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Reforestation with four native tree species after abandoned gold mining in the Peruvian Amazon

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ABSTRACT

Global demand for gold has led to a massive increase in mining activity around the world. During the last decade, gold mining grew significantly in the Amazon becoming a major driver for land degradation and heavy metal contamination. However, few studies have explored soil degradation, reforestation, and plant mercury accumulation after mining operations. In this study, we established a reforestation field experiment in a gold mined area. We tested the outcome of planting seedlings of four native tree species previously grown in nursery polyethylene bags versus planting bare root seedlings, as well as the effect of three levels of biofertilization on seedling survival and growth. Previous to the experiment, we evaluated the level of soil degradation by comparing physical and chemical soil properties between the mined area and the nearest undisturbed reference forest. One year after planting, we also sampled roots, stems, and leaves of the planted species in order to detect possible mercury (Hg) accumulation in plant tissues. Our results revealed that soil texture becomes disproportionately sandy, while organic matter content and cation exchange capacity were seven- and three-fold lower in the mined area than in the reference forest, respectively. Seedling survivorship and growth varied across planting methods, biofertilization intensity, and species. Even in the bare root planting technique seedling survivorship was highly acceptable (75%) and increased with transplanting (83%) and the addition of biofertilizer (92%). Although seedling growth was improved significantly by the addition of diluted and pure biofertilizer, overall growth was found to be poor. Two individuals – distant from each other – out of a total of 60 sampled, showed traces of total Hg. A stem from *Ceiba* registered 8.52 mg Hg/kg and the roots of an *Erythrina* presented 0.60 mg Hg/kg. Total estimates of reforestation costs ranged between $1662 and $3464 per hectare in year 1. Poor soil fertility, slow species growth rates, and traces of Hg in plant tissues indicate that remediation and restoration in areas degraded by gold mining can be very challenging.

1. Introduction

Since the 2008 global financial crisis, increase in global demand and the price of gold have sustained the expansion of formal and informal gold mining (Swenson et al., 2011; Alvarez-Berríos and Aide, 2015). Worldwide, it is estimated that over 100 million people in more than 50 countries depend on smale-scale gold mining, while 15 million are directly employed in it (UNEP, 2013). Cremers and de Theije (2013) estimated more than 500,000 informal gold miners active in five countries of the Amazon region (Peru, Brazil, Colombia, Surinam, and Bolivia) generating 26% of the total gold production in those countries. Peru is currently the sixth largest gold producing country in the world and the first in Latin America. Since 2001, the region of Madre de Dios in the southwestern Amazon basin produces approximately 10% of the total annual Peruvian gold production (Ministry of Energy and Mines, 2015).

Western Amazonian forests in Madre de Dios, Peru, are one of the highest biodiversity regions on Earth (Gentry, 1988; Asner et al., 2012). Natural protected areas cover more than 6.1 million hectares including three National Parks (Manu, Purus, and Bahuaja-Sonene),

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one National Reserve (Tambopata), and two Communal Reserves (Amarakaeri and Purus) (SERNANP-INEI, 2013). Since early 2000, this region has experienced a rapid expansion in informal and illicit gold mining operations, which have transformed large expanses of primary rainforest into denuded and mercury (Hg)-poisoned wastelands (Asner et al., 2013; Elmes et al., 2014). After informal gold mining, areas are impoverished and present severe limitations for agricultural development and for the recovery of the native forest (Mosquera et al., 2009).

On mined lands, certain extreme soil conditions may occur that prevent plant growth, referring particularly to physical properties, extreme lack of certain nutrients, and levels of toxicity from heavy metals (Bradshaw, 1997). In the absence of natural regeneration and with high levels of soil degradation, active restoration interventions are needed to restart the natural process of forest succession and to develop fully functioning soils (Holl and Aide, 2011). However, the current knowledge of reforestation and remediation techniques and their costs is insufficient for expanding ecological restoration in areas degraded by gold mining. The understanding of soil degradation and the tolerance of native tree species against the extreme conditions of mined areas is still incomplete (Cooke and Johnson, 2002).

Despite the growing importance of gold mining in Amazonian rainforests, reforestation in mined areas has rarely been managed experimentally. It is well known that plant species is an essential factor in determining the success of remediation for Hg-contaminated sites (Wang et al., 2012). However, the number of suitable native tropical tree species that can be used for reforestation or phytoremediation is unknown, while the interaction of plant growth with soil degradation and Hg contamination remains poorly understood. Also, very little attention has been given to the application of microorganisms in a complementary way – together with reforestation – not only to improve biological soil properties and increase fertility, but also to bind, transport, and detoxify Hg from organic Hg species to elemental Hg, thereby preventing food chain bioconcentration (Xu et al., 2015).

As part of a long-term research effort aimed at assessing the restoration potential for gold mined areas in Madre de Dios, this paper presents a detailed study of the initial establishment of four native tree species in an area degraded by informal gold mining. The objectives of this study were to: (1) assess the level of soil degradation after informal mining operations, (2) evaluate the survival and growth of four native tree species using two planting methods and three levels of biofertilization, (3) detect possible accumulation of Hg in plant tissues of planted species, and (4) estimate the cost of reforestation in abandoned gold mined areas.

2. Materials and methods

2.1. Study site

The study was conducted in a reforestation concession located near the Manuani river in a representative mining area known as "La Pampa". This can be found in the buffer zone of the Tambopata National Reserve in Madre de Dios, Peru. The elevation of the area is 220 m a.s.l. There is a seasonal tropical climate, with a mean annual rainfall of 2200–2400 mm. The mean annual temperature is 25 °C, and for 3 months a year (July–September) rainfall averages less than 100 mm (Malhi et al., 2002). Soil drainage and natural fertility are poor, with deficiencies in plant available phosphorus and soil organic matter (IIAP, 2002). The forest-types of the Tambopata region are representative of seasonal tropical moist forests in southwestern Amazonia and are recognized worldwide for their exceptionally high biological diversity (Gentry, 1988; Asner et al., 2012; Orrego and Zevallos, 2014).

Deforestation for informal gold mining is rapidly expanding in Madre de Dios at a rate of 6145 ha yr^{-1} and now exceeds all other forms of forest loss combined, including that of ranching, agriculture, and logging (Asner et al., 2013; Finer and Olexy, 2015). Informal miners in Madre de Dios primarily mine secondary alluvial gold deposits found on riverbanks and areas bordering these. Hydraulic mining machines with pumps are currently the predominant mining method, where gold is recovered by adding Hg to the extracted sediments, which binds the gold particles forming an amalgam (Damonte et al., 2013). The gold–mercury amalgam is then heated in the field releasing Hg vapors to the air and increasing the risk of pollution in the soil, plants, animals, and humans (Diringer et al., 2015).

2.2. Species studied

Four native tropical tree species were selected for the experiment: *Apeiba membranacea* Spruce ex Benth., *Ochroma pyramidale* (Cav. ex Lam.) Urb., *Ceiba pentandra* (L.) Gaertn., and *Erythrina ulei* Harms. Seed availability, ease of propagation, rapid growth in open areas, and wide geographical distribution across tropical America were important criteria for selecting these species for the experiment (Román et al., 2012). The species *Apeiba membranacea* and *Ochroma pyramidale* are, in general, found as pioneer colonizers, light-demanding, fast-growing, short-lived, and softwood species. The other two species *Ceiba pentandra* and *Erythrina ulei* are considered mid-successional or long-lived pioneers, since they are also capable of growing in open areas but generally live longer and grow taller than pioneer short-lived species (Rueger et al., 2011). In general, pioneer species were selected given their tolerance to disturbance and potential to restart secondary succession in deforested areas (Condit et al., 1993). Furthermore, pioneer tree species usually have extensive root systems which represent a desired attribute in phytoremediation for Hg contaminated sites (Xu et al., 2015).

2.3. Silvicultural treatments

Two important silvicultural treatments for plantation forestry were selected for the experiment, one related to the plantation technique, and the other to fertilization. The plantation technique, by bare root or by transplant, has decisive implications for the success of seedling establishment and for the cost of transportation of seedlings from the nursery to the plantation site (Grossnickle and El-Kassabi, 2015). The application of organic biofertilizers in highly degraded areas has the potential, not only to provide nutrients necessary for plant growth in the short-term, but also microorganisms for the recovery of the soil biota in the mid- and long-term (Frouz et al., 2001). Also, different intensities of biofertilizer application may have different effects on species seedling performance, as well as on the maintenance costs of the plantation (Kohler et al., 2014; Young et al., 2015). The biofertilizer used for this experiment was produced according to Restrepo (2001) and contained macro nutrients (N, P, K, Ca, Mg, S), micronutrients (Cu, Zn, Fe, B and Mn), and beneficial soil microorganisms (bacteria, fungus-yeast, and nitrogen fixing bacteria).

2.4. Experimental design

The experiment was initiated at the beginning of the rainy season in December 2013. A total of 1111 seedlings of the four native tree species were planted randomly with a 3×3 m-spacing covering a total area of 1 ha. Approximately 1 kg of organic compost was incorporated in the base of all the holes where seedlings were planted. The arrangement of the experiment includes four 0.25 ha blocks (Fig. 1) each containing three subplots (12 subplots in total)

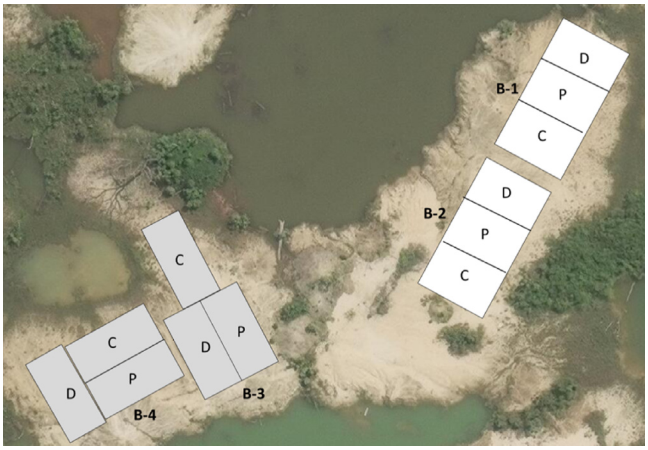

Fig. 1. View of the experimental area including bare-root blocks (white) and transplant blocks (gray). Fertilization subplots are represented with capital letters (C = control, D = diluted, P = pure).

where 93 seedlings of the four native tree species were randomly planted. Seedlings were grown in a full sunlight location during six to eight months and were approximately 35–60 cm tall when planted.

Following a split-plot design, two blocks were planted with bare root seedlings and the other two blocks with seedlings previously grown in nursery black polyethylene bags (Fig. 1). Each block was randomly divided into three subplots representing three fertilization levels: (1) control, without any supply of biofertilizer; (2) diluted, which represents the incorporation of 0.5 L per plant of a diluted biofertilizer (1:10 H_2O); and (3) pure, representing the incorporation of 0.5 L per plant of pure biofertilizer (Fig. 1). The biofertilizer was added to the base of each plant every 15 days during the first 6 months after planting.

2.5. Field measurements

Two weeks before seedling outplanting, soil cores of 31 cm^3 were collected at a depth of 0–20 cm in the midpoint of each of the 12 subplots composing the experimental area. Similarly, 12 soil samples were taken in the adjacent reference forest. Sand, silt, and clay percentages (Bouyoucos), pH (1:1 H_2O), electric conductivity (1:1 H_2O), soil organic matter (Walkley & Black), available phosphorous (Olsen), available potassium (ammonium acetate 1 N pH 7), cation exchange capacity (ammonium acetate 1 N pH 7), and exchangeable cations Ca^{+2}, Mg^{+2}, Na^+, K^+ (ammonium acetate 1 N pH 7), $Al^{+3}H^+$ (Yuan) were determined for each soil sample (Binkley and Fisher, 2013).

The number of live individuals, seedling height (cm), and basal stem diameter (cm) were assessed 7–8 days and 12 months after planting. Diameter was measured with calipers at the stem base and plant height taken with a measuring tape.

One year after seedling outplanting, five live individuals per each of the 12 subplots (60 in total) were randomly sampled to detect possible plant Hg accumulation. The number of individuals sampled from each species per subplot was determined in proportion to the survival percentage of the four species in each subplot. Leaves, stems, and roots of each sampled individual were dried and milled separately for subsequent total Hg analysis through inductive coupled plasma-atomic emission spectrometry (Martin et al., 1994).

Operational costs, materials, and labor requirements were recorded for activities related to seedling propagation (tree nursery bags, seed recollection, substrate, nursery care), plantation labor (site preparation, transportation of seedlings, transplantation), maintenance (fertilization), and monitoring (survivorship, growth, and plant Hg analysis). This data was used to estimate the total plantation cost for each planting method and fertilization treatment on a per-hectare basis.

2.6. Statistical analysis

Comparison of soil properties between the experimental area and the adjacent reference forest was made using one-way analysis of variance (ANOVA). Percentage survival was calculated for each species in each subplot as the percentage of initially planted seedlings still alive 12 months after planting. Diameter and height growth rates (cm/month) were calculated for all surviving individuals to minimize variation in the initial height between individuals and species (Table 1). Effects of planting method and organic fertilization on survivorship and growth rates, as well as plant Hg accumulation, were examined using a split plot design ANOVA (Scheiner and Gurevictch, 2001). Where differences were significant, a post hoc Tukey multiple comparison procedure was performed. The Shapiro–Wilk test was used to assess the normality of the response variables (Fry, 1993). Only survival proportions were arcsin transformed prior to the analysis; height and basal diameter growth showed a normal distribution already and were not transformed.

3. Results

3.1. Soil disturbance

Statistically significant differences were detected in 10 of the 14 soil properties analyzed between the mined area and the contiguous reference forest (Table 1). These results revealed that the soil after informal mining operations had 1.7 times more sand, 4.9 times less silt, and 2.3 times less clay, in comparison to the soil of the adjacent reference forest. Similarly, soil pH was 1.2 times higher in the mined area used for the experiment, while soil organic matter and cation exchange capacity were 7.5 times and 3.2 times higher in the surrounding reference forest soil, respectively. Exchangeable cations such as K^+, Na^+, and $Al^{+3}H^+$ were higher in reference forest soil, while only Mg^{+2} was higher in the mined area (Table 1).

3.2. Seedling survivorship, growth, and mercury accumulation

Overall seedling survivorship and diameter growth rates one year after planting showed statistically significant differences according to the effect of the planting method, fertilization, and the interaction, while height growth rates were significantly different as a result of the effect of fertilization and the interaction between the planting method and fertilization (Fig. 2).

Measurements across treatments revealed higher survivorship and diameter growth rates in the transplant than in the bare root planting method, while fertilization improved survivorship and diameter and height growth rates. The supply of pure biofertilizer increased survivorship, and diameter and height growth rates both within the bare root-control and the transplant-control treatments (Fig. 2).

Survivorship of all four species was affected significantly by the planting method, fertilization, and their interaction (Fig. 3). Mortality of *Ochroma* seedlings was 100% in the bare root treatment, while its survivorship increased in the transplant by adding diluted or pure biofertlizer. Survivorship of *Apeiba* and *Ceiba* also increased by adding pure biofertilizer, both within the bare root and the transplant techniques. Survivorship of *Erythrina* was improved by adding pure biofertilizer but only within the transplant (Fig. 3).

Table 1
Comparison of soil properties at 0–20 cm depth (mean ± SE) between the area of the experiment and the contiguous reference forest (ANOVA, $n = 12$).

Parameter	Abandoned	Reference	Sig.[a]	Optimal for plant growth[b]
Physical				
Sand (%)	87.3 ± 1.6	52.8 ± 1.8	**	
Silt (%)	7.0 ± 0.6	34.0 ± 1.4	**	
Clay (%)	5.8 ± 1.3	13.2 ± 2.1	*	
Chemical				
pH (1:1)	4.33 ± 0.05	3.69 ± 0.04	**	5–8
EC (dS/m)	0.08 ± 0.01	0.09 ± 0.01	ns	<2
SOM (%)	0.25 ± 0.02	1.87 ± 1.19	**	>2
P (ppm)	2.8 ± 0.3	3.7 ± 0.5	ns	>7
K (ppm)	237.9 ± 22.7	247.1 ± 46.4	ns	>100
Cation exchange capacity (cmolc/kg)	2.6 ± 0.2	8.4 ± 0.4	**	>6
Exchangeable cations (cmolc/kg)				
Ca^{+2}	0.65 ± 0.04	0.64 ± 0.04	ns	
Mg^{+2}	0.37 ± 0.06	0.16 ± 0.01	*	
K^{+}	0.13 ± 0.01	0.29 ± 0.01	**	
Na^{+}	0.11 ± 0.01	0.16 ± 0.01	*	
$Al^{+3}H^{+}$	0.3 ± 0.04	2.48 ± 0.12	**	

[a] ns (non-significant), * ($P < 0.05$), ** ($P < 0.001$).
[b] Binkley and Fisher (2013).

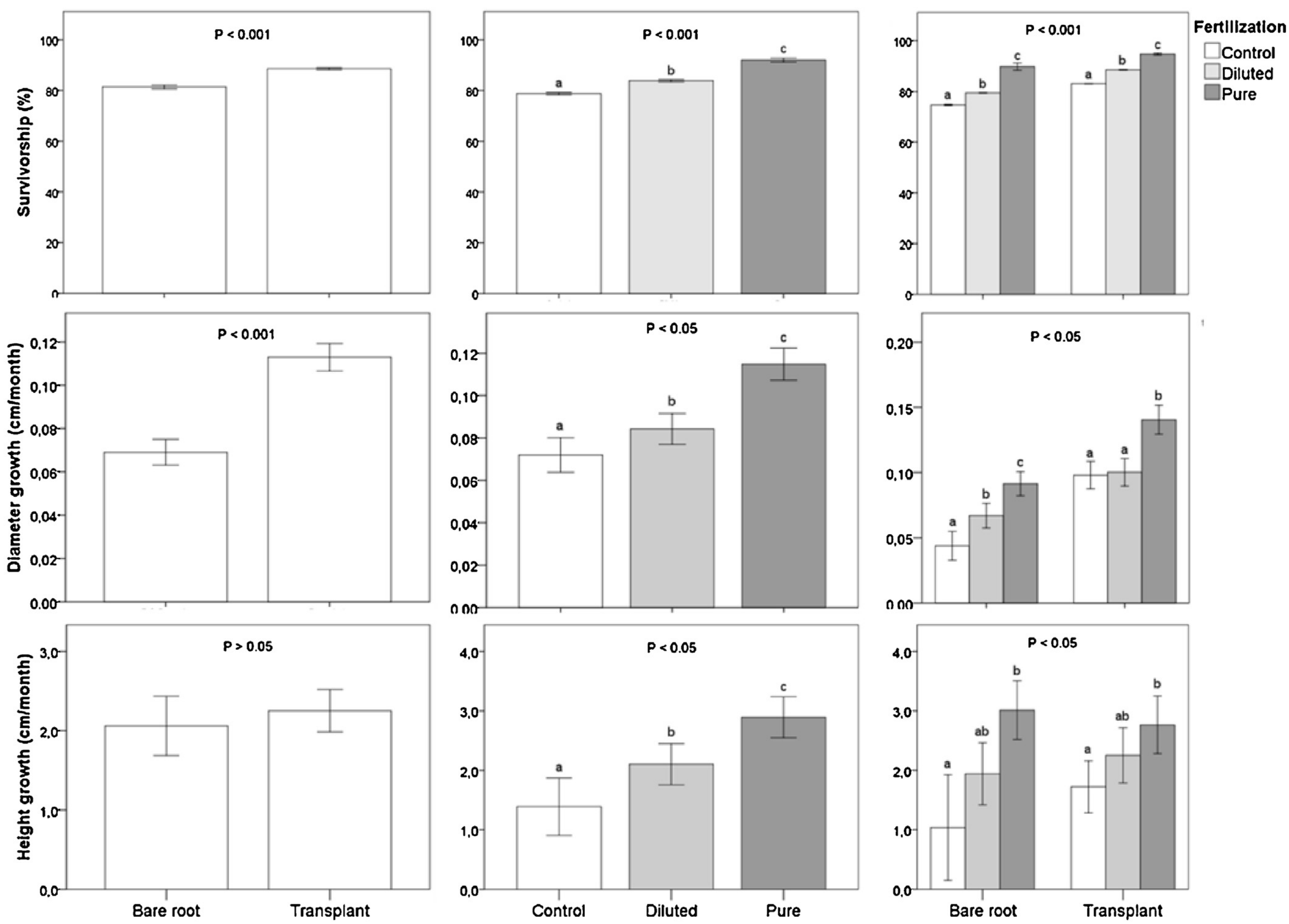

Fig. 2. Treatment effects on the overall survivorship, and diameter and height growth rates. Different letters above error bars mean statistically significant differences (ANOVA, Tukey test, $P < 0.05$).

Diameter and height growth rates of *Ceiba* seedlings were enhanced by adding diluted or pure biofertilizer, both within the bare root and the transplant planting methods. Also, diameter growth rates of *Apeiba* and *Erythrina* seedlings were increased in the bare root by adding diluted or pure biofertilizer (Fig. 3). In general, *Ochroma*, *Ceiba*, and *Erythrina* seedlings reached the highest diameter growth rates, especially in the transplant, while *Ochroma* and *Ceiba* seedlings registered the highest height growth rates. In contrast, the lowest height growth rates were recorded by *Apeiba* and *Erythrina* seedlings, and *Apeiba* seedlings showed the lowest diameter growth rates (Fig. 3).

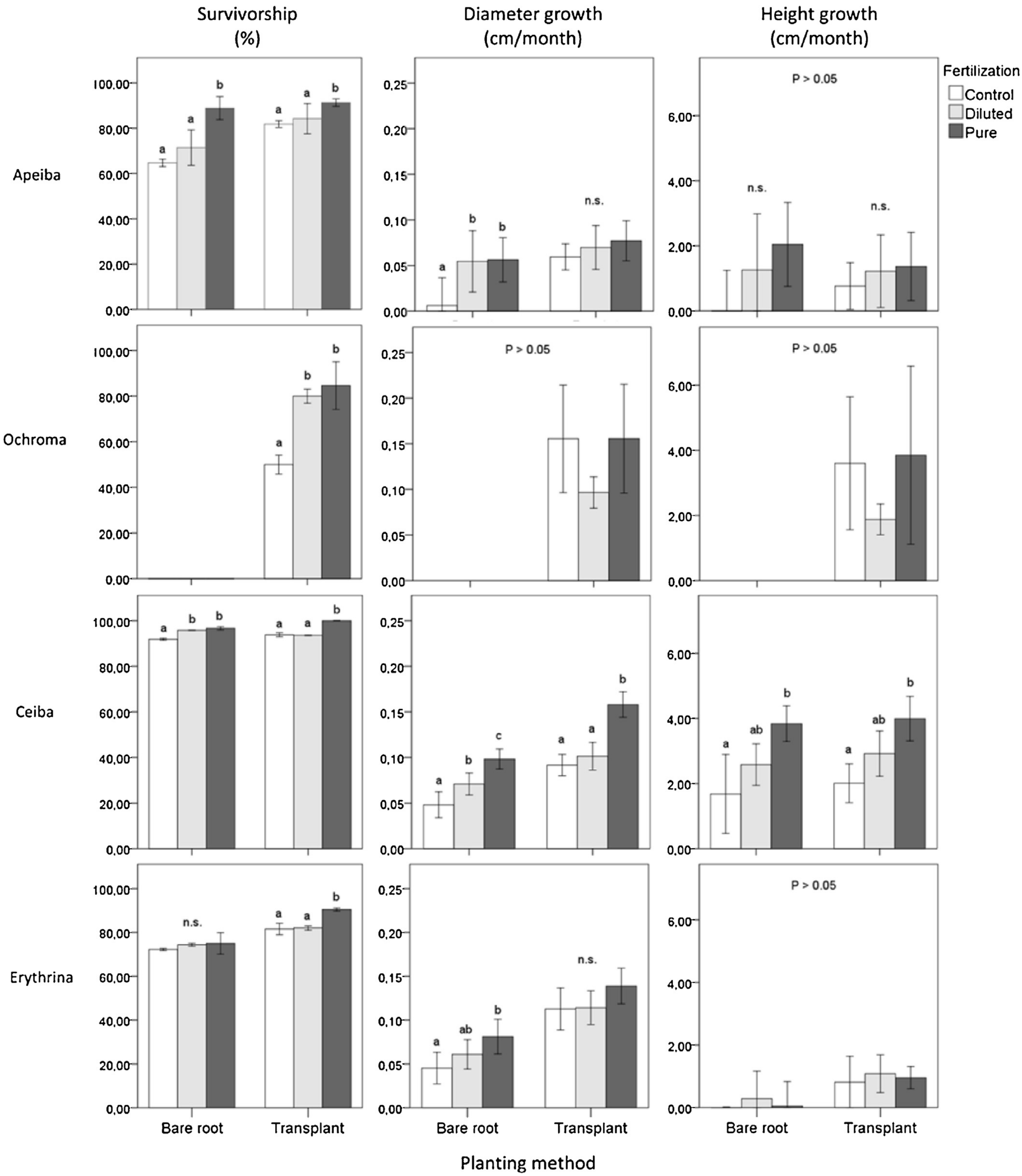

Fig. 3. Survivorship and growth in basal diameter and height of four native tree species across two planting methods and three levels of biofertilization, 12 months after planting in an area degraded by informal gold mining. Different letters above error bars mean statistical differences (ANOVA, Tukey test, $P < 0.05$).

No statistically significant differences were detected in plant Hg accumulation between treatments. However, 2 of the 60 individuals sampled (3.3%) evidenced traces of total Hg in plant tissues. A stem sampled from *Ceiba* registered 8.52 mg Hg/kg in a bare root-diluted treatment, while the roots of an *Erythrina* showed 0.60 mg Hg/kg in a transplant-diluted treatment.

3.3. Costs estimates

The cost of producing seedlings is 40% higher in the transplant than in the bare root planting method because of the need for nursery bags and a greater amount of substrate (Fig. 4). Similarly, the cost of plantation is 40% more expensive in the transplant than in

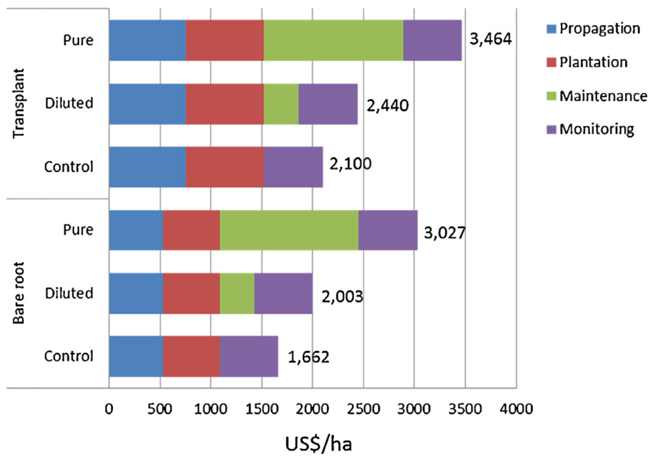

Fig. 4. Cost of reforestation in year 1 in an area degraded by informal gold mining using two planting methods and three levels of biofertilization. Colors represent different stages of the reforestation process. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

the bare root due to increased labor and the time needed for digging the holes, packaging and carrying the seedlings, and for their installation in the field (Fig. 4).

According to the treatments and items considered in this study, the cost of reforesting an area degraded by mining can vary from 1600–3000 US$/ha within the bare root planting method, to 2100–3500 US$/ha in the transplant (Fig. 4). On average, propagation of seedlings represented 27% of the total reforestation cost, plantation 28%, maintenance 19%, and monitoring 25%. However, the cost of maintenance could increase to 39–45% of the total reforestation cost in treatments that involve the addition of pure biofertilizer (Fig. 4).

4. Discussion

4.1. Soil disturbance and plant growth

Our results demonstrate that informal gold mining is an important agent of disturbance and soil degradation in tropical forest ecosystems. After mining, the soil loses structure and fertility through the increased proportion of sand (lacking silt and clay particles) and deficiency in the organic matter content and cation exchange capacity. This causes low water and nutrient holding capacity, thereby decreasing soil fertility to levels insufficient to support normal plant growth (Binkley and Fisher, 2013). Similar percentages of sand, low content of soil organic matter, and poor cation exchange capacity were found after gold mining in Guyana (Burnett, 2013), Australia (Banning et al., 2008), and Canada (Young et al., 2015). In addition, decreased soil microbial populations have been found closely correlated with soil organic matter depletion in mined areas in India (Ghose, 2004) and in French Guiana (Schimann et al., 2012).

The high level of soil degradation in gold mined areas can be explained by the process of mineral extraction. Although different techniques for the extraction of gold are used in Madre de Dios, nearly all utilize vast amounts of water which is usually jetted onto the soil surfaces at a very high pressure, causing disaggregation of the soil particles (Damonte et al., 2013). As described in Peterson and Heemskerk (2001), the resulting gold-bearing slurry is then pumped into a sluice box, which collects gold particles, while mine tailings flow into either an abandoned mining pit or the adjacent forest. As re-sedimentation takes place, the various size fractions settle in separate horizons of varying depths, giving rise to new textural classifications and different soil characteristics (Bradshaw, 1997; Burnett, 2013).

Although seedling survivorship was highly acceptable, species plant growth was found to be poor when compared to other studies which planted the same species in areas with lower levels of soil disturbance (i.e. agriculture, cattle ranching). For example, plants of *Ochroma pyramidale* grew up to 6 m in height and 8 cm in diameter one year after planting in tropical abandoned agricultural areas in Mexico (Douterlungne et al., 2010) and Panama (Breugel et al., 2011). In our study, plants of this species were only 77 cm in height and 2.4 cm in diameter after the same time period. Similarly, Román-Dañobeytia et al. (2012a) reported plants of *Ceiba pentandra* about 2.2 m in height and 4 cm in diameter 18 months after planting in abandoned cattle pastures in Mexico. In our study, plants of this species reached 98 cm in height and 2.7 cm in basal diameter one year after planting. These differences in growth can be explained by the extreme alteration of the physical, chemical, and biological properties of the soil after mining, as well as by the recognized difficulty for pioneer tree species to grow fast in low fertility soils (Paul et al., 2010; Martínez-Garza et al., 2013). Therefore, a slower rate of ecosystem recovery could be expected in areas degraded by mining in comparison to areas previously used for agriculture or cattle ranching.

4.2. Species responses to silvicultural treatments

In our study, plants of *Ochroma* did not resist the bare root planting method, while the plants of *Erythrina* and *Apeiba* suffered the desiccation of the main stem in such treatment (especially in the control); however, many of the desiccated plants survived and resprouted from the base. For this reason, height growth was very low in the latter two species in the bare root-control treatment. Although *Ceiba* seedlings well tolerated bare root planting, the transplant technique maximized its growth and survivorship. It is well known that bare root seedlings suffer more stress and therefore require good soil fertility in the areas to be planted, while seedlings produced in nursery bags can better tolerate stress and adapt to poorer soils (McKay and Morgan, 2001). Planting stress can lead to root growth being limited by the lack of water and photosynthates, which may in turn limit photosynthesis. Thus, a newly planted seedling's ability to overcome planting stress is affected by its root system size and distribution, root–soil contact, and root hydraulic conductivity (Grossnickle, 2005). Nonetheless, more studies are required to test other planting methods (i.e. stakes, containers, biodegradable pots, root trainers) to provide an adequate understanding of how seedlings physiologically respond to the post mining environment after being planted in the field.

Our results showed that diluted and pure biofertilizer amendments can increase plant growth, allowing for the successful establishment of the trees planted. The soil left by informal mining operations at our site was characterized as having low levels of fertility sufficient to negatively affect plant growth (Binkley and Fisher, 2013), similar to other mined areas in the study region (Mosquera et al., 2009; Garate, 2011). In a similar substrate (sandy texture with low organic matter content and poor cation exchange capacity), Young et al. (2015) showed that low levels of organic amendments improved soil fertility and plant cover on old mine tailings. This means that organic matter provides a source of soil biota including bacteria, fungi as well as invertebrates capable of mineralizing the organic matter into plant available nutrients (Frouz et al., 2001; Banning et al., 2008). Furthermore, other studies have reported that addition of compost and microorganisms not only increased soil fertility and plant biomass, but also reduced the concentration of trace elements in plant species growing on metal-contaminated mine soils (Martínez-Fernández et al., 2014; Kohler et al., 2014).

4.3. Mercury accumulation in plant tissues

Our results indicate that Hg could be located in hotspots within mined areas and that the registered plant concentrations, especially in *Ceiba*, can be dangerous for biodiversity and public health. In this regard, Beauford et al. (1977) reported that plants were found to tolerate an external level of 1 mg Hg/kg, but both physiological and biochemical processes were affected between 5 and 10 mg Hg/kg. However, tolerance to Hg-contaminated soils and plant Hg accumulation could vary importantly depending on species, exposure time, and contamination level. In a gold mined Hg-contaminated soil in Colombia, Marrugo-Negrete et al. (2015) showed that growth and development of *Jatropha curcas* plants occurred successfully despite the presence of high amounts of Hg (up to 10 mg Hg/kg) in the soil. Also, the highest Hg concentrations in that study were accumulated after 3–4 months of exposure, mainly in the roots followed by the leaves and stems. Given the few field experiments and data published on this issue in tropical America (Wang et al., 2012; Xu et al., 2015), further studies could be focused on assessing soil contamination, translocation, and bioconcentration of Hg, and other heavy metals (i.e. cadmium, lead, arsenic), in different native tree species at different exposure time periods.

4.4. Reforestation costs

In our study, costs increased with producing, transporting, and transplanting nursery bag seedlings (instead of bare root seedlings), and with the intensity of fertilization. The transport of nursery bag seedlings and application of pure biofertilizer in reforesting abandoned mines that have limited access and infrastructure may be prohibitively expensive for some types of small-scale mining operations. There are also concerns about managing the biomass that could accumulate Hg which may demand more efforts and thereby increase the total reforestation cost. Therefore, it is necessary to perform a cost-benefit analysis to assess the viability of restoration interventions considering different scales of gold mining operations and the instability of gold prices.

Costs registered in this study are reasonable when compared to other reforestation studies in tropical America. In reforesting *Saccharum spontaneum* grasslands in the Panama Canal Watershed, Craven et al. (2009) revealed that total plantation costs for two timber species averaged \$1590–2570 ha^{-1} in year 1 using a 3×3-m spacing grid. Román-Dañobeytia et al. (2012b) showed that reforestation costs in year 1 of tropical abandoned cattle pastures in Mexico ranged from \$1260 to \$1820 ha^{-1} on a 2×2-m spacing grid (2500 trees ha^{-1}). In the Brazilian Atlantic Forest, Rodrigues et al. (2009) reported that costs of high species diversity plantations ranged from \$3000 to \$4500 ha^{-1} with a planting density of 2×3-m (1666 trees ha^{-1}).

In 2014 alone informal gold mining in Madre de Dios produced 258,000 gold troy ounces with an estimated value of \$326.6 million (Ministry of Energy and Mines, 2015). However, degraded areas by informal gold mining in Madre de Dios are currently abandoned, some regenerating slowly while others are highly degraded and without potential to regenerate naturally. Therefore, conducting a real and serious formalization process together with the production of technical information on reforestation and remediation is critical for allowing miners to meet with their environmental responsibilities and to support the practice of ecological restoration.

5. Conclusions

This study demonstrated the potential of four native tree species and different silvicultural treatments to reforest areas abandoned by gold mining. The transplantation technique was important for maximizing seedling survivorship, especially for *Ochroma*, as well as for the initial growth of all four species. The bare root proved to be a low cost and effective treatment for establishing *Ceiba* seedlings. The application of diluted (55 L/ha) or pure (555 L/ha) biofertilizer was found to be useful in maintaining good seedling development across time, particularly in *Apeiba*, *Ceiba* and *Erythrina*. In general, the transplant was found to be more critical and cost-beneficial for successful seedling establishment than the fertilization. However, the decisions on how to produce, install, and maintain restoration plantations will depend on the tolerances of the target species to extremely disturbed soils and on the availability of resources for applying appropriate silvicultural treatments. Poor soil fertility, species slow growth rates, and traces of mercury in plant tissues indicate that remediation and restoration in areas degraded by gold mining can be very challenging. More experimental reforestation and remediation studies are needed to improve the science and practice of forest restoration in gold mined areas.

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