1.13	0.142	**< 0.001**
	Prop decr NBR	60 - 100	-0.68	0.132	**< 0.001**
	Prop decr NBR	80 - 0	1.32	0.139	**< 0.001**
	Prop decr NBR	80 - 100	-0.48	0.131	**0.002**
	Prop decr NBR	80 - 60	0.20775	0.131	0.433
	Prop decr NBR	Rain	0.03	0.008	**< 0.001**
	Area intact patch	100 - 0	-1.07	0.185	**< 0.001**
	Area intact patch	60 - 0	-0.70	0.185	**0.001**
	Area intact patch	60 - 100	0.37	0.173	0.150
	Area intact patch	80 - 0	-0.82	0.182	**< 0.001**
	Area intact patch	80 - 100	0.25	0.172	0.474
	Area intact patch	80 - 60	-0.12	0.172	0.902
	Area intact patch	Rain	-0.03	0.010	**0.006**
	Agg intact patch	100 - 0	-0.85	0.188	**< 0.001**
	Agg intact patch	60 - 0	-0.61	0.189	**0.009**
	Agg intact patch	60 - 100	0.24	0.177	0.537
	Agg intact patch	80 - 0	-0.62	0.187	**0.006**

	Agg intact patch	80 - 100	0.23	0.176	0.569
	Agg intact patch	80 - 60	-0.01	0.176	> 0.999
12	Median NBR	100 - 0	-0.48	0.037	**< 0.001**
	Median NBR	60 - 0	-0.30	0.037	**< 0.001**
	Median NBR	60 - 100	0.18	0.038	**< 0.001**
	Median NBR	80 - 0	-0.32	0.037	**< 0.001**
	Median NBR	80 - 100	0.15	0.038	**< 0.001**
	Median NBR	80 - 60	-0.02	0.038	0.929
	Prop decr NBR	100 - 0	0.64	0.039	**< 0.001**
	Prop decr NBR	60 - 0	0.41	0.039	**< 0.001**
	Prop decr NBR	60 - 100	-0.23	0.037	**< 0.001**
	Prop decr NBR	80 - 0	0.51	0.039	**< 0.001**
	Prop decr NBR	80 - 100	-0.13	0.037	**0.003**
	Prop decr NBR	80 - 60	0.10	0.037	**0.030**
	Area intact patch	100 - 0	-0.18	0.043	**< 0.001**
	Area intact patch	60 - 0	-0.08	0.043	0.294
	Area intact patch	60 - 100	0.11	0.042	**0.054**
	Area intact patch	80 - 0	-0.05	0.043	0.619
	Area intact patch	80 - 100	0.13	0.042	**0.010**
	Area intact patch	80 - 60	0.02	0.041	0.940
	Agg intact patch	100 - 0	-0.19945	0.042806	**< 0.001**
	Agg intact patch	60 - 0	-0.0711	0.043257	0.354
	Agg intact patch	60 - 100	0.128347	0.040947	**0.009**
	Agg intact patch	80 - 0	0.041312	0.042686	0.768
	Agg intact patch	80 - 100	0.240762	0.040876	**< 0.001**
	Agg intact patch	80 - 60	0.112415	0.040896	**0.031**

Table S17: Effect of different intensities of liana removal (60, 80, 100%) on the canopy across 1- and 12-months post treatment in terms of proportion of the canopy with decreased NBR ("Prop decr NBR"), intact patch size ("Area intact patch") and aggregation ("Agg intact patch"), using the 10% threshold for change in NBR. Table gives the coefficient for the difference between each intensity of liana removal and control, the difference between removal treatments, and, where significant, the influence of rain and liana load. 1-month coefficients are from linear mixed effects models, and 12-month coefficients are from generalized additive models. Response variables are normalized prior to running models. Row and column fixed effects are not presented but absorbed some variation.

Months post treatment	**Metric**	**Contrast / Fixed effect**	**Estimate**	**SE**	**P-value**
1	Prop decr NBR	100 - 0	1.87	0.152	< **0.001**
	Prop decr NBR	60 - 0	1.27	0.152	< **0.001**
	Prop decr NBR	60 - 100	-0.60	0.136	< **0.001**
	Prop decr NBR	80 - 0	1.46	0.149	< **0.001**
	Prop decr NBR	80 - 100	-0.41	0.135	**0.014**
	Prop decr NBR	80 - 60	0.19	0.135	0.509
	Prop decr NBR	Rain	0.03	0.008	< **0.001**
	Area intact patch	100 - 0	-1.15	0.187	< **0.001**
	Area intact patch	60 - 0	-1.03	0.186	< **0.001**
	Area intact patch	60 - 100	0.13	0.174	0.882
	Area intact patch	80 - 0	-1.08	0.184	< **0.001**
	Area intact patch	80 - 100	0.08	0.174	0.971
	Area intact patch	80 - 60	-0.05	0.173	0.991
	Area intact patch	Rain	-0.03	0.010	**0.003**
	Agg intact patch	100 - 0	-1.22	0.184	< **0.001**
	Agg intact patch	60 - 0	-0.92	0.184	< **0.001**
	Agg intact patch	60 - 100	0.30	0.171	0.299
	Agg intact patch	80 - 0	-0.96	0.181	< **0.001**
	Agg intact patch	80 - 100	0.23	0.171	0.444
	Agg intact patch	80 - 60	-0.05	0.170	0.993
	Agg intact patch	Rain	-0.02	0.010	**0.015**
12	Prop decr NBR	100 -0	0.59	0.045	< **0.001**
	Prop decr NBR	60 - 0	0.36	0.045	< **0.001**
	Prop decr NBR	60 - 100	-0.23	0.042	< **0.001**
	Prop decr NBR	80 - 0	0.41	0.045	< **0.001**
	Prop decr NBR	80 - 100	-0.19	0.041	< **0.001**
	Prop decr NBR	80 - 60	0.04	0.042	0.740
	Area intact patch	100 -0	-0.22	0.042	< **0.001**
	Area intact patch	60 - 0	-0.16	0.042	**0.001**
	Area intact patch	60 - 100	0.066	0.040	0.359

	Area intact patch	80 - 0	-0.12	0.042	**0.024**
	Area intact patch	80 - 100	0.10	0.040	**0.050**
	Area intact patch	80 - 60	0.04	0.040	0.788
	Area intact patch	Liana load	-0.45	0.211	**0.033**
	Agg intact patch	100 -0	-0.26	0.041	**< 0.001**
	Agg intact patch	60 - 0	-0.15	0.042	**0.002**
	Agg intact patch	60 - 100	0.11	0.039	**0.019**
	Agg intact patch	80 - 0	-0.09	0.041	0.154
	Agg intact patch	80 - 100	0.18	0.039	**< 0.001**
	Agg intact patch	80 - 60	0.06	0.039	0.393
	Agg intact patch	Liana load	-0.46	0.207	**0.026**

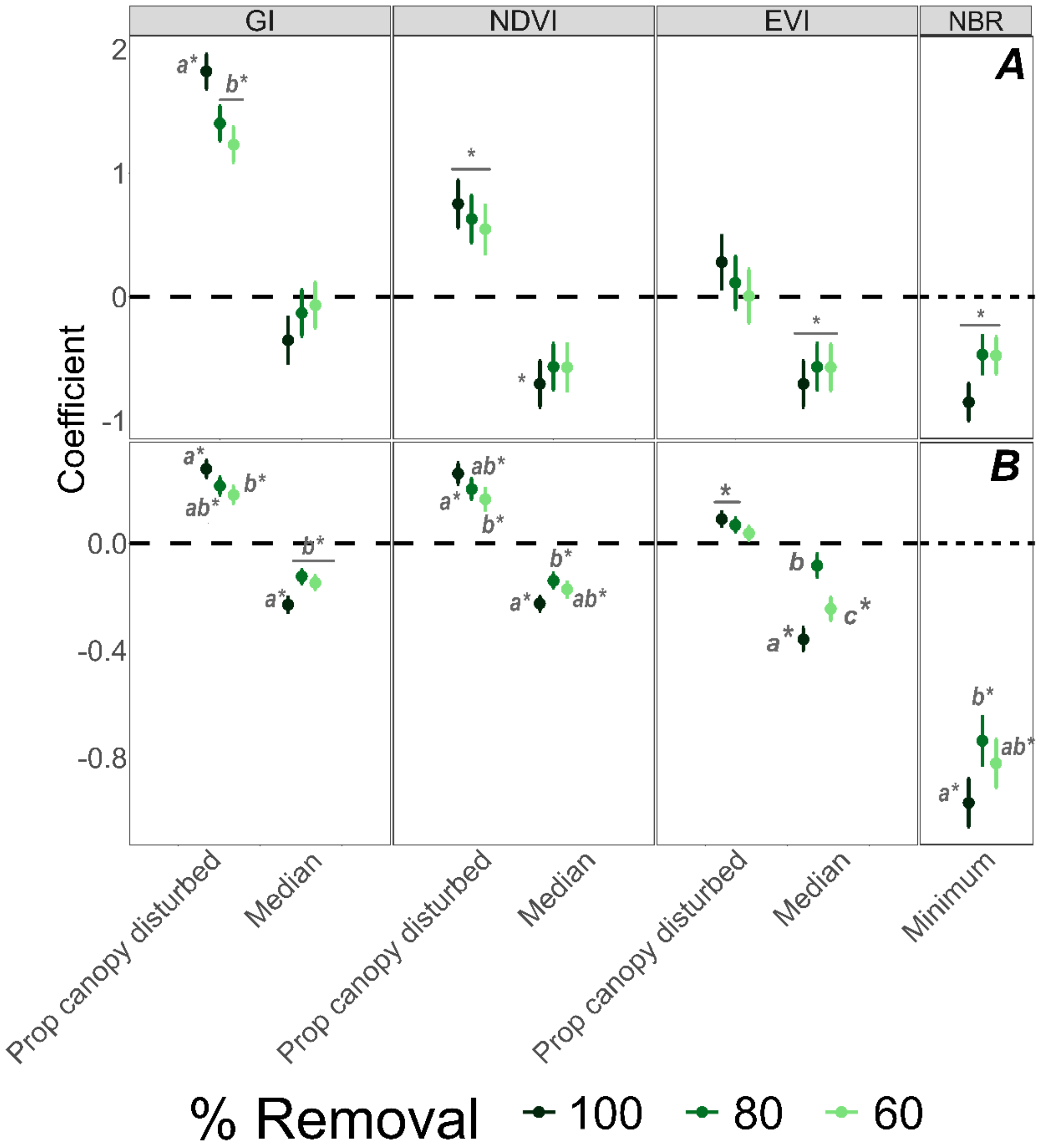


Figure S19: Effects of different intensities of liana removal (60, 80, and 100% removal) on median GI, NDVI, and EVI, the proportion of the canopy with decrease in GI, NDVI, and EVI, and minimum NBR. These metrics are detected from S2 images acquired during 1-month (A) and 12-months (B) post-treatment. Points show coefficients of treatment intensities from linear models in A, and from GAMs in (B); response variables are normalized prior to running models. The dotted line shows control, 0% removal; coefficients below the line indicate a decrease compared to control, and above the line indicate an increase compared to control. Different grey letters indicate a significant difference between percentage removal treatments, and “*” indicates removal treatments that are significantly different from control (zero). Error bars show standard error.

Table S18: Model summaries for the effect of different intensities of removal (60, 80, 100%) on the canopy across 1- and 12-months post treatment in terms of minimum NBR, median GI, NDVI, and EVI, and the proportion of the canopy with decreased GI, NDVI, and EVI ("Prop decr"). 1-month coefficients are from linear mixed effects models, and 12-month coefficients are from generalized additive models. Table gives the difference between each intensity of liana removal and control, the difference between removal treatments, and the influence of rainfall, when significant. Response variables are normalized prior to running models. Row and column fixed effects are not presented.

Months post treatment	Metric	Contrast / Fixed effect	Estimate	SE	P-value
1	Minimum NBR	100 - 0	-0.85	0.150	< **0.001**
	Minimum NBR	60 - 0	-0.47	0.150	**0.009**
	Minimum NBR	60 - 100	0.38	0.206	0.259
	Minimum NBR	80 - 0	-0.47	0.151	**0.012**
	Minimum NBR	80 - 100	0.38	0.207	0.249
	Minimum NBR	80 – 60	0.01	0.207	> 0.999
	Minimum NBR	Rain	0.06	0.006	< **0.001**
	Median GI	60 - 0	-0.07	0.185	0.984
	Median GI	80 - 0	-0.13	0.186	0.898
	Median GI	80 - 60	-0.06	0.176	0.984
	Median GI	100 - 0	-0.35	0.184	0.231
	Median GI	100 - 60	-0.28	0.176	0.372
	Median GI	100 - 80	-0.22	0.175	0.588
	Median NDVI	60 - 0	-0.14	0.179	0.861
	Median NDVI	80 - 0	-0.18	0.179	0.747
	Median NDVI	80 - 60	-0.04	0.170	0.995
	Median NDVI	100 - 0	-0.48	0.178	**0.039**
	Median NDVI	100 - 60	-0.34	0.169	0.193
	Median NDVI	100 - 80	-0.30	0.169	0.291
	Median EVI	60 - 0	-0.57	0.186	**0.014**
	Median EVI	80 - 0	-0.56	0.187	**0.015**
	Median EVI	80 - 60	< 0.01	0.177	0.100
	Median EVI	100 - 0	-0.70	0.185	**0.001**
	Median EVI	100 - 60	-0.13	0.177	0.873
	Median EVI	100 - 80	-0.14	0.176	0.861

	Prop decr GI	100 - 0	1.82	0.141	**<0.001**
	Prop decr GI	60 - 0	1.23	0.141	**<0.001**
	Prop decr GI	60 - 100	-0.59	0.131	**<0.001**
	Prop decr GI	80 - 0	1.40	0.139	**<0.001**
	Prop decr GI	80 - 100	-0.42	0.130	**0.008**
	Prop decr GI	80 - 60	0.17	0.130	0.554
	Prop decr GI	Rain	0.03	0.008	**<0.001**
	Prop decr NDVI	100 - 0	0.75	0.193	**0.001**
	Prop decr NDVI	60 - 0	0.55	0.194	**0.027**
	Prop decr NDVI	60 - 100	-0.20	0.173	0.640
	Prop decr NDVI	80 - 0	0.63	0.191	**0.007**
	Prop decr NDVI	80 - 100	-0.12	0.172	0.895
	Prop decr NDVI	80 - 60	0.08	0.172	0.963
	Prop decr EVI	100 - 0	0.28	0.211	0.543
	Prop decr EVI	60 - 0	0.01	0.216	0.100
	Prop decr EVI	60 - 100	-0.27	0.201	0.522
	Prop decr EVI	80 - 0	0.11	0.212	0.959
	Prop decr EVI	80 - 100	-0.17	0.196	0.830
	Prop decr EVI	80 - 60	0.11	0.202	0.950
	Prop decr EVI	Rain	-0.04	0.012	**0.003**
12	Minimum NBR	100 - 0	-0.97	0.090	< **0.001**
	Minimum NBR	60 - 0	-0.82	0.091	< **0.001**
	Minimum NBR	60 - 100	0.15	0.063	0.0932
	Minimum NBR	80 - 0	-0.74	0.089	< **0.001**
	Minimum NBR	80 - 100	0.23	0.064	**0.0019**
	Minimum NBR	80 - 60	0.08	0.062	0.5223
	Median GI	100 - 0	-0.23	0.026	**<0.001**
	Median GI	60 - 0	-0.15	0.026	**<0.001**
	Median GI	60 - 100	0.08	0.026	**0.011**
	Median GI	80 - 0	-0.12	0.026	**<0.001**
	Median GI	80 - 100	0.11	0.026	**<0.001**
	Median GI	80 - 60	0.02	0.026	0.809
	Median GI	Rain	0.38	0.170	**0.0248**
	Median NDVI	100 - 0	-0.22	0.027	**<0.001**
	Median NDVI	60 - 0	-0.17	0.027	**<0.001**

	Median NDVI	60 - 100	0.05	0.028	0.224
	Median NDVI	80 - 0	-0.14	0.027	**<0.001**
	Median NDVI	80 - 100	0.08	0.028	**0.012**
	Median NDVI	80 - 60	0.03	0.028	0.656
	Median NDVI	Rain	0.44	0.178	**0.014**
	Median EVI	100 - 0	-0.36	0.043	**<0.001**
	Median EVI	60 - 0	-0.24	0.043	**<0.001**
	Median EVI	60 - 100	0.11	0.044	**0.045**
	Median EVI	80 - 0	-0.08	0.043	0.209
	Median EVI	80 - 100	0.27	0.044	**<0.001**
	Median EVI	80 - 60	0.16	0.043	**0.001**
	Median EVI	Rain	0.90	0.280	**0.001**
	Prop decr GI	100 - 0	0.28	0.032	**<0.001**
	Prop decr GI	60 - 0	0.18	0.033	**<0.001**
	Prop decr GI	60 - 100	-0.10	0.031	**0.010**
	Prop decr GI	80 - 0	0.21	0.032	**<0.001**
	Prop decr GI	80 - 100	-0.06	0.031	0.171
	Prop decr GI	80 - 60	0.03	0.031	0.700
	Prop decr NDVI	100 - 0	0.26	0.038	**<0.001**
	Prop decr NDVI	60 - 0	0.16	0.039	**<0.001**
	Prop decr NDVI	60 - 100	-0.10	0.036	**0.039**
	Prop decr NDVI	80 - 0	0.20	0.038	**<0.001**
	Prop decr NDVI	80 - 100	-0.06	0.036	0.363
	Prop decr NDVI	80 - 60	0.04	0.036	0.723
	Prop decr EVI	100 - 0	0.09	0.025	**0.002**
	Prop decr EVI	60 - 0	0.04	0.025	0.439
	Prop decr EVI	60 - 100	-0.05	0.024	0.135
	Prop decr EVI	80 - 0	0.07	0.025	**0.034**
	Prop decr EVI	80 - 100	-0.02	0.024	0.801
	Prop decr EVI	80 - 60	0.03	0.024	0.595

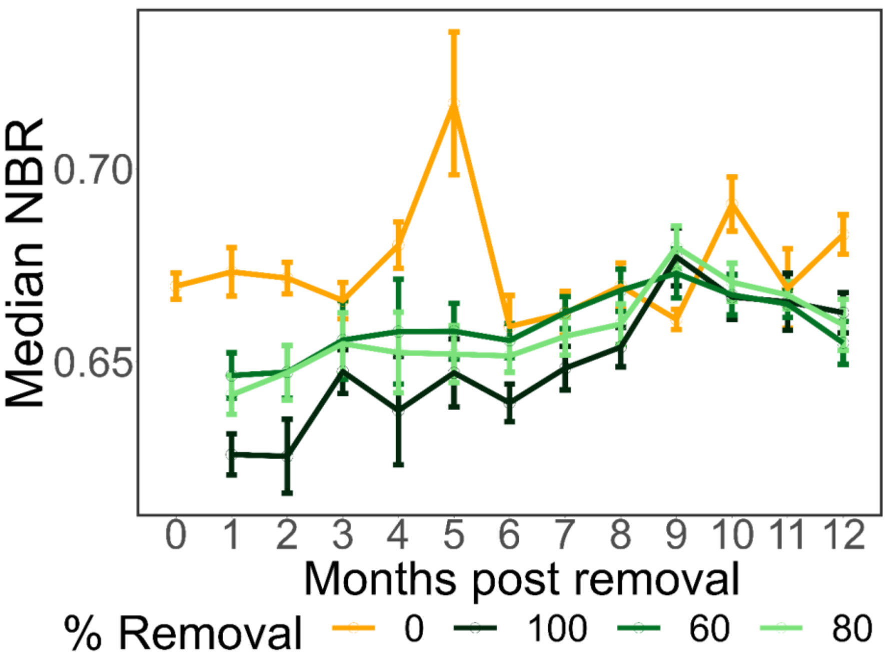


Figure S20: Median NBR across 12-months post treatment for all removal intensities and control (0% removal). Error bars indicate standard error.

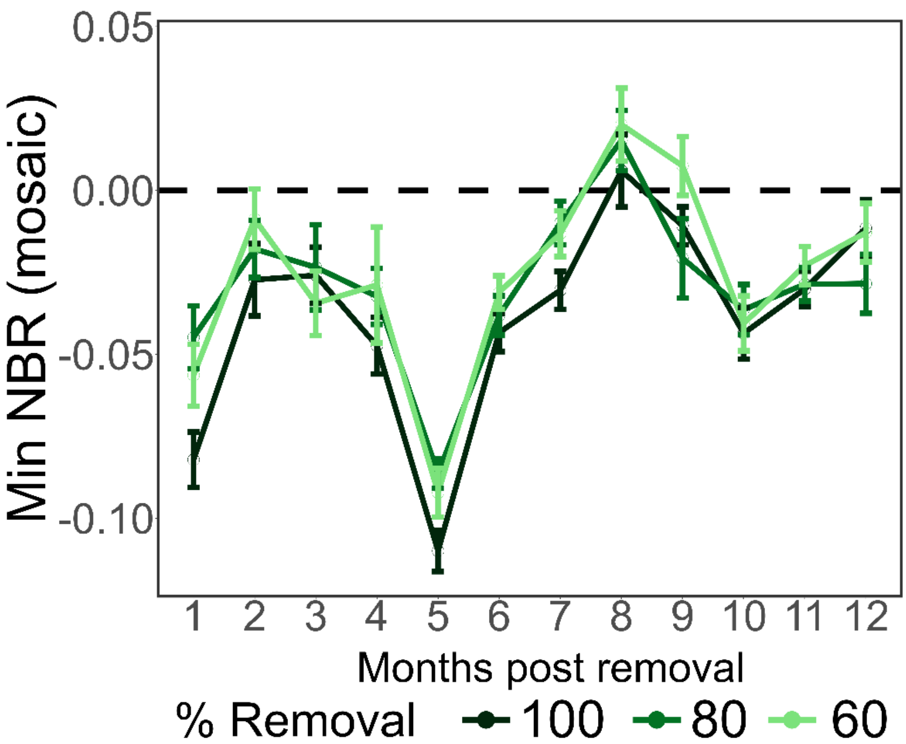


Figure S21: Effect of liana removal on minimum NBR over 12-months post-treatment. Green lines represent the average minimum NBR for each treatment and month post-treatment, and relative to the mean control value at each month (dotted black line at zero). The dotted lines indicate mean values for control blocks, normalised to zero. Values above the line indicate increases relative to control, and values below the line indicate decreases relative to control. Error bars show standard error.

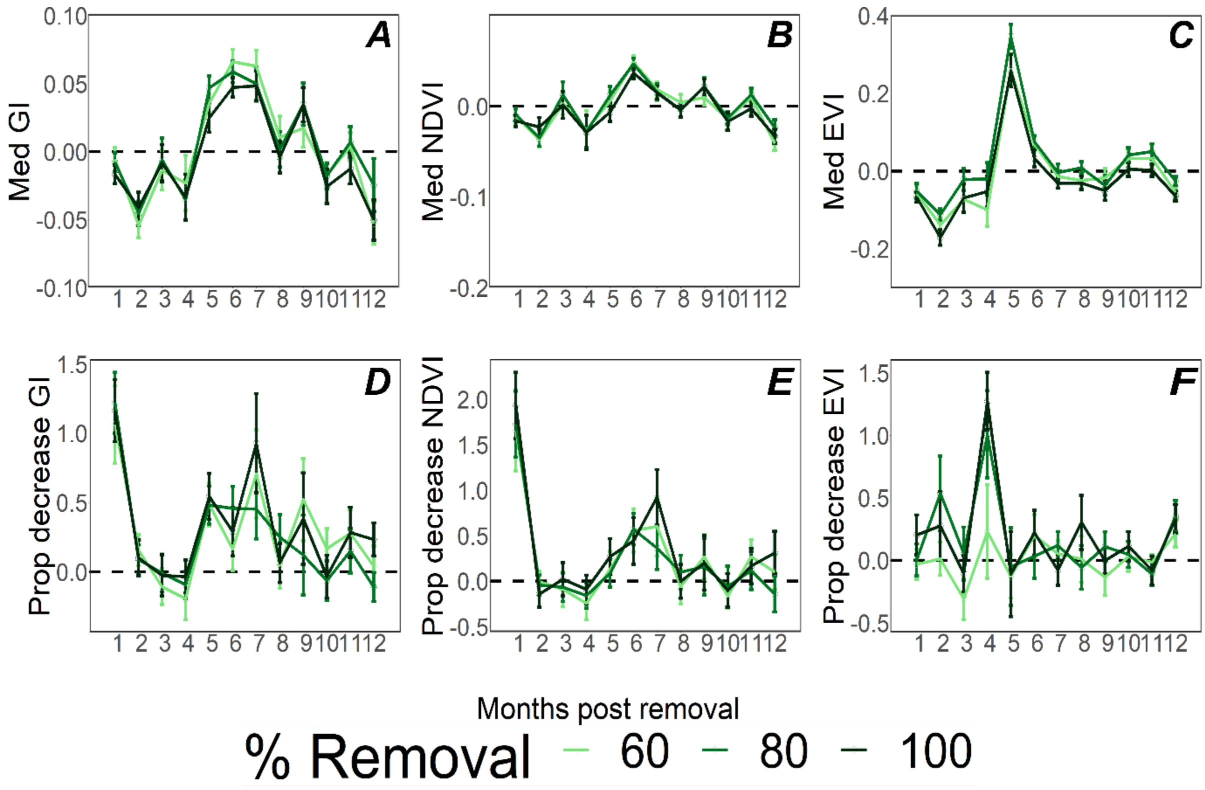
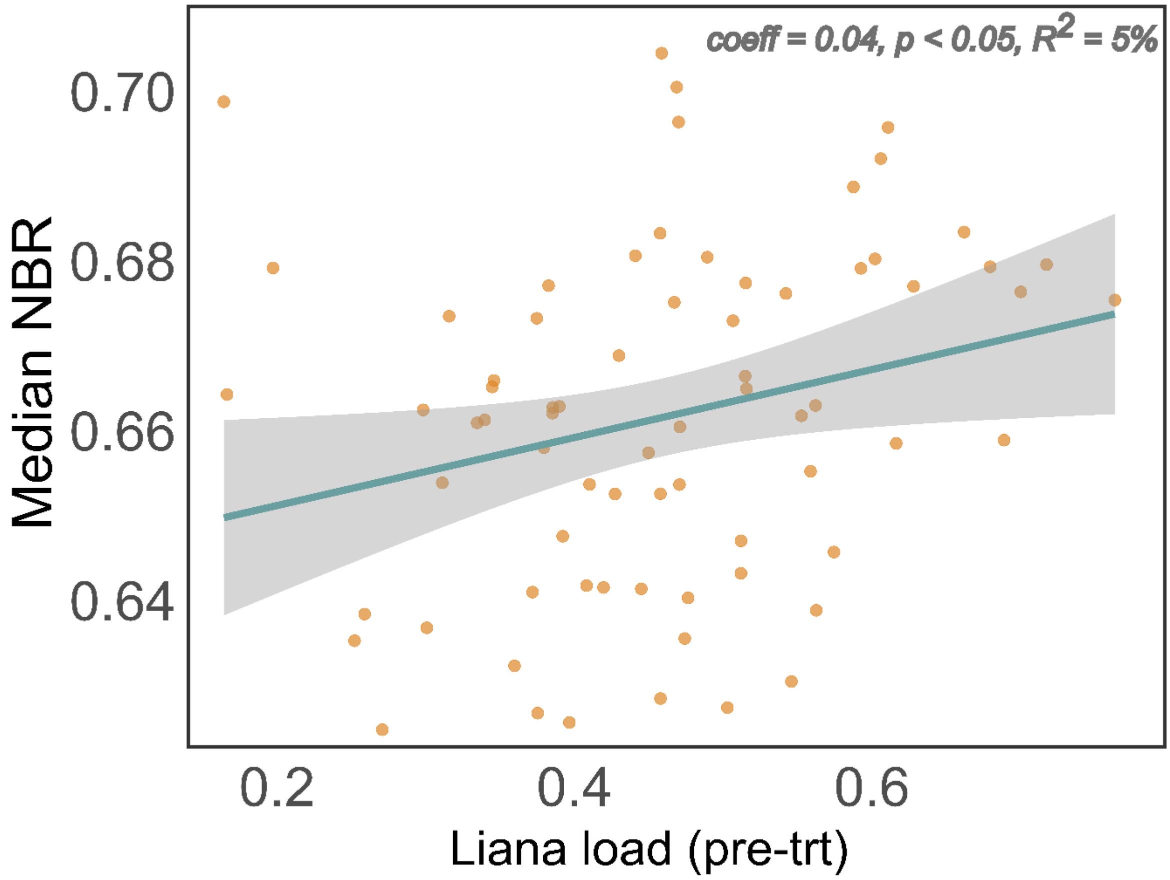


Figure S22: Impact of climber removal on median GI (A), NDVI (B), and EVI (C) values and on the proportion of the canopy with decreased GI (D), NDVI (E), and EVI (F) over 12 months post-treatment. Green lines represent the average of each metric for each treatment and month post-treatment, and relative to the mean control value at each month (dotted black line at zero). The dotted lines indicate mean values for control blocks, normalised to zero. Values above the line indicate increases relative to control, and values below the line indicate decreases relative to control. Error bars show standard error.



2103

2104 **Figure S23:** Correlation between pre-treatment liana load and median NBR. Coefficient, p-value and
2105 r2 are calculated using linear regression.

SUPPLEMENTARY INFORMATION Chapter 4

Table S19: Effect of covariates on minimum NBR and the proportion of the canopy with decreased NBR in generalized additive models, including supplementary models based on subsets of compartments or data. Effect given in terms of coefficient estimates for fixed effects, or EDF for smooth terms. "Months treated" indicates if model includes subset of treatment compartments that were treated within the smallest time-period. "Images during treatment" indicate if model includes Sentinel-2 images that were collecting during the months in which compartments were treated.

NBR metric	***Covariate***	***Months treated[1]***	***Images during treatment[2]***	***Estimate***	***SE***	***EDF***	***F***	***P-value***
Minimum NBR	Intercept	all	Y	0.179	0.016			p<0.001
	Trt (Ref:Trt)	all	Y	0.075	0.030			p < 0.05
	Daily precip	all	Y	0.054	0.004			p<0.001
	Compart area	all	Y	-0.198	0.015			p<0.001
	2017 : 2019	all	Y	-0.083	0.011			p<0.001
	2017 : 2020	all	Y	-0.322	0.011			p<0.001
	2017 : 2021	all	Y	-0.315	0.012			p<0.001
	2020 - 2019	all	Y	-0.239	0.009			p<0.001
	2021 - 2019	all	Y	-0.231	0.009			p<0.001
	2021 - 2020	all	Y	0.007	0.010			p = 0.9
	Month	all	Y			10.947	112.594	p<0.001
	Compartment ID	all	Y			199.768	14.468	p<0.001
	Forest reserve	all	Y			<0.001	<0.001	p = 0.6
	Intercept	fewest	Y	0.153	0.018			p<0.001

	Trt (Ref:Trt)	fewest	Y	0.103	0.035			p<0.01
	Daily precip	fewest	Y	0.049	0.004			p<0.001
	Compart area	fewest	Y	-0.196	0.016			p<0.001
	2017 : 2019	fewest	Y	-0.030	0.013			p<0.05
	2017 : 2020	fewest	Y	-0.282	0.013			p<0.001
	2017 : 2021	fewest	Y	-0.273	0.013			p<0.001
	2020 - 2019	fewest	Y	-0.253	0.009			p<0.001
	2021 - 2019	fewest	Y	-0.243	0.010			p<0.001
	2021 - 2020	fewest	Y	0.009	0.010			p = 0.8
	Month	fewest	Y			10.938	100.006	p<0.001
	Compartment ID	fewest	Y			193.072	13.744	p<0.001
	Forest reserve	fewest	Y			<0.001	<0.001	p = 0.6
	Intercept	all	N	0.208	0.076			p<0.01
	Trt (Ref:Trt)	all	N	0.058	0.113			p = 0.6
	Daily precip	all	N	0.054	0.004			p<0.001
	Compart area	all	N	-0.197	0.017			p<0.001
	2017 : 2019	all	N	-0.083	0.011			p<0.001
	2017 : 2020	all	N	-0.322	0.011			p<0.001
	2017 : 2021	all	N	-0.314	0.012			p<0.001
	2020 - 2019	all	N	-0.239	0.009			p<0.001
	2021 - 2019	all	N	-0.231	0.009			p<0.001
	2021 - 2020	all	N	0.008	0.010			p = 0.9
	Month	all	N			10.987	112.177	p<0.001
	Compartment ID	all	N			197.894	15.486	p<0.001
	Forest reserve*Trt(R)	all	N			1.845	485.242	p=0.1
	Forest reserve*Trt(T)	all	N			1.676	12.640	p=0.1

Proportion of canopy with decrease in NBR	Intercept	all	Y	-0.102	0.045			p<0.05
	Trt (Ref:Trt)	all	Y	-0.148	0.341			p = 0.7
	Daily precip	all	Y	0.010	0.002			p<0.001
	Compart area	all	Y	<0.001	<0.001			p = 0.1
	2017 : 2019	all	Y	0.425	0.036			p<0.001
	2017 : 2020	all	Y	-0.050	0.036			p = 0.1
	2017 : 2021	all	Y	-0.262	0.037			p<0.001
	2020 : 2019	all	Y	-0.474	0.031			p<0.001
	2021 : 2019	all	Y	-0.687	0.037			p<0.001
	2021 : 2020	all	Y	-0.213	0.036			p<0.001
	Trt*Daily precip	all	Y	-0.037	0.011			p<0.001
	Month	all	Y			10.564	72.431	p<0.001
	Compartment ID	all	Y			123.343	1.611	p<0.001
	Forest reserve*Trt(R)	all	Y			<0.001	<0.001	p = 0.5
	Forest reserve*Trt(T)	all	Y			1.878	25.569	p<0.001
	Intercept	fewest	Y	0.049	0.056			p = 0.4
	Trt (Ref:Trt)	fewest	Y	-0.280	0.456			p = 0.5
	Daily precip	fewest	Y	0.011	0.003			p<0.001
	Compart area	fewest	Y	<0.001	<0.001			p = 0.2
	2017 : 2019	fewest	Y	0.310	0.047			p<0.001
	2017 : 2020	fewest	Y	-0.173	0.045			p<0.001
	2017 : 2021	fewest	Y	-0.436	0.047			p<0.001
	Trt*Daily precip	fewest	Y	-0.030	0.013			p<0.05
	2020 : 2019	fewest	Y	-0.483	0.034			p<0.001
	2021 : 2019	fewest	Y	-0.746	0.039			p<0.001
	2021 : 2020	fewest	Y	-0.263	0.037			p<0.001
	Month	fewest	Y			10.308	58.818	p<0.001
	Compartment ID	fewest	Y			112.240	1.385	p<0.001

	Forest reserve*Trt(R)	fewest	Y			0.339	0.720	p = 0.2
	Forest reserve*Trt(T)	fewest	Y			1.885	10.701	p<0.01
	Intercept	all	N	-0.093	0.045			p<0.05
	Trt (Ref:Trt)	all	N	-0.170	0.340			p = 0.6
	Daily precip	all	N	0.010	0.002			p<0.001
	Compart area	all	N	<0.001	<0.001			p = 0.1
	2017 : 2019	all	N	0.416	0.036			p<0.001
	2017 : 2020	all	N	-0.055	0.036			p = 0.1
	2017 : 2021	all	N	-0.272	0.037			p<0.001
	2020 : 2019	all	N	-0.471	0.032			p<0.001
	2021 : 2019	all	N	-0.688	0.037			p<0.001
	2021 : 2020	all	N	-0.217	0.036			p<0.001
	Trt*Daily precip	all	N	-0.051	0.013			p<0.001
	Month	all	N			10.445	72.090	p<0.001
	Compartment ID	all	N			120.516	1.590	p<0.001
	Forest reserve*Trt(R)	all	N			<0.001	<0.001	p = 0.5
	Forest reserve*Trt(T)	all	N			1.848	20.870	p<0.001

[1] Indicates whether model includes all compartments in which > 90% of the area was treated within 12 months (“all”), or whether model only included compartments that were treated within the fewest month in each treatment year (“fewest”: compartments treated within 3 months for those treated in 2017, 2019, and 2020, compartments treated within 6 months for those treated in 2021).

[2] Indicates whether model includes all Sentinel-2 images acquired after the first day of the annual quarter in which each compartment was first treated (“Y”), or only includes Sentinel-2 images acquired after the end of the last annual quarter in which treatment was completed in each compartment (“N”).

Table S20: Effect of covariates on minimum NBR and the proportion of canopy with decreased NBR in sub-compartment linear mixed effect models. There are four models per treatment year: (a) does not include sub-compartment as a random effect, (b) includes sub-compartment as random effect, (c) includes sub-compartment random effect and slope and distance from main roads as fixed effects, and (d) includes sub-compartment random effect and slope and distance from active roads as fixed effects. Main effect coefficients are shown first, and coefficients for interaction with treatment “Trt (R:T) *” are shown second. The model covariates *not* underlined and in **bold** are influenced by pseudo-replication and should be interpreted with caution.

NBR metric	***Model***	***Covariate***	***Estimate***	***SE***	***P-value***
Minimum NBR	a	Intercept	0.638	0.047	p<0.001
	a	Trt (Ref:Trt)	0.035	0.006	p<0.001
	a	Daily precip	0.005	0.001	p<0.001
	a	Forest reserve (BukP:MFR)	-0.197	0.040	p<0.001
	a	Forest reserve (BukP:USFR)	-0.095	0.043	p<0.05
	a	Trt (R:T) * Daily precip	0.022	0.002	p<0.001
	a	Trt (R:T) * Trt year (2017:2019)	-0.057	0.002	p<0.001
	a	Trt (R:T) * Trt year (2017:2020)	-0.990	0.002	p<0.001
	a	Trt (R:T) * Trt year (2017:2021)	-0.930	0.002	p<0.001
	a	Trt (R:T) * Trt year (2020:2019)	-0.933	0.002	p<0.001
	a	Trt (R:T) * Trt year (2021:2019)	-0.873	0.002	p<0.001
	a	Trt (R:T) * Trt year (2021:2020)	0.060	0.002	p<0.001
	a	Trt (R:T) * Forest res (BukP:MFR)	0.035	0.009	p<0.001
	a	Trt (R:T) * Forest res (BukP:USFR)	-0.054	0.010	p<0.001
	b	Intercept	0.651	0.047	p<0.001
	b	Trt (Ref:Trt)	0.037	0.005	p<0.001
	b	Daily precip	0.006	0.001	p<0.001

	b	Forest reserve (BukP:MFR)	-0.194	0.041	p<0.001
	b	Forest reserve (BukP:USFR)	-0.090	0.044	p<0.05
	b	Trt (R:T) * Daily precip	0.022	0.002	p<0.001
	b	Trt (R:T) * Trt year (2017:2019)	-0.058	0.002	p<0.001
	b	Trt (R:T) * Trt year (2017:2020)	-0.998	0.002	p<0.001
	b	Trt (R:T) * Trt year (2017:2021)	-0.938	0.002	p<0.001
	b	Trt (R:T) * Trt year (2020:2019)	-0.940	0.002	p<0.001
	b	Trt (R:T) * Trt year (2021:2019)	-0.880	0.002	p<0.001
	b	Trt (R:T) * Trt year (2021:2020)	0.060	0.002	p<0.001
	b	Trt (R:T) * Forest res (BukP:MFR)	0.032	0.008	p<0.001
	b	Trt (R:T) * Forest res (BukP:USFR)	-0.053	0.009	p<0.001
	c	Intercept	0.635	0.047	p<0.001
	c	Trt (Ref:Trt)	-0.079	0.011	p<0.001
	c	Daily precip	0.006	0.001	p<0.001
	c	Forest reserve (BukP:MFR)	-0.189	0.042	p<0.001
	c	Forest reserve (BukP:USFR)	-0.067	0.045	p = 0.1
	c	**Slope**	-0.028	0.005	p<0.001
	c	**Dist main rd**	-0.006	0.010	p = 0.6
	c	Trt (R:T) * Daily precip	0.022	0.002	p<0.001
	c	Trt (R:T) * Trt year (2017:2019)	-0.058	0.002	p<0.001
	c	Trt (R:T) * Trt year (2017:2020)	-0.998	0.002	p<0.001
	c	Trt (R:T) * Trt year (2017:2021)	-0.938	0.002	p<0.001
	c	Trt (R:T) * Trt year (2020:2019)	-0.961	0.005	p<0.001
	c	Trt (R:T) * Trt year (2021:2019)	-0.862	0.008	p<0.001
	c	Trt (R:T) * Trt year (2021:2020)	0.099	0.009	p<0.001
	c	**Trt (R:T) * Slope**	0.038	0.004	p<0.001
	c	**Trt (R:T) * Dist main rd**	-0.036	0.005	p<0.001
	d	Intercept	0.515	0.061	p<0.001

	d	Trt (Ref:Trt)	0.017	0.010	p = 0.1
	d	Daily precip	0.006	0.001	p<0.001
	d	Forest reserve (BukP:MFR)	0.052	0.088	p = 0.6
	d	Forest reserve (BukP:USFR)	-0.041	0.045	p = 0.4
	d	**Slope**	-0.027	0.005	p<0.001
	d	**Dist second rd**	-0.104	0.033	p<0.05
	d	Trt (R:T) * Daily precip	0.022	0.002	p<0.001
	d	Trt (R:T) * Trt year (2017:2019)	-0.057	0.002	p<0.001
	d	Trt (R:T) * Trt year (2017:2020)	-0.998	0.002	p<0.001
	d	Trt (R:T) * Trt year (2017:2021)	-0.938	0.002	p<0.001
	d	Trt (R:T) * Trt year (2020:2019)	-0.973	0.007	p<0.001
	d	Trt (R:T) * Trt year (2021:2019)	-0.894	0.010	p<0.001
	d	Trt (R:T) * Trt year (2021:2020)	0.080	0.008	p<0.001
	d	**Trt (R:T) * Slope**	0.037	0.004	p<0.001
	d	**Trt (R:T) * Dist second rd**	0.031	0.006	p<0.001
Proportion of canopy with decreased NBR	a	Intercept	-0.191	0.074	p<0.05
	a	Trt (Ref:Trt)	-0.320	0.072	p<0.001
	a	Daily precip	0.010	0.001	p<0.001
	a	Forest reserve (BukP:MFR)	0.148	0.055	p<0.01
	a	Forest reserve (BukP:USFR)	0.062	0.059	p<0.05
	a	Trt (R:T) * Daily precip	-0.100	0.004	p<0.001
	a	Trt (R:T) * Trt year (2017:2019)	0.232	0.004	p<0.001
	a	Trt (R:T) * Trt year (2017:2020)	0.066	0.004	p<0.001
	a	Trt (R:T) * Trt year (2017:2021)	-0.039	0.004	p<0.001
	a	Trt (R:T) * Trt year (2020:2019)	-0.166	0.003	p<0.001
	a	Trt (R:T) * Trt year (2021:2019)	-0.270	0.003	p<0.001
	a	Trt (R:T) * Trt year (2021:2020)	-0.104	0.003	p<0.001
	a	Trt (R:T) * Forest res (Bukp:MFR)	0.227	0.083	p<0.01

	a	Trt (R:T) * Forest res (Bukp:USFR)	0.656	0.087	p<0.001
	b	Intercept	-0.176	0.074	p<0.05
	b	Trt (Ref:Trt)	-0.343	0.071	p<0.001
	b	Daily precip	0.009	0.001	p<0.001
	b	Forest reserve (BukP:MFR)	0.136	0.054	p<0.05
	b	Forest reserve (BukP:USFR)	0.056	0.058	p = 0.3
	b	Trt (R:T) * Daily precip	-0.099	0.003	p<0.001
	b	Trt (R:T) * Trt year (2017:2019)	0.232	0.004	p<0.001
	b	Trt (R:T) * Trt year (2017:2020)	0.065	0.004	p<0.001
	b	Trt (R:T) * Trt year (2017:2021)	-0.041	0.004	p<0.001
	b	Trt (R:T) * Trt year (2020:2019)	-0.168	0.003	p<0.001
	b	Trt (R:T) * Trt year (2021:2019)	-0.273	0.003	p<0.001
	b	Trt (R:T) * Trt year (2021:2020)	-0.105	0.003	p<0.001
	b	Trt (R:T) * Forest res (Bukp:MFR)	0.255	0.082	p<0.01
	b	Trt (R:T) * Forest res (Bukp:USFR)	0.679	0.086	p<0.001
	c	Intercept	-0.176	0.079	p<0.05
	c	Trt (Ref:Trt)	-0.391	0.082	p<0.001
	c	Daily precip	0.009	0.001	p<0.001
	c	Forest reserve (BukP:MFR)	0.138	0.062	p < 0.05
	c	Forest reserve (BukP:USFR)	0.060	0.067	p = 0.4
	c	**Slope**	0.009	0.004	p<0.05
	c	**Dist main rd**	-0.037	0.009	p<0.001
	c	Trt (R:T) * Daily precip	-0.099	0.003	p<0.001
	c	Trt (R:T) * Trt year (2017:2019)	0.232	0.004	p<0.001
	c	Trt (R:T) * Trt year (2017:2020)	0.065	0.004	p<0.001
	c	Trt (R:T) * Trt year (2017:2021)	-0.040	0.004	p<0.001
	c	Trt (R:T) * Trt year (2020:2019)	-0.168	0.003	p<0.001
	c	Trt (R:T) * Trt year (2021:2019)	-0.273	0.003	p<0.001

	c	Trt (R:T) * Trt year (2021:2020)	-0.105	0.003	p<0.001
	c	Trt (R:T) * Forest res (Bukp:MFR)	0.061	0.097	p = 0.5
	c	Trt (R:T) * Forest res (Bukp:USFR)	0.430	0.103	p<0.001
	c	**Trt (R:T) * Slope**	-0.024	0.007	p<0.001
	c	**Trt (R:T) * Dist main rd**	-0.143	0.022	p<0.001
	d	Intercept	-0.074	0.086	p = 0.4
	d	Trt (Ref:Trt)	-0.599	0.119	p<0.001
	d	Daily precip	0.009	0.001	p<0.001
	d	Forest reserve (BukP:MFR)	-0.039	0.092	p = 0.7
	d	Forest reserve (BukP:USFR)	0.008	0.062	p = 0.9
	d	**Slope**	0.009	0.004	p<0.05
	d	**Dist second rd**	0.075	0.032	p<0.05
	d	Trt (R:T) * Daily precip	-0.099	0.003	p<0.001
	d	Trt (R:T) * Trt year (2017:2019)	0.232	0.004	p<0.001
	d	Trt (R:T) * Trt year (2017:2020)	0.065	0.004	p<0.001
	d	Trt (R:T) * Trt year (2017:2021)	-0.041	0.004	p<0.001
	d	Trt (R:T) * Trt year (2020:2019)	-0.168	0.003	p<0.001
	d	Trt (R:T) * Trt year (2021:2019)	-0.273	0.003	p<0.001
	d	Trt (R:T) * Trt year (2021:2020)	-0.105	0.003	p<0.001
	d	Trt (R:T) * Forest res (Bukp:MFR)	0.668	0.181	p<0.001
	d	Trt (R:T) * Forest res (Bukp:USFR)	0.730	0.090	p<0.001
	d	**Trt (R:T) * Slope**	-0.030	0.007	p<0.001
	d	**Trt (R:T) * Dist second rd**	-0.176	0.069	p = 0.1

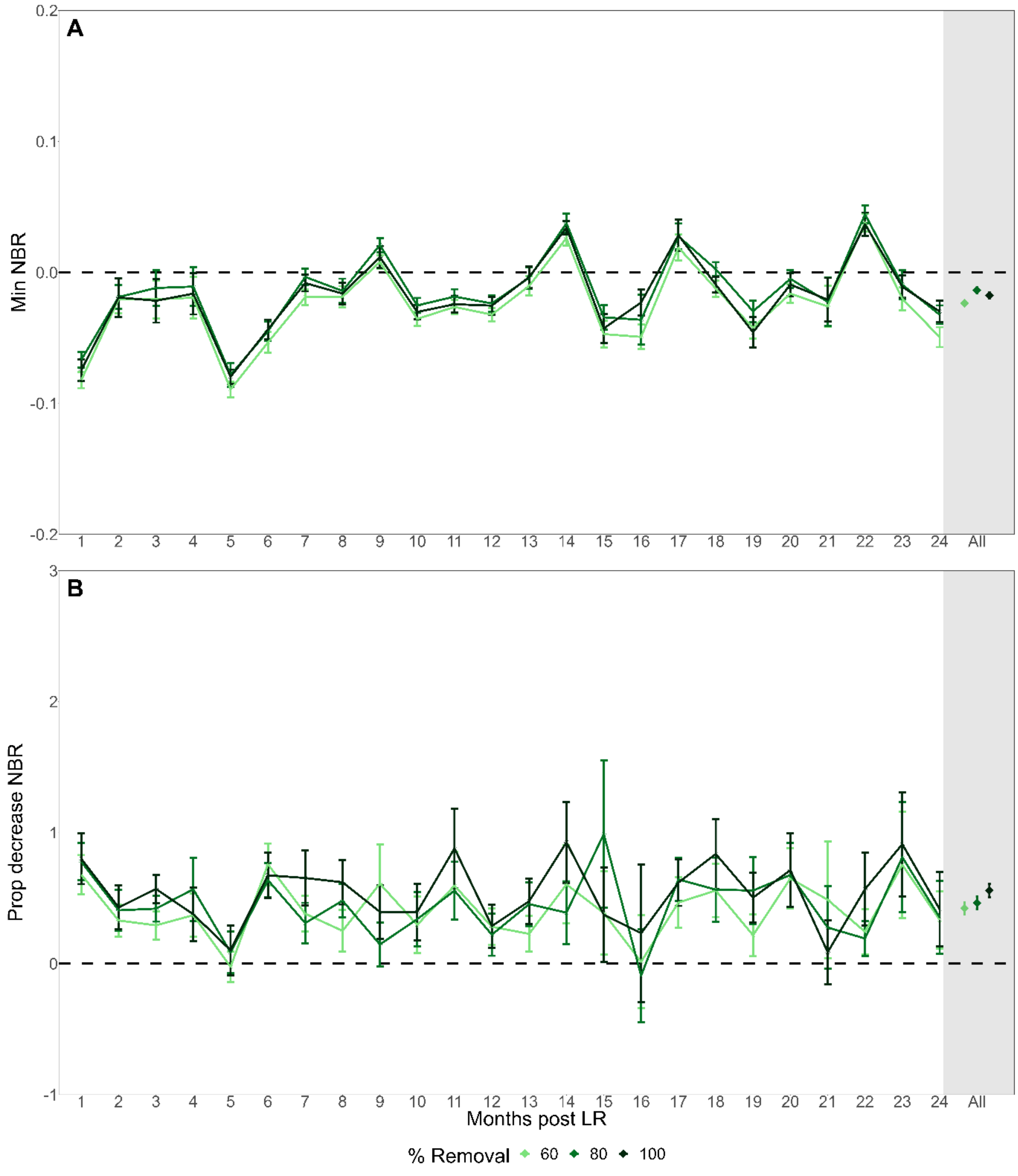


Figure S24: Effect of different intensities of experimental liana removal on minimum NBR (A) and the proportion of the canopy with decreased NBR (B) over 24-months post-treatment. This figure is based on additional Sentinel-2 data acquired for the liana removal experiment in Finlayson and Hethcoat et al (2022). Lines represent predicted values from GAMs, averaged for each removal intensity treatment and month post-treatment, and relative to the mean value for untreated blocks at each month. The dotted lines indicate mean values for untreated blocks, normalised to zero. Values above the line indicate increases in NBR metrics relative to control, and values below the line indicate decreases relative to control. The three points to the far right of each panel are the average over the whole 24 months. Error bars show standard error.

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