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Post-fire forest restoration in the humid tropics: A synthesis of available strategies and knowledge gaps for effective restoration

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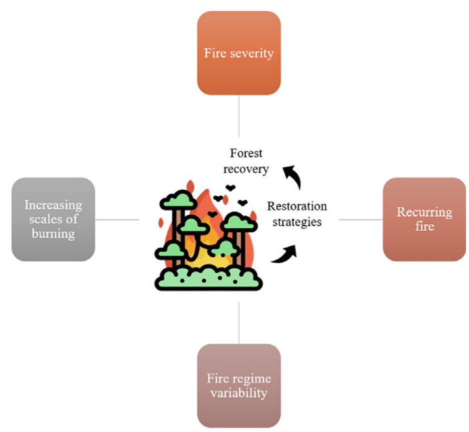
HIGHLIGHTS

- Humid tropical forests are often ill-adapted to fire.
- Post-fire restoration strategies depend strongly on context.
- Restoration practice should account for fire severity.
- Fire poses a recurring and intensifying threat throughout the recovery process.
- Landscape approaches are crucial to address the spatiotemporal dynamics of fire.

GRAPHICAL ABSTRACT



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ABSTRACT

Humid tropical forests are increasingly exposed to devastating wildfires. Major efforts are needed to prevent fire-related tipping points and to enable the effective recovery of fire-affected areas. Here, we provide a synthesis of the most common forest restoration strategies, thereby focusing on post-fire forest dynamics in the humid tropics. A variety of restoration strategies can be adopted in restoring humid tropical forests, including natural regeneration, assisted natural regeneration (i.e. fire breaks, weed control, erosion control, topsoil replacement, peatland rewetting), enrichment planting (i.e. planting nursery-raised seedlings, direct seeding) and commercial restoration (i.e. plantation forests, agroforestry). Our analysis shows that while natural regeneration can be effective under favourable ecological conditions, humid tropical forests are often ill-adapted to fire, and therefore less likely to recover unassisted after a wildfire event. Active restoration practices may be more effective, but can be costly and challenging to implement. We also identify gaps in knowledge needed for effective restoration of humid tropical forests after fire, hereby taking into account the ecosystems and socio-economic conditions in which these fires occur. We suggest to incorporate fire severity in future studies, to better understand and predict post-fire ecosystem responses. In addition, as fire poses a recurring and intensifying threat throughout the recovery process, more emphasis should be placed on post-restoration management and the prevention of fire throughout the different phases of the restoration process. Furthermore, as tropical wildfires are increasing in scale, establishing collaborative capacity and setting priorities for efficient resource allocation should become a major priority for restoration practitioners in the humid

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tropics. Finally, as global fire regimes are changing and expected to intensify in the context of climate change, land use and land cover change, we suggest to put continuous effort into fire monitoring and modelling to inform the development of effective restoration strategies in the long-run.

1. Introduction

While fires are a natural phenomenon in many temporal, boreal and seasonally-dry tropical forests, nearly all forest fires in the humid tropics are of anthropogenic origin (Moore et al., 2003). Large companies and smallholder farmers, for whom shifting cultivation and the associated use of fire are traditional practices, burn forest to clear land for agriculture, ranching, logging and (oil palm) plantations (Carmenta et al., 2013). In combination with temperature increases and more prolonged droughts, such fires easily get out of control and propagate into adjacent forested areas, leaving behind large areas of degraded land (Alencar et al., 2015; Barlow et al., 2012; Nepstad et al., 2008). Since the 1980s, severe wildfire events have been increasing across the globe, including in humid tropical forest landscapes (White Paper Science Team, 2015). These forests belong to the 20% of global habitats that are not naturally adapted to fire and can therefore be considered as fire-sensitive (Shlisky et al., 2009). In 2015, fire occurrence in the Brazilian Amazon had increased by 36% compared to the preceding 12 years (Aragão et al., 2018). In the same year, daily emission rates from Indonesian peat fires exceeded those of the fossil fuel emissions in the United States, causing serious environmental damage and human health problems across Southeast Asia (Lohberger et al., 2018). Most recently, the 2019-2020 Black Summer fires in Australia, which burned over 10 million hectares of forest and woodland area across the country (Dickman and McDonald, 2020), have made it clear that global fire regimes are changing, with fires increasing in intensity, frequency, and scale of impact (Rogers et al., 2020).

Forest fires are responsible for 5–10% of global greenhouse gas emissions (Boden et al., 2017), with 84% of all global fire emissions originating from the tropics (1830 Tg C year^{-1}) (Van Der Werf et al., 2017). Besides their impacts on the global carbon cycle, tropical wildfires are detrimental to forest ecosystems and local communities (Moore et al., 2003). Not only do they reduce the amount of living biomass, they also affect species composition, alter water and nutrient cycles, increase flood risk and erosion, and threaten local livelihoods by burning agricultural land and homes (Cochrane, 2003; Shlisky et al., 2009). In addition, these fires have devastating impacts on local wildlife as animals either are unable to escape from the fires or become threatened by loss of habitat, food and shelter (Griffiths and Brook, 2014).

Several studies have stressed the risk of fire-related tipping-points, beyond which major and irreversible changes in forest structure and composition occur (e.g. Nepstad et al., 2008; Page et al., 2009). Major efforts are needed to prevent such tipping points and to enable the effective recovery of fire-affected forest areas. However, while much attention has been paid to post-fire environments in temperate and (seasonally) dry tropical forests and savannas (e.g. Vallejo et al., 2012; Verma and Jayakumar, 2015; Wohlgemuth et al., 2009), research on post-fire recovery in the humid tropics is still in an early stage (e.g. Cochrane, 2003; Shlisky et al., 2009). In this article, we provide a synthesis of available restoration strategies in the humid tropics, considering post-fire ecosystem dynamics. We also identify current gaps in knowledge needed for effective restoration after fire, hereby taking into account the environmental and socio-economic context in which these fires occur (see Fig. 1 for a visual representation of the interactions as discussed in this article). As such, this work can inform restoration practice and provides directions for further research in the field of humid tropical forest restoration after fire.

2. Post-fire restoration strategies

Ecosystem restoration covers a large variety of strategies, ranging from unassisted recovery, to removing barriers to succession (e.g. fire, competition, erosion), (re)introducing plant species and establishing commercial plantations or agroforestry systems. Selecting a suitable strategy should be based on several factors, including the local restoration context (e.g. topographic characteristics, frequency and severity of disturbance), potential barriers and resource constraints (e.g. available knowledge, technical capacity and funding), and desired project

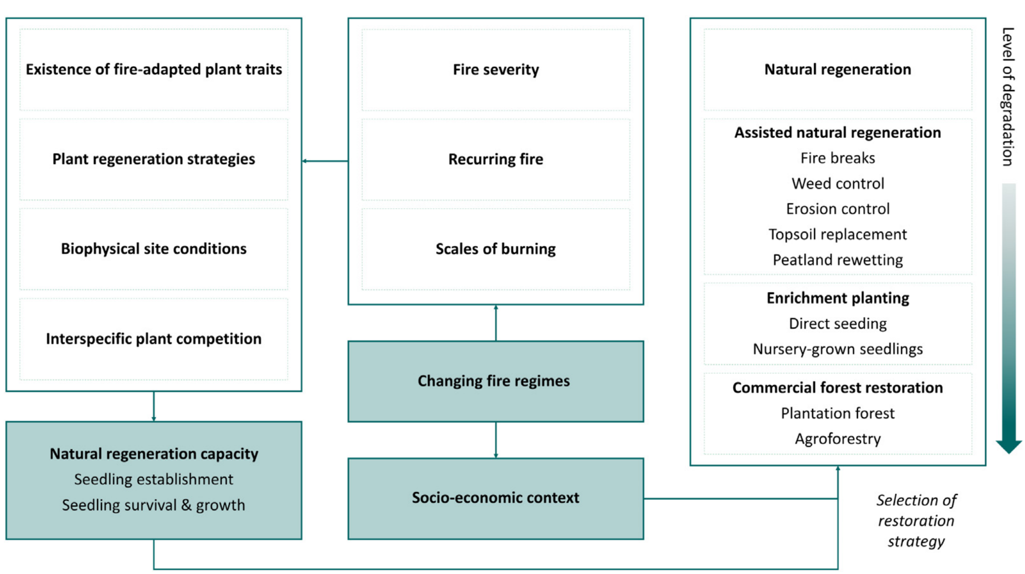

Fig. 1. Post-fire forest restoration.

outcomes (FAO and WRI, 2019; Holl and Aide, 2011). Restoration goals may range from restoring pre-existing forest structure, functioning and species composition, to rehabilitating desired ecosystem services, such as maximizing carbon sequestration or enhancing biodiversity, to creating forest landscapes that serve additional socio-economic purposes (Holl, 2012; FAO, n.d.-a).

2.1. Natural regeneration

Natural regeneration, i.e. the spontaneous recovery of plant and animal species following disturbance, is increasingly promoted as low-cost strategy for large-scale forest restoration projects (Chazdon and Guariguata, 2016). However, degraded tropical forests vary largely in their capacity to recover unassisted. Whether or not natural regeneration is a feasible strategy, depends on a variety of factors affecting seedling establishment (e.g. regeneration strategies, existence of fire-adapted plant traits) and survival and growth (e.g. microclimatic and soil conditions, interspecific plant competition). Understanding post-fire ecosystem dynamics helps to predict the regeneration capacity of a burnt area, so that decision makers know whether to invest in restoration practices or not, and how to allocate their - often limited - resources (Holl and Aide, 2011).

2.1.1. Seedling establishment

High-intensity (sub)surface fires can burn standing vegetation as well as seed-containing soil layers. As propagule availability is essential for natural regeneration, such fires can seriously constraint the recovery process (Holl et al., 2000). However, when soil seedbanks, seedlings or vegetative buds within the post-fire area remain intact, recovery can be successful (Lamb et al., 2005). Because of contrasting plant traits and regeneration strategies, fire-dependent and fire-sensitive ecosystems have different capacities to resist or recover from fire. For instance, in frequently burning savannas, tree species often have a thick bark that decreases post-fire stem mortality, whereas thinner-barked rainforest species are less likely to survive a fire (Ondei et al., 2016). Some savanna species have seeds that can withstand or even need fire to germinate, whereas others are able to resprout from intact vegetative buds (Vallejo et al., 2012). These resprouters grow fast and compete strongly for light, moisture and space, whereas seeders require longer fire-free periods to establish and out-compete invasive understory species (Ondei et al., 2016). While many rainforest species have the capacity to resprout (e.g. 78% in northern Queensland, Australia), basal resprouters (91% of all resprouting rainforest species) are more vulnerable to high-intensity fires, whereas epicormic or aerial resprouters (5% of all resprouting rainforest species), which are more common in savanna ecosystems, experience little mortality after fire (Clarke et al., 2013; Clarke et al., 2015).

Post-fire regeneration can also occur via colonization, depending on fire severity, environmental factors such as vegetation type, topography and meteorology, and proximity of the burnt area to remnant forest patches (Griscom and Ashton, 2011; Watson et al., 2012). While wind-dispersed savanna species generally have a large dispersal range, large-seeded, animal-dispersed species (Cramer et al., 2007), which constitute 50–90% of all tropical rainforest species (Howe and Smallwood, 1982), are more sensitive to forest fragmentation. For instance, Pereira et al. (2013) found that seed dispersal of late successional and animal-dispersed species occurred at up to 4000 m from the seed source. Hence, in large, continuous areas of burnt land, forest seed sources are likely out of reach. In addition, when species depend on animals for seed dispersal, the distance of a forest landscape to sources of disturbance, such as urban areas, roads and rivers, can influence the colonization process (De Rezende et al., 2015).

2.1.2. Seedling survival and growth

If species are able to establish in the burnt area, a number of factors influence seedling survival and growth, including microclimatic conditions, i.e. temperature, solar radiation and humidity, soil nutrient availability, and competition with invasive species (Holl et al., 2000). Also the presence of ash should be considered, as ash can have profound effects on vegetation growth, e.g. by increasing soil pH and nutrient pools, affecting soil texture and hydraulic properties, and stimulating soil microbial activity (Bodí et al., 2014).

Degradation of soil physical, chemical and biological characteristics may increase with wildfire recurrence (Mataix-Solera et al., 2009). As fire reduces protective vegetative cover, soil heating, increased light intensity and reduced soil moisture increase the susceptibility of established seedlings to desiccation (Slik et al., 2008). Such interactions hold the potential to cause irreversible ecosystem damage. For example, Zarin et al. (2005) found evidence of a 50% reduction in biomass accumulation in Amazonian sites that experienced five or more fires, whereas Nepstad et al. (2008) reported a six-fold increase in tree mortality following the reduction of soil water availability to below 30% of its maximum value. In tropical peat swamp forests, fire affects seedling survival by modifying the peatland's hydrological functioning (Dommain et al., 2016). Peat fires hereby increase flood risk by reducing vegetation cover, lowering the peat surface and decreasing the water holding capacity of the soil (Page et al., 2009).

Burnt areas are also prone to competition from non-native, wind-dispersed species such as grasses and ferns that are better adapted to fire and can easily suppress slower reproducing rainforest species (Brooks et al., 2004; Holl et al., 2000). Invasive understory plants, such as *Saccharum spontaneum* in the Panama Canal Watershed (Boeschoten et al., 2020) and *Imperata cylindrica* in Southeast Asia (MacDonald, 2004), often dominate post-fire areas, leading to arrested development of native plant communities (Slik et al., 2008). These species facilitate repeated (high-intensity) fire by increasing fuel loads (Blackham et al., 2014; Ondei et al., 2016), which further reinforces those species (Brooks et al., 2004). Only when trees are able to grow tall, they can shade out invasive species and thereby reduce the risk of repeated fire (Hooper et al., 2005). Nevertheless, both in fire-dependent and fire-sensitive ecosystems, species composition is affected by recurring fire and is unlikely to recover fully within human timescales (Chazdon and Guariguata, 2016; Slik et al., 2008). With each successive fire, species diversity tends to decrease (Page et al., 2009; Verma and Jayakumar, 2015).

2.2. Assisted natural regeneration

From the previous section, we can conclude that natural regeneration can be effective under favourable ecological conditions. However, when soil seedbanks have been combusted and sources of propagules are not available, or when recurring fires reduce the recovery potential of already degraded areas, human intervention may be needed to overcome these barriers (Holl and Aide, 2011). Assisted natural regeneration is the practice of facilitating natural regeneration by reducing biophysical barriers to succession, such as recurring fire, competition with invasive understory species, soil degradation and erosion (Shono et al., 2007). It includes relatively simple, low-cost practices, such as firebreak establishment and weed control, and more complex and costly measures, including erosion control, topsoil replacement and hydrological restoration (i.e. peatland rewetting).

2.2.1. Fire breaks

Despite efforts put into preventing fire, recurring fire remains a major cause of restoration failure (FAO, 2019). As humid tropical forests require relatively long fire-free periods to regenerate, creating natural firebreaks can help to remove stress from repeated fire (Hooper et al., 2005). Such firebreaks are typically cleared strips of land along the boundaries of a deforested area (FAO, 2019) where the highest fire recurrence rates are observed (Alencar et al., 2015). While this practice is not widely implemented in the humid tropics, several studies on tropical forest restoration have underlined the importance of firebreaks to protect planted as well as passively regenerating areas (e.g. Lwanga, 2003; Shono et al., 2007). Firebreaks are a relatively simple and cost-effective measure to prevent the spread of fire, but only if well-maintained and frequently stripped of volatile material (FAO, 2019; Omeja et al., 2011). To motivate local communities to maintain the firebreaks and help prevent the spreading of fire, the FAO (2019) proposes planting food crops in the firebreaks.

2.2.2. Weed control

Removing invasive weeds and grasses can help to increase the diversity of natural regeneration by limiting competition (Holl et al., 2000) and reduce the area's susceptibility to fire by removing fuel (FAO, 2019). In several studies, growth rates of planted seedlings were found to increase after removal of exotic grasses (e.g. Craven et al., 2009; Hooper et al., 2002). A frequently used technique includes marking all naturally regenerating woody species or planted seedlings, and cutting the surrounding grasses (Shono et al., 2007). Although relatively cheap and easy to implement, cutting is a labour-intensive practice that is only suitable for small-scale intensive restoration projects (Hooper et al., 2005). In addition, cutting tends to stimulate faster regrowth and should therefore be repeated frequently (FAO, 2019).

An alternative to cutting is pressing weeds with a wooden board. Pressed vegetation blocks sunlight, therewith killing the lower weed layers and suppressing further weed germination. Weed pressing also functions as an erosion control measure and helps to reduce fire intensity as pressed plants are less flammable (FAO, 2019). In addition, pressed weeds can have beneficial effects on seedling survival, for instance by reducing soil temperature and light intensity (Lazos-Chavero et al., 2016). Although pressing is a labour-intensive practice, it requires less repetition than cutting.

Another cheap practice to control competing vegetation is chemical weeding, or the application of herbicides. Especially in regions with high labour costs, chemical weeding is often applied instead of manual cutting (Little et al., 2006). Manual, mechanical and aerial application of herbicides is used in both small and large-scale restoration efforts. A major disadvantage of chemical weeding is its harmful environmental impact. Herbicides can be toxic to animals and non-target vegetation, hence – if used at all - strict compliance with regulations and application instructions is necessary (Löf et al., 2012).

2.2.3. Erosion control

Burnt forest areas, particularly on steep slopes, are vulnerable to soil erosion resulting from reduced ground and canopy cover and altered soil physical properties (Labrière et al., 2015; Vallejo et al., 2012). To minimize this effect, measures of erosion control can be implemented. One such practice is mulching, i.e. covering the surface with a layer of organic material for soil and water conservation and to facilitate plant growth (Jordán et al., 2011). A variety of materials can be used for this purpose, the most common of which are straw, forestry residues and hydromulch, a slurry of wood, paper fibre and non-water-soluble binder (Wohlgemuth et al., 2009). The effect of mulching is immediate and since several mulching materials have proven to be effective erosion treatments, the use of local (forest) materials can be promoted.

Although the practice is often recommended to stabilize hillslopes and enhance post-fire plant productivity (Bautista et al., 2009; Jordán et al., 2011), mulch effectiveness varies considerably depending on site characteristics and materials used. For example, some studies have shown neutral effects of mulching on vegetation recovery after severe wildfire (e.g. Fernández et al., 2019). Others describe negative side-effects related to the use of straw and hydromulch. While straw is known to be a vector for invasive, non-native species, excessive application of hydromulch can suppress vegetation growth (Bautista et al., 2009). Finally, studies conducted on post-fire mulching have mostly been short-term, and focused on dry environments (e.g. Fernández et al., 2019; Morgan et al., 2015), whereas its long-term effects on vegetation recovery and species composition, particularly in tropical regions, have not yet been assessed.

Furthermore, the costs of mulching vary greatly, depending on the material used, method of application and site characteristics. Straw can be distributed by air and is relatively cheap (US$1850 ha^{-1} in Southern California), whereas manual placement is more labour intensive, but often results in better soil coverage. The aerial distribution of wood mulch and, in particular, hydromulch (US$4000 ha^{-1} in Southern California) is useful for large-scale implementation but relatively expensive, whereas road hydromulching, i.e. application through truck-mounted hoses, has a limited coverage range (Wohlgemuth et al., 2009).

2.2.4. Topsoil replacement

Topsoil replacement, i.e. the transfer of soil containing leaf litter, soil microorganisms, plant fragments and seeds from a newly cleared area to a degraded site, is a practice to accelerate natural regeneration and restore plant communities (Ferreira and Vieira, 2017), typically used for post-mining restoration (Parrotta and Knowles, 1999; Rokich et al., 2000). Although literature on the use of topsoil to restore burnt tropical forests is not available, the treatment is likely to improve the conditions for seedling establishment, especially in areas where seed-containing soil layers and topsoil organic matter have been combusted by a severe fire. Topsoil replacement was found to facilitate the propagation of trees, lianas, shrubs and herbs in degraded tropical dry forests (Ferreira and Vieira, 2017). In the same study, species richness and tree density were found to increase after the deposition treatment.

The costs of the treatment include soil preparation at the deposition site, topsoil stripping, transportation and application. Costs depend on the thickness of the applied topsoil layer, the distance to the deposition site and whether or not fixed costs (e.g. avoided costs of transportation to landfills) can be deducted (Ferreira and Vieira, 2017). As stockpiling topsoil reduces the potential of seedling recruitment, the topsoil should be used immediately after stripping (Rokich et al., 2000). Other constraints of topsoil replacement include logistical challenges associated with large-scale application, the need for technical expertise to ensure appropriate topsoil handling and uncertainty related to post-treatment species composition (Parrotta and Knowles, 1999).

2.2.5. Peatland rewetting

The spread of fire promoted by artificial drainage, often due to canals dug for the transportation of logged trees, is a major factor driving tropical peatland degradation, particularly in Indonesia (Jaenicke et al., 2010). Dry peat is highly susceptible to ignition, has a long burning period and can smoulder belowground, resulting in fire spreading well beyond the source of ignition. As altered hydrological conditions are a major barrier to tropical peat forest regeneration (Graham et al., 2017; Page et al., 2009), restoring this balance should be the starting point of any peat forest restoration effort. Rewetting is essential in restoring the peat's hydrological functions, reduces the risk of repeated fire and flooding (Page et al., 2009), prevents further soil subsidence, peat decomposition and associated CO_2 emissions (Jaenicke et al., 2010) and allows for the restoration of peatland biodiversity (Dommain et al., 2016).

Blocking drainage canals is a common method to raise groundwater levels (Jaenicke et al., 2010). This can be done by placing dams, ranging from small locally-constructed dams to larger, more sophisticated designs (Page et al., 2009). Since 2005, numerous dams have been placed by Wetlands International and WWF-Indonesia in Central Kalimantan (Ritzema et al., 2014). Several obstacles were identified during these efforts; in particular material transport was extremely challenging due to waterlogged conditions and a low bearing capacity of the peat. In addition, the high permeability of the peat caused seepage, internal erosion and dam instability. Poor accessibility is another major constraint to restoring tropical peatlands, as the rewetting process often involves blocking illegal logger canals located at highly inaccessible sites (Jaenicke et al., 2010).

2.3. Enrichment planting

One of the most commonly used strategies to accelerate forest recovery is enrichment planting, i.e. the introduction of valuable species to degraded forests without the elimination of valuable species already present (Montagnini et al., 1997). Severely burnt areas often have low numbers of natural regenerants, low species diversity and contain fragmented forest patches (FAO, 2019), and may therefore need some form of seeding or planting.

2.3.1. Direct seeding

Direct seeding is a relatively inexpensive method to (re)introduce native species in fragmented tropical forests and typically involves seed collection from nearby forest patches, storage and sowing (FAO, 2019). As the costs of raising, transporting and planting seedlings (see Section 2.3.2) are avoided and seed distribution is fairly simple, direct seeding is suitable for largescale implementation, also at less accessible sites (Cole et al., 2011; FAO, 2019).

While direct seeding is considered an efficient way to enrich existing systems, it is less suitable for restoring severely degraded landscapes (Cole et al., 2011; Lamb et al., 2005). In recently burnt areas, seed mortality is generally high due to poor soil conditions, water stress and desiccation (Palma and Laurance, 2015). Delivering seeds with nutrient-rich (hydro)mulch can help to facilitate establishment (Palma and Laurance, 2015), while reducing the risk of predation (FAO, 2019), whereas selecting species with thick seed coats (e.g. leguminous trees) can help to overcome the problem of desiccation (FAO, 2019). In peat swamp forests, careful selection of species that can withstand both flooding and drought is needed to enhance the survival of seeds and seedlings (Lampela et al., 2017).

As several studies have shown the potential of late-successional, large-seeded species to establish in the understory of successional forests (e.g. Cole et al., 2011; Hooper et al., 2002), direct seeding could complement the planting of fast-growing nurse trees (see Section 2.3.2).

2.3.2. Planting nursery-grown seedlings

Planting nursery-raised seedlings is the most widely adopted method to restore degraded humid tropical forests (Lamb et al., 2005). When the goal is to restore a forest ecosystem to its original state, the reintroduction of native species should be prioritized, especially in unique ecosystems such as tropical peat swamp forests (Lampela et al., 2017). However, seed availability (Palma and Laurance, 2015) and knowledge of nursery and planting techniques (Kobayashi, 2004) are often limited to exotic or commercial species. To date, the most common approach is therefore to plant a number of early-successional, exotic seedlings that function as nurse plants by shading out weeds and grasses, improving soil conditions, reducing fire risk and facilitating seed dispersal and colonization by increasing forest connectivity (Griscom and Ashton, 2011; Holl et al., 2000; Lamb et al., 2005). Planting pioneer species can facilitate natural establishment of native species in the understory (Omeja et al., 2011) or the promotion of late-successional species by under-planting (Ashton et al., 2001) or seeding (Cole et al., 2011).

The most reported disadvantage of planting seedlings is its high cost as it involves nursery management, transportation and field labour (FAO, n.d-a; Omeja et al., 2011). Additional expenses on weed control, mulch or fertilizer application or extensive monitoring can hereby add up quickly (Palma and Laurance, 2015). Cole et al. (2011) estimated the costs of planting nursery-raised seedlings to be 10 to 30 times those of direct seeding (see Section 2.3.1), while costs varied largely depending on planting density, site characteristics and post-planting maintenance. As high-density planting is often restricted to small-scale restoration projects, applied nucleation, also referred to as the 'tree island' method, is sometimes recommended as a cost-effective alternative. This method is based on planting clusters of trees to serve as focal areas for recovery instead of rows of trees covering the entire area (Corbin and Holl, 2012). Holl et al. (2011) compared the two planting designs and found survival rates of seedlings to be similar in both designs, whereas the growth rate of seedlings was slightly lower in the cluster set-up. In addition, Cole et al. (2010) found that plantations and medium-to-large-sized tree clusters (64–144 m^2) have comparable potential to enhance seed dispersal of early-successional species by providing habitat for seed-dispersing birds and bats, while the costs of tree clusters were only a third of those for plantation establishment.

Another potential limitation of this method is the complexity of raising and transplanting seedlings. Creating suitable nursery conditions is essential to enhance seedling survival in postfire areas (Graham et al., 2017; FAO, n.d.-a). For instance, practicing with water stress facilitates the development of drought-resistant plant traits (Palma and Laurance, 2015), whereas inoculating seedlings with native mycorrhizae reduces the chance of a transplantation shock (Urgiles et al., 2009). Furthermore, in areas that are prone to flooding, such as burnt peatlands, seedlings may need to be planted on artificial mounds to reduce the change of prolonged inundation (Dommain et al., 2016; Graham et al., 2017). Overall, enrichment planting requires broad knowledge of an ecosystem's structure and functioning. For instance, as peatlands naturally support few and relatively small trees, excessive tree planting can have counterproductive effects by further increasing the area's susceptibility to high-intensity fires (Wilkinson et al., 2018). In addition, species selection is crucial, as not all species are as effective at stimulating succession (Griscom and Ashton, 2011).

2.4. Commercial restoration

The strategies discussed so far can be categorized as ecological restoration strategies, intended to rehabilitate ecosystem functioning without generating commercial value. However, commercial forests, which may provide less ecological value but offer more immediate socioeconomic benefits, account for a substantial portion of restored forest area. Commercial restoration may include plantation forests (e.g. for the production of timber or paper pulp) and agroforestry practices.

2.4.1. Plantation forests

The environmental value of plantation forests is much discussed and their contribution to forest restoration strongly depends on the characteristics of the plantation and the state of the ecosystem it replaces (Brockerhoff et al., 2008). When areas are degraded to the extent that soils are exhausted, seed sources absent and native species unlikely to become established, plantation forests can function as a last resort to restore site productivity. In addition, plantation forests can help to reduce pressure on intact natural forests and support local economies (Löf et al., 2019). Overall, plantations are intensive restoration projects that require significant investment in terms of time and money, while providing opportunity for financial returns (Griscom and Ashton, 2011). In many tropical countries, monocultures of exotic fast-growing species are favoured for the production of high quality timber, whereas plantations with slower growing native species can add structure, improve habitat quality and enhance carbon sequestration, but are less common (Lamb et al., 2005; Shono et al., 2007). Monoculture plantations generally have low capacity to resist or recover from disturbances such as plant disease or wildfire (Chazdon and Guariguata, 2016). In addition, while plantations of both exotic and native species can enhance forest recovery on severely degraded sites where ecological barriers would otherwise prevent succession, intensively-managed, short-rotational exotic plantations are of little value for biodiversity conservation (Brockerhoff et al., 2008). To meet objectives of both plantation productivity and biodiversity, mixtures of pioneer and high-value native trees can be planted (Hooper et al., 2002).

2.4.2. Agroforestry

Another approach to commercial restoration is the agro-successional approach, where a range of agroforestry techniques are employed during the early stages of forest restoration (Vieira et al., 2009). Valuable native tree species can hereby be underplanted with fast-growing agricultural cash crops such as coffee and cocoa (Lamb et al., 2005). Restoration through agroforestry offers a number of environmental benefits, such as biodiversity conservation, carbon sequestration, water storage, and soil fertility (Jose, 2009). When located in the buffer zone of a natural forest area, agroforestry systems can provide a critical refuge for forest species after a catastrophic fire event (Griffith, 2000). Other advantages of the agro-successional approach include the provisioning of additional sources of income and the reduction of restoration costs as site management is generally taken care of by local landowners (Vieira et al., 2009).

3. Knowledge gaps and recommendations

There is no one-size-fits-all approach to restore burnt humid tropical forests and the suitability of each restoration strategy is highly context-dependent. When looking at forest restoration in general, a broad consensus seems to exist on the use of a multiple-phase restoration pathway, which typically encompasses planning-and-design, implementation and monitoring-and-evaluation phases (e.g. Lazos-Chavero et al., 2016; WRI, 2015). Challenges may emerge in each of these phases and ways to overcome such challenges are much discussed. For instance, WRI and IUCN initiated the Restoration Diagnostic, a structured method to identify key success factors in existing forest restoration projects. These factors are grouped into three major themes: 1) stakeholders should be inspired or motivated to engage in the restoration process, 2) ecological, market, policy, social and institutional conditions that create a favourable restoration context should be in place and 3) capacity and resources for sustained implementation should be mobilized (WRI, 2015). Other relevant contributions to effectively design, implement and monitor restoration projects, include the Forest and Landscape Restoration Guide of the FAO and the WRI (FAO and WRI, 2019) and the Standards for the Practice of Ecological Restoration developed by the Society for Ecological Restoration (Gann et al., 2019).

While such studies provide useful insights into the success factors of restoration in general, there are some important implications of fire as a driver of humid tropical forest degradation, that are not sufficiently accounted for in existing restoration efforts. These implications, including fire severity, recurring fire, increasing scales of burning, and fire regime variability, are discussed below and translated into knowledge gaps for effective restoration after fire.

3.1. Accounting for fire severity

Although the fact that global fire regimes are changing is widely acknowledged, the implications of such changes for restoration practice have so far received little attention. Particularly, knowledge on the characteristics of forest fires preceding restoration efforts is limited. This is remarkable, given that the recovery potential of a burnt forest is largely determined by these characteristics (Cochrane, 2003). Ecosystem responses, including among others vegetation regeneration, faunal recolonization, hydrological processes and soil erosion, are affected by fire severity, which is described as the loss of above– and belowground organic matter. Fire severity can, in turn, be explained by the physical characteristics of fire, such as fire intensity, burning depth and duration of soil heating (Keeley, 2009). These fire characteristics – and their effects – are not universal, hence should be analysed in their specific context.

Despite the fact that humid tropical forests burn less frequently than dry forests, they are more vulnerable to the effects of fire and recover more slowly than forests with a longer fire history (see also Section 2.1). While severe fires in temperate and boreal forests are mostly fast-spreading, high-intensity surface or crown fires, most fires in the humid tropics are neither fast-spreading nor intense due to high levels of fuel moisture content. They can, nevertheless, be severe as slow-spreading surface or ground fires often result in persistent heating (i.e. the longest burning fires (>60 days) have been observed in the tropics (Andela et al., 2019)), which causes high tree mortality (Cochrane, 2003) and damage to belowground biota. The sterilizing effect of heat on soil microorganisms hereby disrupts soil system functioning and the re-establishment of (native) vegetation (Mataix-Solera et al., 2009). Hence, in the absence of fire-resistant plant traits, forest structure and species composition can change dramatically following fire (Barlow et al., 2003). Such changes are, in turn, likely to disrupt faunal communities, therewith affecting crucial seed dispersal processes. While some studies describe the responses of trees (e.g. Barlow et al., 2003), bird communities (e.g. Barlow et al., 2002), small mammals, amphibians and reptile communities (e.g. Fredericksen and Fredericksen, 2002) after a single, low-intensity fire, the interactions between fire, plants and animals under different fire regimes remain poorly understood. Also, while soil degradation is considered one of the most damaging post-fire processes in (seasonally) dry forest systems (Vallejo et al., 2012), documentation of the impacts of fire on soil aspects in the humid tropics, including soil microbiology as well as soil physical (e.g. water repellency and soil structure) and chemical properties (e.g. nutrient availability), is limited.

Overall, the recovery potential of a burnt forest decreases with increasing fire severity. However, to be able to better understand and predict ecosystem responses to fire and develop effective restoration methods, the specific interactions between measures of fire severity, soils and biota require further study. This is particularly true for the humid tropics, where the starting point for post-fire restoration is different and less well known compared with fire-adapted systems.

3.2. Preventing recurring fire

Another essential difference between fire and other drivers of forest degradation is the fact that the former poses a recurring and intensifying threat throughout the recovery process, whereas disturbances such as logging and land clearance often have a more temporary impact on the successional process (Ashton et al., 2001). Previously burnt forests in the humid tropics are particularly susceptible to fire (Cochrane, 2003), even during wetter-than-average periods (Alencar et al., 2015). The continuous build-up of fuel load in the understory over time hereby increases fire intensity (Cochrane, 2003). In the context of a changing climate, with rising temperatures and more pronounced El Niño and Southern Oscillation events (Shlisky et al., 2009), and weak environmental regulation in the countries that are most subject to these fires (e.g. Indonesia, Brazil), the trend towards increased fire incidence is likely to continue.

Preventing recurring fire should be a fundamental aspect of the restoration process, especially when the forest recovery period is longer than the fire-return interval (Barlow et al., 2012). In practice, loss due to fire is an often reported cause of failure of restoration projects in the humid tropics (e.g. Boeschoten et al., 2020; Carmenta et al., 2013). These failures typically reflect a lack of proper post-restoration management and long-term perspective (Graham et al., 2017). A common assumption is that systems will recover naturally after their assisted onset. Consequently, failure to recognize that restoration is an ongoing process and requires periodic attention to minimize the risk of repeated fire can result in disappointing outcomes, despite significant investments in time and money (Hilderbrand et al., 2005). In addition, the long time required to generate visible results of restoration efforts and the need for long-term funding can reduce the motivation of different stakeholders to engage in restoration (Schweizer et al., 2019; Zahawi et al., 2014). Hence, to promote sustainability of restoration activities, more efforts should be invested into post-restoration management and the prevention of fire throughout the different phases of the restoration process. Approaches to stimulate continued support for restoration efforts are much discussed and include, among many others, educating practitioners on the risks of altered fire regimes (e.g. by quantifying the socioeconomic costs of fire) (Shlisky et al., 2009), training specialized restoration agents to work with landowners and develop tailored restoration plans (Chazdon and Guariguata, 2016), ensuring transparent communication throughout the restoration process (e.g. by communicating early successes and failures) (WRI, 2015), strictly monitoring restoration activities and employing performance-based payment mechanisms (Boeschoten et al., 2020), providing for productive restoration activities (e.g. by allowing the incorporation of commercial trees or crops in the affected area) (Schweizer et al., 2019) and actively involving local communities in decision-making processes (Jaenicke et al., 2010; Page et al., 2009). Participatory approaches are important to consider as misalignments between policy requirements and local practices are often reported (Boedhihartono et al., 2018; Carmenta et al., 2013). Hence, in order to reduce fire risk, insights are needed into the underlying factors that facilitate or undermine individual motivations to make such changes. Measures should thereby not only aim at preventing harmful practices, but also offer alternatives to sustain or improve human livelihoods (Jaenicke et al., 2010).

3.3. Addressing increasing scales of burning

As tropical wildfires are increasing in scale and its impacts stretch beyond (national) borders, upscaling restoration is now more important than ever. To enable forest restoration at the scales required, it is critical to create and sustain collaborative capacity and focus on setting priorities for efficient resource allocation.

3.3.1. Establishing collaborative capacity

While upscaling requires both horizontal (i.e. establishing cross-sectoral partnerships) and vertical (i.e. balancing local, regional, and global priorities) integration, frequently mentioned barriers include the lack of institutional capacity to establish such collaborations (Chazdon and Guariguata, 2016) and limited political will (Schweizer et al., 2019). In addition, conflicting interests complicate successful cooperation between different stakeholders involved in the restoration process. This is particularly evident in tropical countries, where trade-offs between environmental outcomes and human livelihoods are often reported (Lamb et al., 2005). The widespread production of cash crops in tropical rural areas hereby often results in the conversion, rather than the restoration of degraded forests (Van der Laan et al., 2017). A growing number of studies underline the importance of landscape-based restoration approaches (e.g. Vermunt et al., 2020), and initiatives such as the Forest and Landscape Restoration Mechanism (FAO, n.d.-b), provide guidance for upscaling restoration efforts in accordance with national or regional contexts, support multi-level collaboration and explore investment opportunities to create restoration-based value chains. However, while such integrated approaches are widely promoted, concrete examples of post-fire restoration on landscape scale are lacking, and the establishment of collaborative capacity, needed to improve restoration outcomes at local, regional and global scales, requires further attention.

3.3.2. Setting priorities for efficient resource allocation

Another major constraint to forest restoration in the humid tropics is the lack of financial resources (Vieira et al., 2009). While an estimated amount of US$350 billion per year is needed globally for conservation and restoration, only US$50 billion is currently available (Credit Suisse et al., 2014). This often restricts large-scale restoration to low-cost natural regeneration, which is not always effective (Holl and Aide, 2011). Likewise, the fact that many forest areas in the humid tropics are remote, without vehicular access, complicates large-scale restoration, particularly when involving human labour (Elliott, 2016). Besides supporting the development of cost-effective alternatives to existing restoration strategies and methods of propagule distribution (e.g. aerial seeding and drone-based weeding), emphasis should be placed on identifying priority areas, i.e. areas where restoration is expected to be both beneficial and feasible, to ensure that limited resources are allocated as efficiently as possible. Brancalion et al. (2019) identified global restoration hotspots in tropical rainforest landscapes based on a number of socio-environmental benefits (e.g. potential for providing habitat for vulnerable species, reducing atmospheric CO_2 concentrations and reducing water security risk), investment costs and risks (e.g. land opportunity costs and landscape variation in restoration success). Implementing the hotspot approach at regional scales, i.e. by adding context-specific criteria, can help to guide cost-effective restoration outcomes.

3.4. Dealing with fire regime variability

Forest fires can spread quickly over large areas and change landscapes abruptly, while other drivers of forest degradation, such as logging, operate more continuously in space and time. This dynamic nature of fire regimes highlights the need for restoration initiatives that are flexible enough to respond to unexpected disturbance and changing climatic and ecological factors (Carmenta et al., 2013). Future studies should therefore not only take into account current fire regimes, but also address fire regime variability, as unanticipated changes in fire characteristics, fire frequencies and scales of burning (see Sections 3.1, 3.2 and 3.3) can have major implications for restoration practice. Predicting fire behaviour and its effects on ecosystems can hereby help to inform policy and facilitate effective restoration in the long run. While fire modelling is extremely challenging, considerable progress has recently been made due to the growing availability of field observations and satellite imagery, technological advancements and increasingly accurate models (see Rogers et al., 2020). In addition, efforts to map regional and global fire occurrence by using remote sensing have intensified, with online platforms such as the Global Forest Watch (Global Forest Watch, n.d.) or the Global Fire Atlas (Andela et al., 2019) providing near-real time data to monitor forest fires. As global fire regimes keep changing and are expected to intensify, continuous effort should be put into fire monitoring and modelling in the context of climate, land use and land cover change to inform the development of effective restoration strategies.

4. Conclusion

In this synthesis paper, we discussed a variety of restoration strategies that can be employed to restore fire-affected areas in the humid tropics. Depending on the specific ecological and socioeconomic restoration context, different combinations of restoration practices can be adopted. Our analysis shows that empirical evidence of successful post-fire restoration in the humid tropics is scarce. In particular, little information is available on what happens after the initial stage of a restoration project. Long-term studies of post-fire restoration pathways are needed to determine the rates of forest recovery, and to identify how different interventions match with fire characteristics and ecological conditions. Humid tropical forests hereby require specific attention as they are more vulnerable to the effects of fire, and are expected to follow different successional routes than fire-adapted ecosystems. Furthermore, to ensure effective post-fire restoration in the long run, we suggest putting more emphasis on the severity of fire preceding restoration efforts, accounting for the role of fire as a recurring disturbance and taking into account the spatiotemporal dynamics of (global) fire regimes. In addition, the majority of studies on post-fire restoration in the humid tropics focuses on the (Brazilian) Amazon and – when it concerns peatland restoration – Indonesia, with relatively little coverage of Africa, other parts of South America, Southeast Asia and the Pacific. As the impacts of fire as well as the effects of restoration practices vary across regions, it is particularly critical that the geographical focus of restoration studies is expanded.

CRediT authorship contribution statement

Anke C. Scheper: Investigation, Writing – original draft. **Pita A. Verweij:** Conceptualization, Funding acquisition, Supervision, Writing – review & editing. **Marijke van Kuijk:** Conceptualization, Funding acquisition, Supervision, Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Alencar, A.A., Brando, P.M., Asner, G.P., Putz, F.E., 2015. Landscape fragmentation, severe drought, and the new Amazon forest fire regime. Ecol. Appl. 25 (6), 1493–1505. https://doi.org/10.1890/14-1528.1.

Andela, N., Morton, D.C., Giglio, L., Paugam, R., Chen, Y., Hantson, S., van der Werf, G.R., Randerson, J.T., 2019. The Global Fire Atlas of individual fire size, duration, speed and direction. Earth Syst. Sci. Data 11 (2), 529–552. https://doi.org/10.5194/essd-2018-89.

Aragão, L.E., Anderson, L.O., Fonseca, M.G., Rosan, T.M., Vedovato, L.B., Wagner, F.H., Silva, C.V.J., Silva Junior, C.H.L., Arai, E., Aguiar, A.P., Barlow, J., Berenguer, E., Deeter, M.N., Domingues, L.G., Gatti, L., Gloor, M., Maldi, Y., Marengo, J.A., Miller, J.B., Phillips, O.L., Saatchi, S., 2018. 21st Century drought-related fires counteract the decline of Amazon deforestation carbon emissions. Nat. Commun. 9 (1), 1–12. https://doi.org/10.1038/s41467-017-02771-y.

Ashton, M.S., Gunatilleke, C.V.S., Singhakumara, B.M.P., Gunatilleke, I.A.U.N., 2001. Restoration pathways for rain forest in southwest Sri Lanka: a review of concepts and models. For. Ecol. Manag. 154 (3), 409–430. https://doi.org/10.1016/S0378-1127(01)00512-6.

Barlow, J., Haugaasen, T., Peres, C.A., 2002. Effects of ground fires on understorey bird assemblages in Amazonian forests. Biol. Conserv. 105 (2), 157–169. https://doi.org/10.1016/S0006-3207(01)00177-X.

Barlow, J., Peres, C.A., Lagan, B.O., Haugaasen, T., 2003. Large tree mortality and the decline of forest biomass following Amazonian wildfires. Ecol. Lett. 6 (1), 6–8. https://doi.org/10.1046/j.1461-0248.2003.00394.x.

Barlow, J., Parry, L., Gardner, T.A., Ferreira, J., Aragão, L.E.O.C., Carmenta, R., Berenguer, E., Vieira, I.C.G., Souza, C., Cochrane, M.A., 2012. The critical importance of considering fire in REDD+ programs. Biol. Conserv. 154, 1–8. https://doi.org/10.1016/j.biocon.2012.03.034.

Bautista, S., Robichaud, P.R., Bladé, C., 2009. Post-fire mulching. In: Cerdà, A., Robichaud, P.R. (Eds.), Fire Effects on Soils and Restoration Strategies. vol. 5. Science Publishers, Enfield, NH, pp. 353–372.

Blackham, G.V., Webb, E.L., Corlett, R.T., 2014. Natural regeneration in a degraded tropical peatland, Central Kalimantan, Indonesia: implications for forest restoration. For. Ecol. Manag. 324, 8–15. https://doi.org/10.1016/j.foreco.2014.03.041.

Boden, T.A., Marland, G., Andres, R.J., 2017. Global, Regional, and National Fossil-fuel CO2 Emissions (1751–2014) (v. 2017). Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, TN https://doi.org/10.3334/CDIAC/00001_V2017.

Bodí, M.B., Martin, D.A., Balfour, V.N., Santín, C., Doerr, S.H., Pereira, P., Cerdà, A., Mataix-Solera, J., 2014. Wildland fire ash: production, composition and eco-hydro-geomorphic effects. Earth Sci. Rev. 130, 103–127. https://doi.org/10.1016/j.earscirev.2013.12.007.

Boedhihartono, A.K., Bongers, F., Boot, R.G., van Dijk, J., Jeans, H., van Kuijk, M., Koster, H., Reed, J., Sayer, J., Sunderland, T., Turnhout, E., van Vianen, J., Zuidema, P.A., 2018. Conservation science and practice must engage with the realities of complex tropical landscapes. Trop. Conserv. Sci. 11, 1940082918779571. https://doi.org/10.1177/1940082918779571.

Boeschoten, L.E., Van Breugel, M., Bailon, M., Balbuena, J., Nuñez, M., Cerezo, A., Hall, J.S., 2020. Framework species approach proves robust in restoring forest on fire prone invasive grass: a case study from Panama. J. Sustain. For., 1–19 https://doi.org/10.1080/10549811.2020.1746915.

Brancalion, P.H., Niamir, A., Broadbent, E., Crouzeilles, R., Barros, F.S., Zambrano, A.M.A., Baccini, A., Aronson, J., Goetz, S., Leighton Reid, J., Strassburg, B.B., Wilson, S., Chazdon, R.L., 2019. Global restoration opportunities in tropical rainforest landscapes. Sci. Adv. 5 (7), eaav3223. https://doi.org/10.1126/sciadv.aav3223.

Brockerhoff, E.G., Jactel, H., Parrotta, J.A., Quine, C.P., Sayer, J., 2008. Plantation forests and biodiversity: oxymoron or opportunity? Biodivers. Conserv. 17 (5), 925–951. https://doi.org/10.1007/s10531-008-9380-x.

Brooks, M.L., D'antonio, C.M., Richardson, D.M., Grace, J.B., Keeley, J.E., DiTomaso, J.M., Hobbs, R.J., Pellant, M., Pyke, D., 2004. Effects of invasive alien plants on fire regimes. BioScience 54 (7), 677–688. https://doi.org/10.1641/0006-3568(2004)054[0677:EOIAPO]2.0.CO;2.

Carmenta, R., Vermeylen, S., Parry, L., Barlow, J., 2013. Shifting cultivation and fire policy: insights from the Brazilian Amazon. Hum. Ecol. 41 (4), 603–614. https://doi.org/10.1007/s10745-013-9600-1.

Chazdon, R.L., Guariguata, M.R., 2016. Natural regeneration as a tool for large-scale forest restoration in the tropics: prospects and challenges. Biotropica 48 (6), 716–730. https://doi.org/10.1111/btp.12381.

Clarke, P.J., Lawes, M.J., Midgley, J.J., Lamont, B.B., Ojeda, F., Burrows, G.E., Enright, N.J., Knox, K.J.E., 2013. Resprouting as a key functional trait: how buds, protection and resources drive persistence after fire. New Phytol. 197 (1), 19–35. https://doi.org/10.1111/nph.12001.

Clarke, P.J., Lawes, M.J., Murphy, B.P., Russell-Smith, J., Nano, C.E.M., Bradstock, R., Enright, N.J., Fontaine, J.B., Gosper, C.R., Radford, I., Midgley, J.J., Gunton, R.M., 2015. A synthesis of postfire recovery traits of woody plants in Australian ecosystems. Sci. Total Environ. 534, 31–42. https://doi.org/10.1016/j.scitotenv.2015.04.002.

Cochrane, M.A., 2003. Fire science for rainforests. Nature 421 (6926), 913–919. https://doi.org/10.1038/nature01437.

Cole, R.J., Holl, K.D., Zahawi, R.A., 2010. Seed rain under tree islands planted to restore degraded lands in a tropical agricultural landscape. Ecol. Appl. 20 (5), 1255–1269. https://doi.org/10.1890/09-0714.1.

Cole, R.J., Holl, K.D., Keene, C.L., Zahawi, R.A., 2011. Direct seeding of late-successional trees to restore tropical montane forest. For. Ecol. Manag. 261 (10), 1590–1597. https://doi.org/10.1016/j.foreco.2010.06.038.

Corbin, J.D., Holl, K.D., 2012. Applied nucleation as a forest restoration strategy. For. Ecol. Manag. 265, 37–46. https://doi.org/10.1016/j.foreco.2011.10.013.

Cramer, J.M., Mesquita, R.C., Williamson, G.B., 2007. Forest fragmentation differentially affects seed dispersal of large and small-seeded tropical trees. Biol. Conserv. 137 (3), 415–423. https://doi.org/10.1016/j.biocon.2007.02.019.

Craven, D., Hall, J., Verjans, J.M., 2009. Impacts of herbicide application and mechanical cleanings on growth and mortality of two timber species in Saccharum spontaneum grasslands of the Panama Canal Watershed. Restor. Ecol. 17 (6), 751–761. https://doi.org/10.1111/j.1526-100X.2008.00408.x.

Credit Suisse, WWF, McKinsey & Company, 2014. Conservation finance: moving beyond donor funding toward an investor-driven approach. Retrieved from. https://www.cbd.int/financial/privatesector/g-private-wwf.pdf (Zurich).

De Rezende, C.L., Uezu, A., Scarano, F.R., Araujo, D.S.D., 2015. Atlantic Forest spontaneous regeneration at landscape scale. Biodivers. Conserv. 24 (9), 2255–2272. https://doi.org/10.1007/s10531-015-0980-y.

Dickman, C., McDonald, T., 2020. Some personal reflections on the present and future of Australia's fauna in an increasingly fire-prone continent. Ecol. Manag. Restor. 21 (2), 86–96. https://doi.org/10.1111/emr.12403.

Dommain, R., Dittrich, I., Giesen, W., Joosten, H., Rais, D.S., Silvius, M., Wibisono, I.T.C., 2016. Ecosystem services, degradation and restoration of peat swamps in the South-east Asian tropics. In: Bonn, A., Allott, T., Evans, M., Joosten, H., Stoneman, R. (Eds.), Peatland Restoration and Ecosystem Services: Science, Policy and Practice. Cambridge University Press, Cambridge, pp. 253–288.

Elliott, S., 2016. The potential for automating assisted natural regeneration of tropical forest ecosystems. Biotropica 48 (6), 825–833. https://doi.org/10.1111/btp.12387.

FAO, 2019. Restoring Forest Landscapes Through Assisted Natural Regeneration (ANR) – A Practical Manual. Retrieved from. http://www.fao.org/3/ca4191en/CA4191EN.pdf (Bangkok).

FAO, a. Forest restoration and rehabilitation. Retrieved April, 2020, from. http://www.fao.org/sustainable-forest-management/toolbox/modules/forest-restoration-and-rehabilitation/basic-knowledge/en/ (n.d.-a).

FAO, b. The forest and landscape restoration mechanism. Retrieved May, 2020, from. http://www.fao.org/in-action/forest-landscape-restoration-mechanism/background/en/ (n.d.-b).

FAO and WRI, 2019. The Road to Restoration: A Guide to Identifying Priorities and Indicators for Monitoring Forest and Landscape Restoration. Rome, Washington, DC Retrieved from. http://www.fao.org/3/ca6927en/CA6927EN.pdf.

Fernández, C., Vega, J.A., Fontúrbel, T., 2019. Does helimulching after severe wildfire affect vegetation recovery in a coastal area of Northwest Spain? Landsc. Ecol. Eng. 15 (4), 337–345. https://doi.org/10.1007/s11355-019-00386-7.

Ferreira, M.C., Vieira, D.L.M., 2017. Topsoil for restoration: resprouting of root fragments and germination of pioneers trigger tropical dry forest regeneration. Ecol. Eng. 103, 1–12. https://doi.org/10.1016/j.ecoleng.2017.03.006.

Fredericksen, N.J., Fredericksen, T.S., 2002. Terrestrial wildlife responses to logging and fire in a Bolivian tropical humid forest. Biodivers. Conserv. 11 (1), 27–38. https://doi.org/10.1023/A:1014065510554.

Gann, G.D., McDonald, T., Walder, B., Aronson, J., Nelson, C.R., Jonson, J., Hallett, J.G., Eisenberg, C., Guariguata, M.R., Liu, J., Hua, F., Echeverría, C., Gonzales, E., Shaw, N., Decleer, K., Dixon, K.W., 2019. International principles and standards for the practice of ecological restoration. Restor. Ecol. 27 (S1), S1–S46. https://doi.org/10.1111/rec.13035.

Global Forest Watch, d. Forest monitoring, land use & deforestation trends. Retrieved March, 2020, from. https://www.globalforestwatch.org (n.d.).

Graham, L.L., Giesen, W., Page, S.E., 2017. A common-sense approach to tropical peat swamp forest restoration in Southeast Asia. Restor. Ecol. 25 (2), 312–321. https://doi.org/10.1111/rec.12465.

Griffith, D.M., 2000. Agroforestry: a refuge for tropical biodiversity after fire. Conserv. Biol. 14 (1), 325–326. https://doi.org/10.1046/j.1523-1739.2000.99101.x.

Griffiths, A.D., Brook, B.W., 2014. Effect of fire on small mammals: a systematic review. Int. J. Wildland Fire 23 (7), 1034–1043. https://doi.org/10.1071/WF14026.

Griscom, H.P., Ashton, M.S., 2011. Restoration of dry tropical forests in Central America: a review of pattern and process. For. Ecol. Manag. 261 (10), 1564–1579. https://doi.org/10.1016/j.foreco.2010.08.027.
Hilderbrand, R.H., Watts, A.C., Randle, A.M., 2005. The myths of restoration ecology. Ecol. Soc. 10 (1). https://doi.org/10.5751/es-01277-100119.
Holl, K.D., 2012. Restoration of tropical forests. In: van Andal, J., Aronson, J. (Eds.), Restoration Ecology: The New Frontier, 2nd ed. Wiley-Blackwell, Hoboken, NJ, pp. 103–114.
Holl, K.D., Aide, T.M., 2011. When and where to actively restore ecosystems? For. Ecol. Manag. 261 (10), 1558–1563. https://doi.org/10.1016/j.foreco.2010.07.004.
Holl, K.D., Loik, M.E., Lin, E.H., Samuels, I.A., 2000. Tropical montane forest restoration in Costa Rica: overcoming barriers to dispersal and establishment. Restor. Ecol. 8 (4), 339349. https://doi.org/10.1046/j.1526-100x.2000.80049.x.
Holl, K.D., Zahawi, R.A., Cole, R.J., Ostertag, R., Cordell, S., 2011. Planting seedlings in tree islands versus plantations as a large-scale tropical forest restoration strategy. Restor. Ecol. 19 (4), 470–479. https://doi.org/10.1111/j.1526-100X.2010.00674.x.
Hooper, E., Condit, R., Legendre, P., 2002. Responses of 20 native tree species to reforestation strategies for abandoned farmland in Panama. Ecol. Appl. 12 (6), 1626–1641. https://doi.org/10.1890/1051-0761(2002)012[1626:RONTST]2.0.CO;2.
Hooper, E., Legendre, P., Condit, R., 2005. Barriers to forest regeneration of deforested and abandoned land in Panama. J. Appl. Ecol. 42 (6), 1165–1174. https://doi.org/10.1111/j.1365-2664.2005.01106.x.
Howe, H.F., Smallwood, J., 1982. Ecology of seed dispersal. Annu. Rev. Ecol. Evol. Syst. 13 (1), 201–228. https://doi.org/10.1146/annurev.es.13.110182.001221.
Jaenicke, J., Wösten, H., Budiman, A., Siegert, F., 2010. Planning hydrological restoration of peatlands in Indonesia to mitigate carbon dioxide emissions. Mitig. Adapt. Strateg. Glob. Chang. 15 (3), 223–239. https://doi.org/10.1007/s11027-010-9214-5.
Jordán, A., Zavala, L.M., Muñoz-Rojas, M., 2011. Mulching, effects on soil physical properties. In: Glinski, J., Horabik, J., Lipiec, J. (Eds.), Encyclopedia of Agrophysics. Springer, Dordrecht.
Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: an overview. Agrofor. Syst. 76 (1), 1–10. https://doi.org/10.1007/s10457-009-9229-7.
Keeley, J.E., 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. Int. J. Wildland Fire 18 (1), 116–126. https://doi.org/10.1071/WF07049.
Kobayashi, S., 2004. Landscape rehabilitation of degraded tropical forest ecosystems: case study of the CIFOR/Japan project in Indonesia and Peru. For. Ecol. Manag. 201 (1), 13–22. https://doi.org/10.1016/j.foreco.2004.06.009.
Labrière, N., Locatelli, B., Laumonier, Y., Freycon, V., Bernoux, M., 2015. Soil erosion in the humid tropics: a systematic quantitative review. Agric. Ecosyst. Environ. 203, 127139. https://doi.org/10.1016/j.agee.2015.01.027.
Lamb, D., Erskine, P.D., Parrotta, J.A., 2005. Restoration of degraded tropical forest landscapes. Science 310 (5754), 1628–1632. https://doi.org/10.1126/science.1111773.
Lampela, M., Jauhiainen, J., Sarkkola, S., Vasander, H., 2017. Promising native tree species for reforestation of degraded tropical peatlands. For. Ecol. Manag. 394, 52–63. https://doi.org/10.1016/j.foreco.2016.12.004.
Lazos-Chavero, E., Zinda, J., Bennett-Curry, A., Balvanera, P., Bloomfield, G., Lindell, C., Negra, C., 2016. Stakeholders and tropical reforestation: challenges, trade-offs, and strategies in dynamic environments. Biotropica 48 (6), 900–914. https://doi.org/10.1111/btp.12391.
Little, K.M., Willoughby, I., Wagner, R.G., Adams, P., Frochot, H., Gava, J., Gous, S., Lautenschlager, R.A., Örlander, G., Sankaran, K.V., Wei, R.P., 2006. Towards reduced herbicide use in forest vegetation management. South. Afr. For. J. 207 (1), 63–79. https://doi.org/10.2989/10295920609505254.
Löf, M., Dey, D.C., Navarro, R.M., Jacobs, D.F., 2012. Mechanical site preparation for forest restoration. New For. 43 (5–6), 825–848. https://doi.org/10.1007/s11056-012-9332-x.
Löf, M., Madsen, P., Metslaid, M., Witzell, J., Jacobs, D.F., 2019. Restoring forests: regeneration and ecosystem function for the future. New For. 50, 139–151. https://doi.org/10.1007/s11056-019-09713-0.
Lohberger, S., Stängel, M., Atwood, E.C., Siegert, F., 2018. Spatial evaluation of Indonesia's 2015 fire-affected area and estimated carbon emissions using Sentinel-1. Glob. Chang. Biol. 24 (2), 644–654. https://doi.org/10.1111/gcb.13841.
Lwanga, J.S., 2003. Forest succession in Kibale National Park, Uganda: implications for forest restoration and management. Afr. J. Ecol. 41 (1), 9–22. https://doi.org/10.1046/j.1365-2028.2003.00400.x.
MacDonald, G.E., 2004. Cogongrass (Imperata cylindrica)—biology, ecology, and management. Crit. Rev. Plant Sci. 23 (5), 367–380. https://doi.org/10.1080/07352680490505114.
Mataix-Solera, J., Guerrero, C., García-Orenes, F., Bárcenas, G.M., Torres, M.P., Barcenas, M., 2009. Forest fire effects on soil microbiology. In: Cerdà, A., Robichaud, P.R. (Eds.), Fire Effects on Soils and Restoration Strategies. vol. 5. Science Publishers, Enfield, NH, pp. 133–176.
Montagnini, F., Eibl, B., Grance, L., Maiocco, D., Nozzi, D., 1997. Enrichment planting in overexploited subtropical forests of the Paranaense region of Misiones, Argentina. For. Ecol. Manag. 99 (1–2), 237–246. https://doi.org/10.1016/S0378-1127(97)00209-0.
Moore, P., Hardesty, J., Kelleher, S., Maginnis, S., Myers, R., 2003. Forests and wildfires: fixing the future by avoiding the past. XII World Forestry Congress Retrieved from. http://www.fao.org/3/xii/0829-b3.htm.
Morgan, P., Moy, M., Droske, C.A., Lewis, S.A., Lentile, L.B., Robichaud, P.R., Hudak, A.T., Williams, C.J., 2015. Vegetation response to burn severity, native grass seeding, and salvage logging. Fire Ecol. 11 (2), 31–58. https://doi.org/10.4996/fireecology.1102031.
Nepstad, D.C., Stickler, C.M., Filho, B.S., Merry, F., 2008. Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. Philos. Trans. R. Soc. B Biol. Sci. 363 (1498), 1737–1746. https://doi.org/10.1098/rstb.2007.0036.
Omeja, P.A., Chapman, C.A., Obua, J., Lwanga, J.S., Jacob, A.L., Wanyama, F., Mugenyi, R., 2011. Intensive tree planting facilitates tropical forest biodiversity and biomass accumulation in Kibale National Park, Uganda. For. Ecol. Manag. 261 (3), 703–709. https://doi.org/10.1016/j.foreco.2010.11.029.
Ondei, S., Prior, L.D., Vigilante, T., Bowman, D.M., 2016. Post-fire resprouting strategies of rainforest and savanna saplings along the rainforest–savanna boundary in the Australian monsoon tropics. Plant Ecol. 217 (6), 711–724. https://doi.org/10.1007/s11258-015-0531-3.
Page, S., Hosciło, A., Wösten, H., Jauhiainen, J., Silvius, M., Rieley, J., Ritzema, H., Tansey, K., Graham, L., Vasander, H., Limin, S., 2009. Restoration ecology of lowland tropical peatlands in Southeast Asia: current knowledge and future research directions. Ecosystems 12 (6), 888–905. https://doi.org/10.1007/s10021-008-9216-2.
Palma, A.C., Laurance, S.G., 2015. A review of the use of direct seeding and seedling plantings in restoration: what do we know and where should we go? Appl. Veg. Sci. 18 (4), 561568. https://doi.org/10.1111/avsc.12173.
Parrotta, J.A., Knowles, O.H., 1999. Restoration of tropical moist forests on bauxite-mined lands in the Brazilian Amazon. Restor. Ecol. 7 (2), 103–116. https://doi.org/10.1046/j.1526-100X.1999.72001.x.
Pereira, L., Oliveira, C., Torezan, J.M.D., 2013. Woody species regeneration in Atlantic Forest restoration sites depends on surrounding landscape. Natureza & Conservação 11 (2), 138–144. https://doi.org/10.4322/natcon.2013.022.
Ritzema, H., Limin, S., Kusin, K., Jauhiainen, J., Wösten, H., 2014. Canal blocking strategies for hydrological restoration of degraded tropical peatlands in Central Kalimantan, Indonesia. Catena 114, 11–20. https://doi.org/10.1016/j.catena.2013.10.009.
Rogers, B.M., Balch, J.K., Goetz, S.J., Lehmann, C.E., Turetsky, M., 2020. Focus on changing fire regimes: interactions with climate, ecosystems, and society. Environ. Res. Lett. 15 (3), 030201. https://doi.org/10.1088/1748-9326/ab6d3a.
Rokich, D.P., Dixon, K.W., Sivasithamparam, K., Meney, K.A., 2000. Topsoil handling and storage effects on woodland restoration in Western Australia. Restor. Ecol. 8 (2), 196208. https://doi.org/10.1046/j.1526-100x.2000.80027.x.
Schweizer, D., Van Kuijk, M., Meli, P., Bernardini, L., Ghazoul, J., 2019. Narratives across scales on barriers and strategies for upscaling forest restoration: a Brazilian case study. Forests 10 (7), 530. https://doi.org/10.3390/f10070530.
Shlisky, A., Alencar, A.A., Nolasco, M.M., Curran, L.M., 2009. Overview: global fire regime conditions, threats, and opportunities for fire management in the tropics. In: Cochrane, M.A. (Ed.), Tropical Fire Ecology: Climate Change, Land Use and Ecosystem Dynamics. Springer, Berlin Heidelberg, pp. 65–83.
Shono, K., Cadaweng, E.A., Durst, P.B., 2007. Application of assisted natural regeneration to restore degraded tropical forestlands. Restor. Ecol. 15 (4), 620–626. https://doi.org/10.1111/j.1526-100X.2007.00274.x.
Slik, J.F., Bernard, C.S., Van Beek, M., Breman, F.C., Eichhorn, K.A., 2008. Tree diversity, composition, forest structure and aboveground biomass dynamics after single and repeated fire in a Bornean rain forest. Oecologia 158 (3), 579. https://doi.org/10.1007/s00442-008-1163-2.
Urgiles, N., Loján, P., Aguirre, N., Blaschke, H., Günter, S., Stimm, B., Kottke, I., 2009. Application of mycorrhizal roots improves growth of tropical tree seedlings in the nursery: a step towards reforestation with native species in the Andes of Ecuador. New For. 38 (3), 229–239. https://doi.org/10.1007/s11056-009-9143-x.
Vallejo, V.R., Arianoutsou, M., Moreira, F., 2012. Fire ecology and post-fire restoration approaches in Southern European forest types. In: Moreira, F., Arianoutsou, M., Corona, P., De las Heras, J. (Eds.), Post-fire Management and Restoration of Southern European Forests. vol. 24. Springer, Dordrecht, pp. 93–119.
Van der Laan, C., Wicke, B., Verweij, P.A., Faaij, A.P., 2017. Mitigation of unwanted direct and indirect land-use change–an integrated approach illustrated for palm oil, pulpwood, rubber and rice production in North and East Kalimantan, Indonesia. GCB Bioenergy 9 (2), 429–444. https://doi.org/10.1111/gcbb.12353.
Van Der Werf, G.R., Randerson, J.T., Giglio, L., Van Leeuwen, T.T., Chen, Y., Rogers, B.M., Mu, M., van Marle, M.J.E., Morton, D.C., Collatz, G.J., Yokelson, R.J., Kasibhatla, P.S., 2017. Global fire emissions estimates during 1997-2016. Earth Syst. Sci. Data 9 (2), 697–720. https://doi.org/10.5194/essd-9-697-2017.
Verma, S., Jayakumar, S., 2015. Post-fire regeneration dynamics of tree species in a tropical dry deciduous forest, Western Ghats, India. For. Ecol. Manag. 341, 75–82. https://doi.org/10.1016/j.foreco.2015.01.005.
Vermunt, D.A., Verweij, P.A., Verburg, R.W., 2020. What hampers implementation of integrated landscape approaches in rural landscapes? Current Landscape Ecology Reports, 1–17. https://doi.org/10.1007/s40823-020-00057-6.
Vieira, D.L., Holl, K.D., Peneireiro, F.M., 2009. Agro-successional restoration as a strategy to facilitate tropical forest recovery. Restor. Ecol. 17 (4), 451–459. https://doi.org/10.1111/j.1526-100X.2009.00570.x.
Watson, S.J., Taylor, R.S., Nimmo, D.G., Kelly, L.T., Clarke, M.F., Bennett, A.F., 2012. The influence of unburnt patches and distance from refuges on post-fire bird communities. Anim. Conserv. 15 (5), 499–507. https://doi.org/10.1111/j.1469-1795.2012.00542.x.
White Paper Science Team, 2015. Vegetation fires and global change. Challenges for concerted international action. A white paper directed to the United Nations and International Organizations. Planet@ risk 3 (1), 45–57 Retrieved from. https://gfmc.online/wp-content/uploads/Planet-at-Risk-Vol-3-No-1-2015-Fire-White-Paper.pdf.
Wilkinson, S.L., Moore, P.A., Flannigan, M.D., Wotton, B.M., Waddington, J.M., 2018. Did enhanced afforestation cause high severity peat burn in the Fort McMurray Horse River wildfire? Environ. Res. Lett. 13 (1), 014018. https://doi.org/10.1088/1748-9326/aaa136.
Wohlgemuth, P.M., Beyers, J.L., Hubbert, K.R., 2009. Rehabilitation strategies after fire: the California, USA experience. In: Cerdà, A., Robichaud, P.R. (Eds.), Fire Effects on Soils and Restoration Strategies. vol. 5. Science Publishers, Enfield, NH, pp. 511–536.

WRI, 2015. The Restoration Diagnostic: A Method for Developing Forest Landscape Restoration Strategies by Rapidly Assessing the Status of Key Success Factors. Washington, DC. Retrieved from https://files.wri.org/s3fs-public/WRI_Restoration_Diagnostic_1.pdf.

Zahawi, R.A., Reid, J.L., Holl, K.D., 2014. Hidden costs of passive restoration. Restor. Ecol. 22 (3), 284–287. https://doi.org/10.1111/rec.12098.

Zarin, D.J., Davidson, E.A., Brondizio, E., Vieira, I.C.G., Sá, T., Feldpausch, T., Schuur, E.A.G., Mesquita, R., Moran, E., Delamonica, P., Ducey, M.J., Hurtt, G.C., Salimon, C., Denich, M., 2005. Legacy of fire slows carbon accumulation in Amazonian forest regrowth. Front. Ecol. Environ. 3 (7), 365–369. https://doi.org/10.1890/1540-9295(2005)003[0365:LOFSCA]2.0.CO;2.