

Article

Soil and Water Bioengineering for Riparian Restoration: Species Performance, Establishment Dynamics and Ecosystem Responses in Tropical River Systems

Paula Letícia Wolff Kettenhuber ^{1,*}, Sebastião Venâncio Martins ¹, Fagner Darlan Dias Corrêa ¹, Maria da Costa Cardoso ¹, Diego Aniceto dos Santos Oliveira ² and Enzo Mauro Fioresi ¹

¹ Forest Restoration Laboratory, Department of Forest Engineering, Universidade Federal de Viçosa, Viçosa 36570-900, Brazil; venancio@ufv.br (S.V.M.); fagner.correa@ufv.br (F.D.D.C.); maria.c.cardoso@ufv.br (M.d.C.C.); enzo.fioresi@ufv.br (E.M.F.)

² Biotic Environment Management, Vale S.A., Nova Lima 34019-899, Brazil; diego.aniceto@vale.com

* Correspondence: paula.kettenhuber@ufv.br

Abstract

Soil and water bioengineering (SWBE) is increasingly used as a nature-based solution for riverbank stabilization and riparian restoration, yet its effectiveness in tropical environments remains constrained by limited field-based evidence of species performance under hydrological disturbance. This study evaluated the establishment success and ecological effectiveness of four native riparian species (*Croton urucurana* Baill., *Sesbania virgata* (Cav.) Pers., *Lochroma arborescens* (L.) J.M.H.Shaw, and *Gymnanthes schottiana* Müll.Arg.) installed as live cuttings on a riprap structure exposed to recurrent flooding along the Paraopeba River, Brazil. A total of 160 live cuttings were monitored over a 33-month establishment period to assess survival, structural development, spontaneous vegetation recruitment, and changes in soil chemical properties and soil organic carbon stocks. Flooding acted as a dominant ecological filter, causing substantial early mortality, with overall survival declining sharply during a 70-day inundation period that included 58 consecutive days of submergence. *Croton urucurana* exhibited the highest survival and structural development, reaching median heights exceeding 5 m and cumulative shoot diameters greater than 100 mm after 33 months, whereas *Gymnanthes schottiana* showed complete mortality within the first year. Vegetation establishment facilitated spontaneous recruitment of native woody species, with 22 individuals recorded in planted sections compared to only 3 in adjacent non-planted areas. Soil organic carbon stocks increased from 38.9 to 60.6 Mg C ha⁻¹ in the 0–40 cm soil profile, indicating rapid soil development. These results demonstrate that SWBE interventions can simultaneously promote riverbank stabilization, vegetation recovery, and soil carbon accumulation. By providing quantitative field-based evidence under realistic hydrological disturbance conditions, this study advances the understanding of species selection and the ecological effectiveness of SWBE interventions in tropical riparian ecosystems.

Keywords: nature-based solutions; live cuttings; functional traits; flood disturbance; riparian ecology; ecosystem services

Academic Editor: Anna De Marco

Received: 27 January 2026

Revised: 16 February 2026

Accepted: 25 February 2026

Published: 28 February 2026

Copyright: © 2026 by the authors.

Licensee MDPI, Basel, Switzerland.

This article is an open access article distributed under the terms and conditions of the [Creative Commons Attribution \(CC BY\)](https://creativecommons.org/licenses/by/4.0/) license.

1. Introduction

Riparian zones are critical transitional ecosystems linking terrestrial and aquatic environments and provide essential regulating ecosystem services, including riverbank stabilization, sediment and nutrient retention, flood mitigation, water quality protection, and carbon sequestration, while supporting high levels of biodiversity and landscape connectivity [1,2]. Despite their ecological and societal importance, riparian ecosystems have been increasingly degraded worldwide because of land-use change, river regulation and channel modification, mining activities, and the intensification of extreme hydrological events, which can disrupt fluvial processes and compromise ecosystem resilience [3,4]. In addition, in tropical river systems, climate change is expected to further amplify flood magnitude and frequency, intensify rainfall extremes, and increase erosion and sediment redistribution along riverbanks, thereby exacerbating vegetation loss and geomorphological instability and increasing the need for effective restoration and stabilization strategies capable of simultaneously addressing structural and ecological objectives [5,6]. This challenge has stimulated increasing interest in nature-based stabilization approaches capable of integrating engineering performance with ecological recovery at local and global scales [7–9].

Soil and water bioengineering (SWBE) is a nature-based solution that combines vegetation with inert materials to reduce erosion while simultaneously restoring ecological functions, offering advantages over conventional rigid structures by promoting geomorphological adjustment, vegetation succession, and ecosystem service provision [8,10]. SWBE has been successfully applied across a wide range of climatic and geomorphological contexts, including temperate regions of Europe and North America as well as tropical environments [11–17], demonstrating its potential to simultaneously achieve engineering stabilization and ecological restoration objectives [11,18,19]. However, the effectiveness of SWBE interventions depends strongly on species-specific performance, rooting capacity, and tolerance to site-specific hydrological disturbance, which vary considerably among species and environmental conditions [20]. In highly dynamic tropical river systems, where recurrent flooding and sediment redistribution impose strong ecological filters [19,21,22], uncertainties remain regarding the field performance and establishment success of native species used in bioengineering interventions.

Plant species performance represents a critical determinant of SWBE effectiveness. Successful establishment depends on the ability of species to rapidly develop root systems capable of providing mechanical anchorage and resistance to hydraulic disturbance [23,24]. Functional traits such as rooting capacity, growth rate, vegetative propagation ability, and tolerance to flooding play key roles in determining establishment success and long-term stability [25,26]. While numerous studies have evaluated species suitability for bioengineering applications in temperate regions [27], critical knowledge gaps persist regarding the field performance of tropical riparian species, particularly under highly dynamic hydrological conditions characterized by recurrent flooding, sediment transport, and fluctuating water levels.

Previous research in Brazil and other tropical regions has investigated vegetative propagation and early establishment of native riparian species under controlled or greenhouse conditions, identifying species with promising rooting capacity and growth potential [28,29]. However, a critical gap remains between controlled experimental findings and actual field performance under natural disturbance regimes. Species that demonstrate successful vegetative propagation under controlled conditions may exhibit reduced establishment success in the field due to hydrodynamic disturbance, sediment burial, prolonged submergence, and other environmental stressors [22,30]. This limitation represents a major source of uncertainty for the effective application of SWBE in tropical riparian

environments, where extreme hydrological variability can strongly influence plant establishment, structural development, and long-term persistence.

In addition to structural stabilization, SWBE interventions have the potential to deliver multiple ecosystem services by promoting vegetation establishment, facilitating spontaneous recruitment of native species, and enhancing soil development processes [21,31,32]. These processes contribute to riparian habitat recovery, improved soil quality, increased carbon storage, and enhanced ecosystem resilience [33]. However, field-based evidence quantifying these ecological responses under real hydrological disturbance conditions remains limited, particularly in tropical river systems and engineered riprap structures commonly used for riverbank stabilization [12].

The effectiveness of SWBE interventions can therefore be evaluated based on multiple performance indicators, including plant survival, growth and structural development, resistance to hydrological disturbance, spontaneous recruitment of native species, and improvements in soil properties associated with ecological recovery [11,20,34]. Understanding how species functional traits influence these performance metrics under real disturbance regimes is essential for improving species selection and optimizing bioengineering design in tropical environments.

In this context, this study aimed to evaluate the establishment success and ecological effectiveness of native riparian species installed as live cuttings in a riprap structure exposed to recurrent flooding. Specifically, we (i) evaluated species survival and growth dynamics in response to recurrent flooding on an instrumented riprap structure simulating a common bioengineering scenario; (ii) quantified spontaneous vegetation recruitment and assessed changes in soil chemical properties associated with vegetation establishment; and (iii) advanced the understanding of SWBE's potential to deliver multiple ecosystem services, including bank stabilization through plant structural development and riparian habitat recovery through vegetation recruitment and soil enhancement. By providing field-based evidence of species performance, ecological responses, and soil development under realistic hydrological disturbance conditions, this study addresses a critical knowledge gap and contributes practical guidance for improving species selection and implementation strategies for soil and water bioengineering interventions in tropical riparian ecosystems.

2. Materials and Methods

2.1. Study Area

The experimental site is located in a reach of the Paraopeba River (20°09'30.11" S, 44°09'30.51" W), in the municipality of Brumadinho, Minas Gerais, southeastern Brazil (Figure 1). This river reach was directly impacted by the collapse of the Córrego do Feijão iron ore tailings dam in January 2019, which released approximately 12 million m³ of mining tailings into the Paraopeba River channel. In this affected river segment, tailings deposition caused burial of riparian zones, channel modification, loss of native vegetation, and pronounced riverbank instability [30,31].

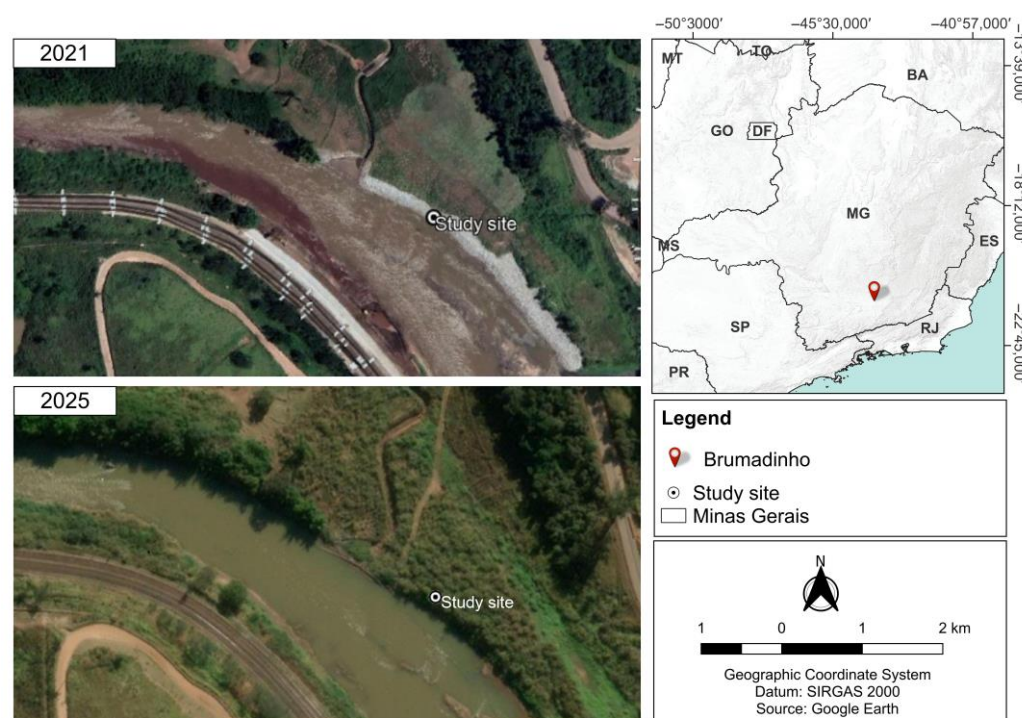


Figure 1. Location of the field experiment along the Paraopeba River in Brumadinho, Minas Gerais, Brazil, illustrating the riprap structure used for riverbank stabilization and the area where live cuttings of native riparian species were planted.

In the years following the disaster, emergency engineering measures were implemented to stabilize riverbanks and reduce erosion and sediment remobilization. At the study reach, a riprap structure was constructed along the impacted river margin, extending for approximately 215 m. The structure was built using basalt rock blocks with dimensions of approximately 0.4–1.0 m and presents an estimated slope of approximately 30°. The underlying soil beneath the riprap was characterized by particle-size analysis, showing a predominance of sand and silt fractions, with clay contents of approximately 15% in the 0–20 cm layer and 11% in the 20–40 cm layer, corresponding to loam to sandy-loam textural classes. While effective in providing immediate mechanical stabilization, such hard engineering solutions offer limited ecological functionality and do not promote the recovery of riparian vegetation or ecosystem processes. In this context, SWBE has been proposed as a complementary nature-based solution to enhance the ecological performance of engineered riverbanks, integrating structural stabilization with vegetation establishment, sediment retention and ecosystem service recovery.

The regional climate is classified as humid subtropical, characterized by dry winters and hot summers, according to the Köppen classification [35]. Monthly precipitation and air temperature during the monitoring period (July 2022 to May 2025) are presented in Figure 2.

River discharge dynamics and river stage were analyzed over the monitoring period to characterize the hydrological regime at the study reach, including the magnitude and persistence of high-flow conditions. Daily discharge and river stage records were obtained from the hydrometric station located adjacent to the experimental site and operated by the Geological Survey of Brazil (CPRM—Companhia de Pesquisa de Recursos Minerais). An operational inundation stage threshold was defined for the experimental riprap planting zone based on field observations and is indicated in Figure 3 as a dashed line.

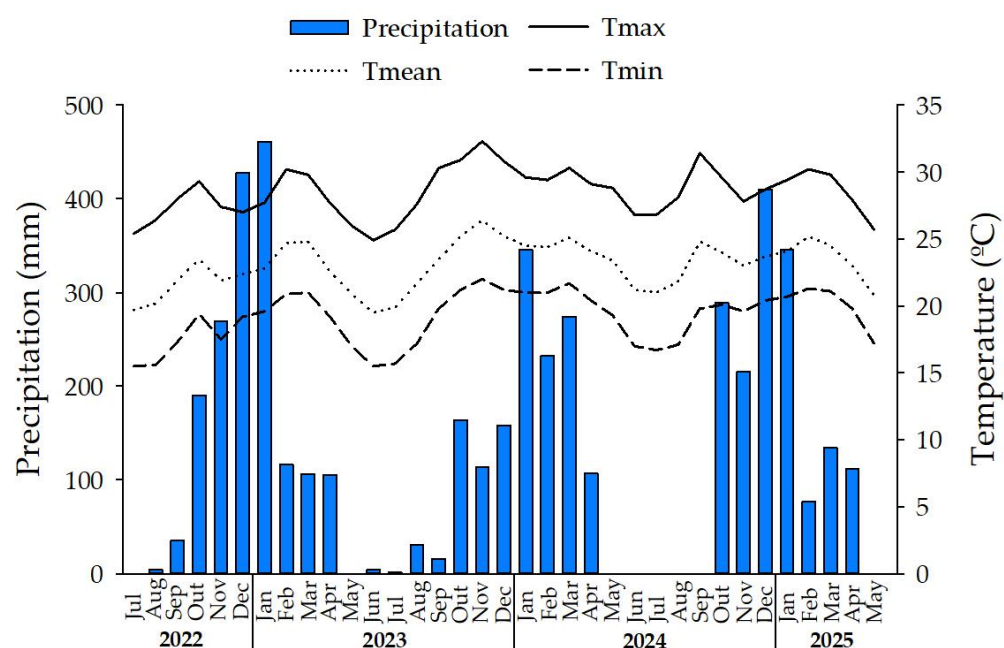


Figure 2. Monthly accumulated precipitation and maximum, mean, and minimum air temperatures recorded in Brumadinho, Minas Gerais, Brazil, from July 2022 to May 2025. Source: [36].

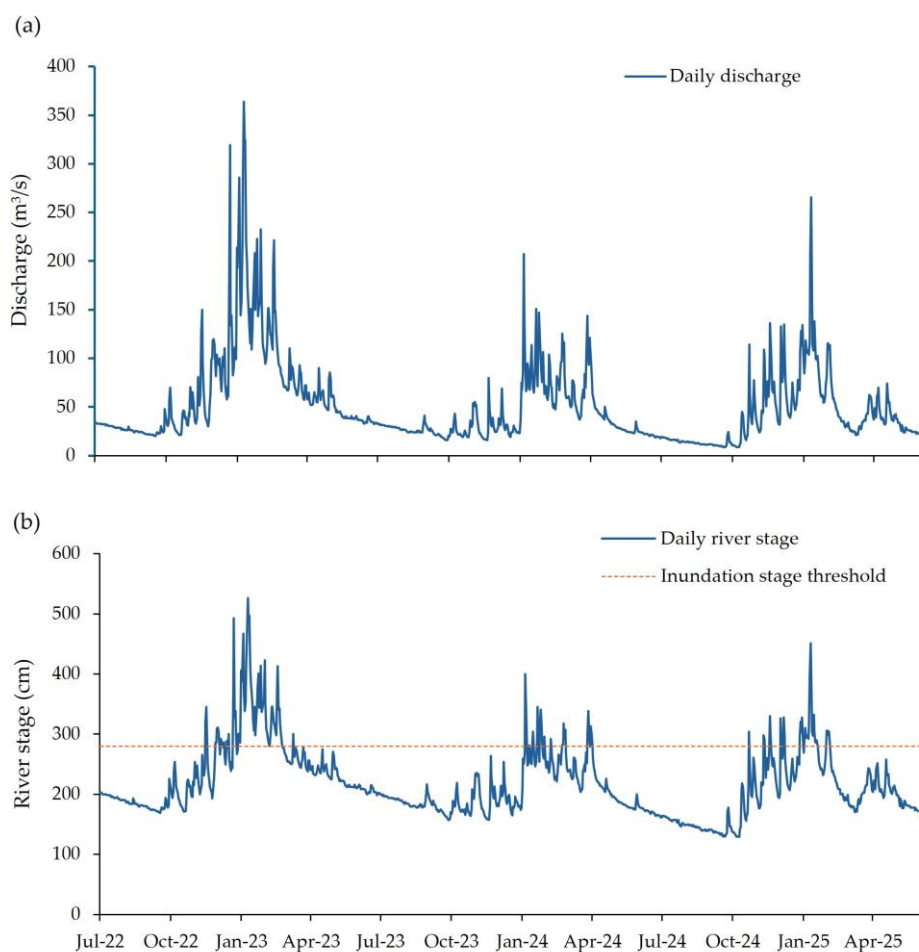


Figure 3. Hydrological conditions at the study reach from July 2022 to May 2025. (a) Daily river discharge ($\text{m}^3 \text{s}^{-1}$). (b) Daily river stage (cm). The dashed line indicates the estimated inundation stage threshold for the experimental riprap planting zone, defined based on field observations.

2.2. Species Selection and Cutting Collection

Four native riparian species from the Paraopeba River basin were selected (Table 1) based on three complementary criteria: (i) rooting and resprouting performance of live cuttings reported in a previous greenhouse experiment [21]; (ii) ecological and functional information from the literature, including rapid establishment, hardy growth strategies, tolerance to flooding and sediment burial, drought resistance, and functional traits relevant to riparian restoration; and (iii) local availability of sufficient donor material to obtain standardized cuttings for field installation.

Table 1. Species used in the field live-staking experiment and main characteristics of the cuttings. Ecological trait codes compiled from literature sources.

Species	Species Abbreviation	Family	Growth Form	Mean Cutting Basal Diameter (mm) $\pm$ SD	Ecological Traits*
<i>Croton urucurana</i> Baill.	Cu	Euphorbiaceae	tree	13.8 $\pm$ 2.4	FT, BT, DR, WF
<i>Gymnanthes schottiana</i> Müll.Arg.	Gs	Euphorbiaceae	shrub	10.1 $\pm$ 2.0	FT, BT, DR
<i>Ichroma arborescens</i> (L.) Schltdl.	Ia	Solanaceae	shrub	12.9 $\pm$ 3.0	FT, BT, DR, WF
<i>Sesbania virgata</i> (Cav.) Poir.	Sv	Fabaceae	shrub	11.5 $\pm$ 2.0	FT, BT, DR, NF

* Ecological trait codes: FT = flood tolerance; BT = burial tolerance; DR = drought resistance; WF = wildlife food provision; NF = biological nitrogen fixation.

Live cuttings were collected one day prior to field installation from multiple donor plants of each species. For each species, at least 20 mother plants were sampled, distributed across different natural populations located in riparian stretches of the Paraopeba River near the experimental site. Only healthy donor plants with similar apparent age and morphological condition were selected to reduce variability in vegetative material quality. Vegetative material was preferentially obtained from branches produced during the most recent vegetative growth cycle. After collection, branches were immediately placed in sealed plastic bags to maintain moisture and prevent desiccation and were transported to the experimental site, where they were stored under shaded conditions until cutting preparation on the following day.

2.3. Field Experiment Conditions

The field experiment was implemented on 21 July 2022 in a defined section of the riprap bank protection structure. Live stakes were prepared from the central portion of branches, with all leaves and lateral shoots removed, measuring approximately 50 cm in length and 10–14 mm in basal diameter (Table 1). A bevel cut was made at the basal end to facilitate soil insertion, and a straight cut was made at the apical end to preserve stake polarity. No pre-planting treatments were applied. Stakes were planted on the same day they were prepared. Approximately 40 cm of each stake was inserted vertically into the soil using a rubber mallet, leaving about 10 cm exposed aboveground. Stakes were carefully driven into the substrate between the rocks of the riprap structure to ensure firm contact with the underlying soil.

Planting was carried out in three parallel planting lines, with species arranged in an alternating sequence within and among lines. The first planting line was positioned immediately above the prevailing waterline at the lower portion of the riprap slope, while the subsequent lines were installed progressively upslope. Spacing was 50 cm between stakes and between planting lines, with a 0.5 m buffer margin maintained on both sides of the planted area. A total of 40 stakes per species were installed (160 stakes in total). The planted section corresponded to a delimited portion of the riprap structure, covering

approximately 56 m². A schematic representation of the planting layout is provided in Figure A1.

Due to the absence of rainfall during the first two months after planting, supplemental manual irrigation was applied every two days, with approximately 3 L of water per stake per irrigation event. Weed control was carried out mechanically, at intervals of approximately 20–30 days, depending on vegetation regrowth.

2.4. Growth and Survival Measurements

Survival and growth performance of live stakes were monitored at 3, 9, 15, and 33 months after planting. Survival was defined as the presence of shoots at the time of assessment. For each surviving stake, the following variables were recorded: shoot number, cumulative shoot length, cumulative shoot diameter, and total plant height. Shoot number was determined by counting all living shoots emerging from the stake.

Shoot length was measured using a graduated ruler, and cumulative shoot length was calculated as the sum of the lengths of all shoots per stake. Shoot diameter was measured with a digital caliper at the base of each shoot, and cumulative shoot diameter was obtained by summing the basal diameters of all shoots per stake. Total height was measured with a graduated ruler as the vertical distance from the ground surface at the stake insertion point to the apex of the tallest shoot.

2.5. Assessment of Recruited Vegetation, Soil Attributes, and Carbon

Natural regeneration was assessed across the entire experimental area during the same monitoring campaigns used for stake performance evaluation (3, 9, 15, and 33 months after planting). All spontaneously recruited woody individuals (shrubs and trees) were recorded and identified to species level.

To provide contextual comparison, spontaneous vegetation was also surveyed along adjacent non-planted riprap sections during the same monitoring period. The assessment was conducted within the same elevation range as the experimental segment, ensuring comparable exposure to river stage fluctuations and flooding. All woody individuals were counted and identified to species level. Herbaceous and grass species were recorded qualitatively to characterize the dominant vegetation cover in these non-planted sections, but their abundance was not quantified. Taxonomic classification followed the Flora do Brasil 2020 database [37].

Composite soil samples were collected across the entire experimental area at depths of 0–20 cm and 20–40 cm prior to the establishment of the experiment and again 33 months after planting. At each sampling campaign and depth, one composite sample was formed by combining soil collected from five randomly selected sampling points within the planted riprap segment, independent of individual stake positions (i.e., not stratified by species). Samples were analyzed for physical and chemical properties using standard laboratory procedures.

Soil organic carbon (SOC) stocks were calculated following [38], based on soil organic carbon concentration, estimated bulk density, and sampling depth, according to:

$$C_{stock} = \frac{SOC \times BD \times depth}{10}$$

where SOC is soil organic carbon content (g kg^{−1}), BD is soil bulk density (g cm^{−3}), and depth is layer thickness (cm), yielding SOC stock in Mg C ha^{−1}.

Bulk density was not measured directly using core cylinders due to the coarse and heterogeneous riprap substrate. Instead, bulk density was estimated using the pedotransfer function proposed by [38]:

$$BD = 0.4123 + 1.0326 e^{-0.0413 \times SOC}$$

Because sampling was conducted in the soil matrix between riprap stones, coarse rock fragments (>2 mm) from the protective structure were not included in the soil samples and therefore were not incorporated into bulk density or SOC stock calculations. Total SOC stocks to 40 cm depth were obtained by summing the values of the two layers. CO₂-equivalent stocks were calculated from SOC stocks using a molecular weight conversion factor (3.67), and total CO₂-equivalent values were likewise obtained by summing both depth layers.

2.6. Statistical Analysis

Growth analyses were conducted considering only surviving cuttings at each evaluation date. Due to the reduced number of surviving individuals per species over time and the unbalanced dataset caused by mortality, non-parametric Kruskal–Wallis tests were used to compare species for each growth variable at each monitoring period. Effect sizes were estimated using epsilon-squared (ϵ^2), calculated as:

$$\epsilon^2 = \frac{H - k + 1}{n - k}$$

where H is the Kruskal–Wallis statistic, k is the number of groups, and n is the total sample size.

Descriptive statistics are presented as medians with 95% bootstrap confidence intervals estimated from 5000 resamples with replacement. Bootstrap confidence intervals were not calculated when sample size was lower than two. Inferential tests were not performed for time points in which fewer than two species had $n \geq 2$. All statistical analyses were performed in R software (version 4.2.2).

3. Results

3.1. Survival Dynamics

Survival of live cuttings varied markedly among species and over time (Figure 4). A pronounced decline in surviving individuals occurred between December 2022 and February 2023, coinciding with prolonged flooding events (Figure 3). During this period, river stage exceeded the inundation threshold (cota 280) for 70 days, including a continuous 58-day period of submergence. Maximum river stage reached 526, with a peak discharge of 364.08 m³ s^{−1} and a mean discharge of 159.4 m³ s^{−1} during inundated days. In subsequent hydrological years, inundation events were less intense and of shorter duration. Over the entire monitoring period, river stage exceeded the defined inundation threshold for a total of 138 days (Figure 3). Cuttings were therefore exposed to extended submergence and mechanical disturbance, resulting in uprooting and sediment burial.

At three months after planting, survival remained relatively high across species. However, by nine months, substantial mortality had occurred, particularly for *Gymnanthes schottiana*, which exhibited complete mortality. Thereafter, survival tended to stabilize among the remaining species. *Croton urucurana* maintained the highest number of surviving individuals throughout the monitoring period, whereas *Sesbania virgata* showed a progressive reduction in surviving individuals toward the final evaluation.

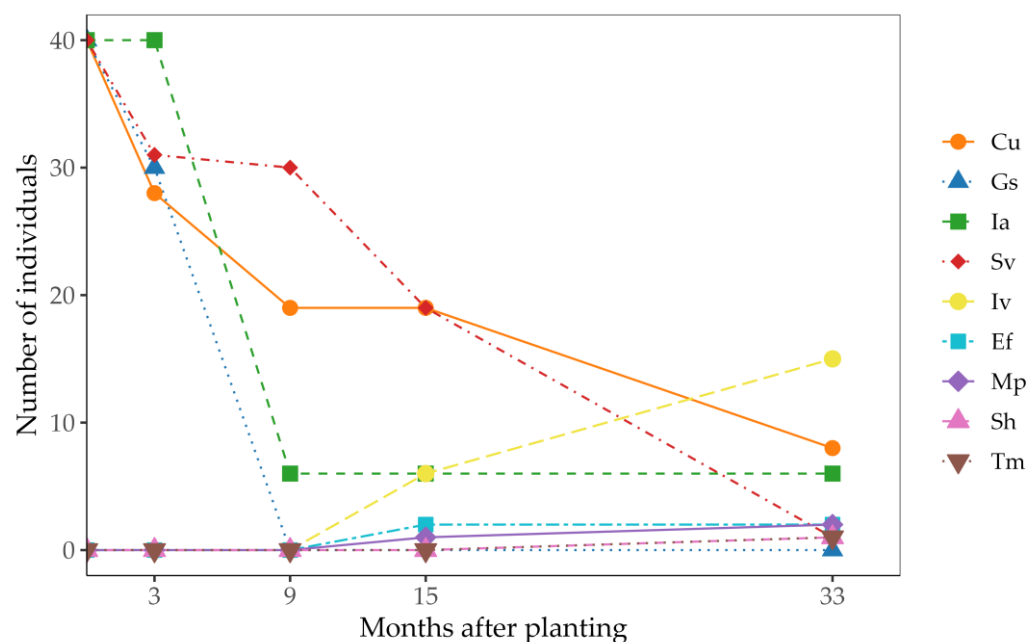


Figure 4. Temporal variation in the number of individuals recorded in the field experiment along the Paraopeba River, Brumadinho, Minas Gerais, Brazil, from planting to 33 months after establishment. Planted species: *Croton urucurana* (Cu), *Gymnanthes schottiana* (Gs), *Iochroma arborescens* (Ia), *Sesbania virgata* (Sv); non-planted species: *Inga vera* (Iv), *Erythrina falcata* (Ef), *Salix humboldtiana* (Sh), *Mimosa pigra* (Mp), *Trema micrantha* (Tm).

3.2. Spontaneous Woody Recruitment

Spontaneous woody recruitment was recorded within the planted experimental segment during the monitoring period (Figure 4). Recruited species included *Inga vera*, *Erythrina falcata*, *Salix humboldtiana*, *Mimosa pigra*, and *Trema micrantha*. A total of 22 woody individuals were recorded within the planted segment at 33 months (Table 2), with *Inga vera* being the most abundant species.

In adjacent non-planted riprap sections, woody recruitment was markedly limited. Only three woody individuals were recorded (two *Inga vera* and one *Mimosa pigra*) across the surveyed area (Table 2). Vegetation cover in these non-planted sections was predominantly composed of herbaceous and grass species, largely invasive, including *Urochloa* spp., *Ricinus communis*, and *Cenchrus purpureus*.

Table 2. Spontaneous woody recruitment recorded at 33 months in planted and adjacent non-planted riprap segments.

Species	Planted Segment	Non-Planted Segment
<i>Erythrina falcata</i> Benth.	2	0
<i>Inga vera</i> Willd.	15	2
<i>Mimosa pigra</i> L.	2	1
<i>Salix humboldtiana</i> Willd.	1	0
<i>Trema micrantha</i> (L.) Blume	1	0
Total woody individuals	22	3

3.3. Growth Performance

Growth trajectories of surviving cuttings are presented in Figure 5, and statistical comparisons among species at each monitoring period are summarized in Table 3. Descriptive statistics expressed as medians with 95% bootstrap confidence intervals are provided in Appendix A, Table A1.

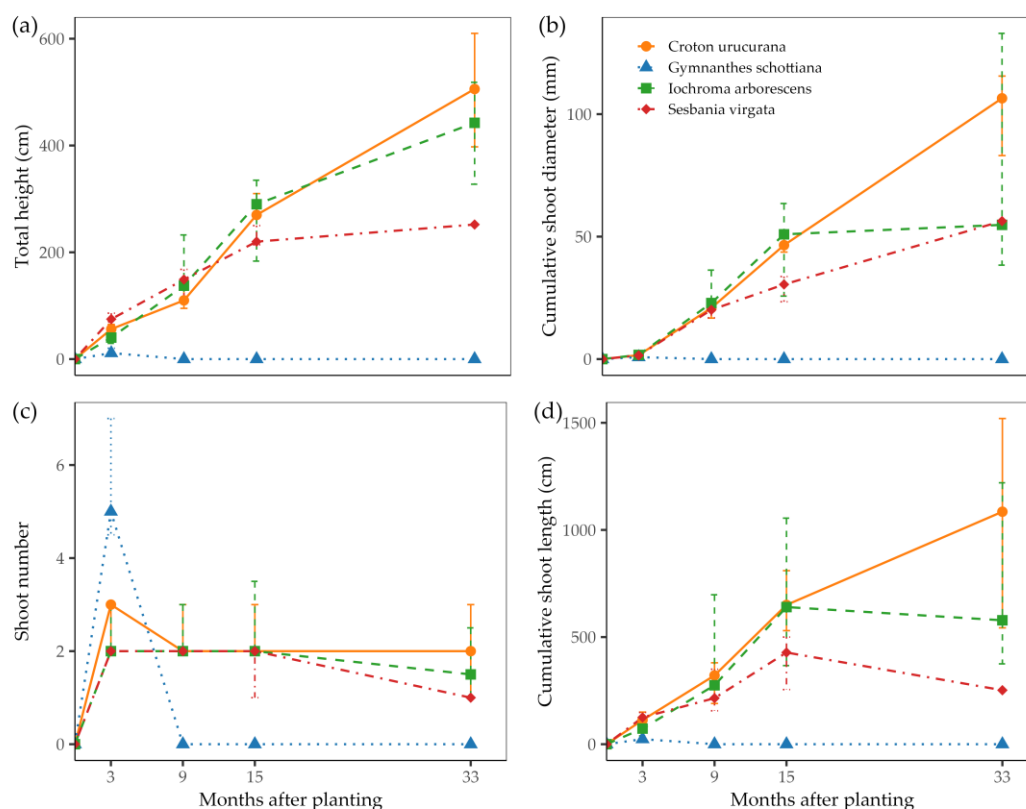


Figure 5. Temporal variation in growth performance of riparian species established as live cuttings on a riprap bank along the Paraopeba River. Panels show (a) total height, (b) cumulative shoot diameter, (c) shoot number, and (d) cumulative shoot length measured at 3, 9, 15, and 33 months after planting. Points represent median values, and error bars indicate 95% bootstrap confidence intervals (5000 resamples), calculated using only surviving individuals at each monitoring period.

Significant interspecific differences were detected at 3 months after planting for all growth variables (Kruskal–Wallis, $p < 0.001$), with moderate to large effect sizes (ϵ^2 ranging from 0.12 to 0.36). At this stage, median height ranged from 11.25 cm in *Gymnanthes schottiana* to 74.8 cm in *Sesbania virgata*, while *Croton urucurana* and *Iochroma arborescens* exhibited intermediate values of 56.15 cm and 40.25 cm, respectively. Similarly, cumulative shoot length was highest in *S. virgata* (125.1 cm) and *C. urucurana* (112.75 cm), whereas *G. schottiana* showed substantially lower structural development (23.75 cm).

By 9 months, interspecific differences were no longer significant for height, cumulative shoot diameter, shoot number, or cumulative shoot length ($p > 0.05$), and effect sizes were small ($\epsilon^2 < 0.05$), indicating temporary convergence in growth performance among surviving individuals. Median height values ranged from 110 cm in *C. urucurana* to 149 cm in *S. virgata*, while cumulative shoot length varied from 215 cm to 321 cm across species.

Table 3. Results of Kruskal–Wallis tests comparing growth variables among species at each monitoring period. The table presents the Kruskal–Wallis statistic (H), associated p -value, and effect size estimated using epsilon-squared (ϵ^2). Effect size values indicate the magnitude of interspecific differences, with higher values representing stronger effects.

Metric	Period	H	df	p -Value	ϵ^2
Height (cm)	3 months	37.92	3	<0.001	0.279
	9 months	2.19	2	0.335	0.004
	15 months	16.2	2	<0.001	0.346
	33 months	-	-	-	-
Cumulative shoot diameter (mm)	3 months	21.54	3	<0.001	0.148
	9 months	2.49	2	0.288	0.009
	15 months	20.22	2	<0.001	0.444
	33 months	-	-	-	-
Shoot number	3 months	48.47	3	<0.001	0.364
	9 months	4.66	2	0.097	0.051
	15 months	4.59	2	0.101	0.063
	33 months	-	-	-	-
Cumulative shoot length (cm)	3 months	18.38	3	<0.001	0.123
	9 months	2.06	2	0.358	0.001
	15 months	11.09	2	0.004	0.222
	33 months	-	-	-	-

Clear divergence in growth resumed at 15 months. Significant interspecific differences were observed for height ($H = 16.2$, $p < 0.001$, $\epsilon^2 = 0.346$), cumulative shoot diameter ($H = 20.22$, $p < 0.001$, $\epsilon^2 = 0.444$), and cumulative shoot length ($H = 11.09$, $p = 0.004$, $\epsilon^2 = 0.222$). Median height reached 290 cm in *I. arborescens* and 270 cm in *C. urucurana*, compared with 220 cm in *S. virgata*. Similarly, cumulative shoot diameter was highest in *I. arborescens* (50.94 mm) and *C. urucurana* (46.5 mm), whereas *S. virgata* exhibited lower values (30.48 mm).

At 33 months, growth patterns remained consistent with previous trends, although inferential statistical comparisons were not performed due to reduced sample size and unbalanced species representation. Median height reached 505.85 cm in *C. urucurana*, substantially exceeding values observed in *I. arborescens* (442.5 cm) and *S. virgata* (252 cm). Likewise, cumulative shoot diameter reached 106.47 mm in *C. urucurana*, nearly twice the values observed in the other species.

Overall, growth trajectories clearly distinguished species in terms of field performance. *Croton urucurana* consistently exhibited superior structural development, achieving the highest height, diameter, and shoot length values throughout the monitoring period. *Iochroma arborescens* showed intermediate but sustained growth among surviving individuals, whereas *Sesbania virgata* exhibited reduced structural development associated with low survival. *Gymnanthes schottiana* failed to establish successfully under field conditions due to early mortality. The temporal evolution of the bioengineered riverbank, including periods of partial and complete submergence and subsequent vegetation recovery, is illustrated in Figure 6.



Figure 6. Temporal evolution of a soil and water bioengineering intervention using live cuttings along the Paraopeba River, Brazil. (a) Immediately after planting, showing freshly installed cuttings on the riprap slope; (b) early establishment phase, approximately three months after planting, with initial shoot development and partial vegetation cover; (c) flood event during the establishment phase, illustrating plant exposure to inundation and fluvial disturbance; (d) advanced post-establishment phase, showing dense vegetation cover and structural integration of planted and spontaneously recruited species on the riverbank.

3.4. Soil Properties and Carbon Stock Changes

Soil chemical properties and soil organic carbon (SOC) stocks increased between pre-planting conditions and 33 months after the establishment of live cuttings (Tables 4 and 5). Because soil samples were collected as composite samples representing the experimental area, statistical comparisons were not performed, and observed changes are presented descriptively.

Table 4. Chemical properties of the soil before planting and 33 months after the establishment of live cuttings.

Soil Attribute	Before Planting		33 Months After	
	0–20 cm	20–40 cm	0–20 cm	20–40 cm
Organic matter ¹ (g kg ⁻¹)	15.50	7.80	22.10	20.5
pH in water (1:2.5)	6.94	6.75	6.38	6.40
Available P ² (mg dm ⁻³)	4.50	4.00	3.70	3.40
Exchangeable K ² (mg dm ⁻³)	42.0	30.0	60.0	42.0
Exchangeable Ca ³ (cmolc dm ⁻³)	2.69	1.67	2.68	2.91
Exchangeable Mg ³ (cmolc dm ⁻³)	1.13	0.58	1.24	1.27
Exchangeable Al ³ (cmolc dm ⁻³)	1.00	0.90	1.50	1.50
Available Fe ² (mg dm ⁻³)	676.50	553.90	624.00	643.9
Available Cu ² (mg dm ⁻³)	4.19	3.07	3.95	4.01
Available Mn ² (mg dm ⁻³)	264.20	181.00	273.40	170.9
Available Zn ² (mg dm ⁻³)	3.78	2.90	5.25	5.15
CEC at pH 7.0 (cmolc dm ⁻³)	4.93	3.23	5.60	5.81

Base saturation (%)	3.93	2.33	4.10	4.31
---------------------	------	------	------	------

¹ Walkley–Black method; ² extracted by Mehlich-1; ³ extracted by KCl 1 mol L⁻¹.

Soil organic carbon content increased at both sampled depths, from 11.7 to 16.7 g kg⁻¹ in the 0–20 cm layer and from 5.9 to 15.5 g kg⁻¹ in the 20–40 cm layer (Table 5). Bulk density decreased from 1.05 to 0.93 g cm⁻³ in the 0–20 cm layer and from 1.22 to 0.96 g cm⁻³ in the 20–40 cm layer.

These changes were reflected in SOC stock values. SOC stock increased from 24.5 to 31.0 Mg C ha⁻¹ in the 0–20 cm layer and from 14.4 to 29.6 Mg C ha⁻¹ in the 20–40 cm layer. When both layers were combined, total SOC stock in the 0–40 cm soil profile increased from 38.9 to 60.6 Mg C ha⁻¹, corresponding to a net increase of 21.7 Mg C ha⁻¹ over the 33-month monitoring period (Table 5).

Table 5. Soil organic carbon content, carbon stock, and CO₂-equivalent before planting and 33 months after the establishment of live cuttings at two soil depths.

	Soil Depth (cm)	Soil Organic Carbon (g kg ⁻¹)	Bulk Density (g cm ⁻³)	Carbon Stock (Mg C ha ⁻¹)	CO ₂ -eq (Mg CO ₂ -eq ha ⁻¹)
Before planting	0–20 cm	11.7	1.05	24.5	90.0
	20–40 cm	5.9	1.22	14.4	52.7
	Total (0–40 cm)	-	-	38.9	142.7
33 months after	0–20 cm	16.7	0.93	31.0	113.8
	20–40 cm	15.5	0.96	29.6	108.6
	Total (0–40 cm)	-	-	60.6	222.4

Similarly, CO₂-equivalent stocks increased from 142.7 to 222.4 Mg CO₂-eq ha⁻¹ for the 0–40 cm profile. Soil chemical properties also changed over the monitoring period, including increases in organic matter content and exchangeable base cations at both depths, while soil pH showed a slight decrease (Table 4).

4. Discussion

4.1. Species Performance and Selection Criteria

The present study demonstrates pronounced interspecific differences in survival and structural development during the establishment phase of live cuttings under highly dynamic riparian conditions. Among the evaluated species, *Croton urucurana* consistently exhibited the highest survival and growth performance, reaching median heights exceeding 5 m and cumulative shoot diameters greater than 100 mm after 33 months (Figures 4 and 5). In contrast, *Gymnanthes schottiana* experienced complete mortality during the first year, while *Iochroma arborescens* and *Sesbania virgata* showed intermediate performance, with reduced survival but substantial structural development among surviving individuals. These contrasting responses provide clear evidence of species-specific differences in establishment success and resilience to hydrodynamic disturbance, offering practical guidance for species selection in soil and water bioengineering interventions.

The marked interspecific differences observed in this study highlight the critical importance of functional trait-based species selection, particularly traits associated with early growth and structural resilience under hydrodynamic stress. Differences in growth performance were already evident prior to the onset of prolonged flooding. At three months after planting, *C. urucurana* exhibited substantially greater structural development, with median height reaching approximately 56 cm and cumulative shoot length exceeding 110 cm, whereas *G. schottiana* showed limited growth, with median height below 12 cm and cumulative shoot length below 25 cm (Figure 5 and Table A1). This early

structural advantage likely facilitated faster root system establishment and improved anchorage capacity, increasing resistance to mechanical disturbance during subsequent flooding events. Rapid early growth promotes both structural reinforcement and anchorage development, enhancing resistance to hydraulic forces and reducing susceptibility to uprooting. Consistent with this interpretation, previous studies have demonstrated that survival, early growth rates, root system development, and associated structural traits are key determinants of establishment success and stabilization effectiveness in SWBE systems [12,39–41].

Consistent with these mechanisms, *Croton urucurana* and *Sesbania virgata* exhibited key functional attributes related to early growth and anchorage development, supporting their suitability for SWBE applications in tropical riparian environments. In particular, the rapid early growth and structural reinforcement observed in both species likely enhanced resistance to hydraulic forces and improved root anchorage stability, facilitating survival during prolonged submergence and sediment disturbance. This interpretation aligns with previous findings indicating that rapid early growth and the development of functionally relevant morphological traits are critical for effective soil reinforcement in bioengineering systems [42–44]. Experimental studies have further demonstrated that vegetation contributions to soil stability increase markedly during the early establishment phase, highlighting the engineering importance of rapid biomass accumulation [45–47]. In particular, studies evaluating the vertical pullout resistance of live cuttings have shown strong positive relationships between morphological traits, especially shoot cross-sectional area, and root system characteristics such as total root length and spatial distribution, which directly influence anchorage capacity and resistance to uprooting [48,49].

Although *S. virgata* exhibited lower long-term persistence compared to *C. urucurana*, it showed the highest survival immediately following the prolonged inundation event and demonstrated the greatest root system development under greenhouse conditions [29]. This extensive early root development likely enhanced anchorage stability during the critical establishment phase, increasing resistance to uprooting under prolonged submergence. In contrast, the superior long-term persistence and structural development observed in *C. urucurana* indicates a greater capacity to maintain mechanical stability under repeated hydrodynamic disturbance. These findings highlight the importance of selecting species with complementary functional strategies to enhance SWBE effectiveness and resilience.

In addition to structural development and anchorage capacity, physiological tolerance to flooding represents a critical determinant of species persistence in riparian environments. Flood-tolerant species possess adaptations such as adventitious roots, aerenchyma formation, and metabolic adjustments that enable survival under hypoxic conditions [50]. Controlled experiments that have demonstrated survival and growth responses under flooding or intermittent saturation can vary substantially among species, highlighting that tolerance mechanisms do not necessarily translate into sustained field performance [27,51]. Among the evaluated species, *S. virgata*, *C. urucurana*, and *G. schottiana* are widely recognized as flood-tolerant riparian species [52–55], whereas *I. arborescens* lacks documented adaptations to prolonged inundation and is typically associated with well-drained to moderately moist soils [56]. Its reduced persistence observed in this study may therefore reflect limited physiological tolerance to prolonged submergence.

Despite its well-documented flooding tolerance and extensive early root development observed under greenhouse conditions, *Sesbania virgata* exhibited reduced long-term persistence under field conditions. Previous studies have reported adaptive morpho-anatomical responses in this species under flooding, including adventitious root formation, hypertrophied lenticels, stem fissures, and aerenchyma development, which enhance oxygen transport and enable survival under hypoxic conditions [53,54]. Additionally, its

capacity for biological nitrogen fixation contributes to improved soil fertility and nutrient cycling in degraded substrates, enhancing its ecological value for soil recovery and the revegetation of degraded riparian environments [57,58]. However, its relatively short life cycle [59] and potential susceptibility to herbivory and insect attack [28,60], may limit long-term persistence despite strong early establishment.

In contrast, *G. schottiana* showed complete mortality despite initial sprouting, indicating failure to establish functional root anchorage during the critical early establishment phase. Although live cuttings can initially sustain shoot growth using stored reserves, delayed root system development increases vulnerability to uprooting under hydrodynamic stress [43,48]. This temporal mismatch between aboveground development and effective root anchorage likely compromised mechanical stability during prolonged submergence and sediment disturbance, explaining its failure under field conditions despite its inherent tolerance to flooding reported in the literature [55,61]. Previous experimental evaluations have similarly reported slow vegetative development in this species, particularly characterized by limited root production and slower rooting rates compared to other riparian species, which may further constrain anchorage establishment during early development stages [62].

I. arborescens exhibited limited persistence, likely reflecting its lack of specific adaptations to prolonged inundation. Although surviving individuals showed post-disturbance recovery, its overall performance suggests limited suitability for SWBE applications in highly dynamic riparian environments.

Collectively, these findings demonstrate that successful establishment in SWBE interventions depends on the combined influence of rapid structural development, effective anchorage establishment, and physiological tolerance to flooding.

4.2. The Hydrological Filter as the Primary Driver of Establishment Success

Hydrological disturbance acted as the dominant ecological filter controlling establishment success in this study. The extreme flooding event recorded between December 2022 and February 2023, during which river stage exceeded the inundation threshold for 70 days, including 58 consecutive days of submergence, coincided with the critical establishment period when root systems were still developing (Figure 3). This event coincided with the sharp decline in survival observed between the 3- and 9-month assessments (Figure 4), indicating that hydrological disturbance was the primary determinant of early establishment success. Hydrological conditions, including rainfall intensity, flooding frequency, and soil moisture dynamics, represent critical determinants of vegetation establishment and bioengineering performance, particularly in tropical environments where seasonal hydrological variability strongly influences rooting success and plant survival [12,14].

This prolonged submergence likely generated multiple stress mechanisms, including mechanical uprooting by flow forces, sediment burial, and hypoxic stress associated with prolonged soil saturation. Such disturbance regimes are well recognized as primary drivers of riparian vegetation assembly, selectively favoring species capable of rapid anchorage development and structural resistance [22,34,63]. The magnitude and duration of this event were substantially greater than those of subsequent inundation periods, which were shorter and less continuous, explaining the marked mortality observed during early establishment (Figure 3). These results highlight the establishment phase as a critical ecological bottleneck, during which plant survival is strongly conditioned by the interaction between species functional traits and disturbance intensity.

Importantly, riparian species are typically distributed along microtopographic and hydrological gradients, reflecting differences in tolerance to inundation depth, duration, and sediment dynamics. In this study, all species were planted along the same elevation

band near the active waterline, which likely exposed less flood-tolerant species to stress levels beyond their optimal ecological range. In natural riparian systems, species zonation reflects adaptation to distinct hydrological niches, with some species occupying frequently disturbed zones near the channel and others establishing in more stable positions further upslope. Incorporating species-specific hydrological preferences into planting design may improve establishment success by aligning species traits with site-specific disturbance regimes and reducing exposure to hydraulic stress exceeding species tolerance thresholds.

These findings are particularly relevant under projected climate change scenarios, which are expected to increase the frequency and intensity of extreme hydrological events in tropical regions. Increased flood magnitude and duration may further constrain establishment success and amplify the importance of species selection based on structural resilience, rapid root anchorage development, and disturbance tolerance. Bioengineering interventions must therefore incorporate climate-resilient species and consider hydrological zonation in planting design to enhance establishment success and long-term stabilization under increasingly dynamic disturbance regimes.

4.3. Ecosystem Engineering Effects: Vegetation Establishment, Recruitment, and Soil Development

Beyond structural stabilization, the establishment of live cuttings initiated broader ecological recovery processes. Spontaneous recruitment of woody species was substantially higher within the planted segment (22 individuals) compared to adjacent non-planted riprap sections (3 individuals), which remained dominated by herbaceous and invasive species (Table 2). This contrast provides strong evidence that vegetation establishment facilitated ecological succession by modifying local hydraulic and geomorphological conditions.

Vegetation establishment increases surface roughness, reduces near-bank flow velocity, and enhances sediment and propagule retention, thereby creating favorable microsites for seed deposition and germination [11,20]. These vegetation–flow feedbacks represent fundamental ecosystem engineering processes, whereby plants modify physical environmental conditions and influence sediment dynamics, substrate stability, and vegetation development trajectories. By altering local hydrological and geomorphological conditions, early establishing vegetation facilitates the recruitment and persistence of additional riparian species [64]. Similar ecosystem engineering effects have been widely documented in tropical soil bioengineering interventions, where vegetation contributes to erosion control, substrate stabilization, and progressive ecosystem restoration [12,14].

The spontaneous recruitment of native species observed during the monitoring period provides further evidence of the ecological effectiveness of the SWBE intervention. Similar successional responses have been reported in bioengineered riverbanks, where introduced vegetation acts as a catalyst for natural regeneration and accelerates plant community development [11]. In contrast to inert riprap structures, which provide primarily mechanical stabilization, SWBE interventions actively promote colonization and successional advancement by restoring vegetation–flow feedback mechanisms and facilitating ecosystem recovery [20]. Comparative studies further demonstrate that bioengineered riverbanks support higher plant diversity and more natural successional trajectories than conventional hard engineering solutions [18].

In parallel with vegetation development, increases in soil organic carbon stocks and changes in soil chemical properties indicate progressive soil development. Soil organic carbon stocks increased from 38.9 to 60.6 Mg C ha^{−1} in the 0–40 cm profile, likely reflecting combined contributions from plant-derived organic inputs, including litter deposition and root biomass accumulation, as well as sediment-associated organic material

deposited during flooding events (Tables 4 and 5). Because soil samples were collected as composite samples and control plots were not available, causal attribution of soil changes exclusively to vegetation establishment cannot be established definitively. However, the observed increases are consistent with vegetation-driven soil development processes widely reported in riparian restoration and bioengineered systems [13,65].

Additional changes in soil chemical properties, including increased potassium (K) concentrations and base saturation values (V%), further suggest enhanced nutrient cycling associated with vegetation establishment. These changes likely reflect litter decomposition, root turnover, and biogeochemical cycling processes characteristic of early stages of soil development in restored riparian environments [66–68]. Carbon accumulation in deeper soil layers is particularly relevant for long-term ecosystem functioning, as deeper carbon pools tend to exhibit greater stability and longer residence times.

From a broader ecosystem services perspective, these findings indicate that SWBE interventions can promote multiple ecosystem functions beyond erosion control, including vegetation recovery, soil development, and enhanced biogeochemical cycling [19,33]. By facilitating vegetation establishment and initiating ecosystem engineering processes, SWBE interventions can accelerate the transition from structurally stabilized but ecologically simplified riverbanks toward self-sustaining riparian ecosystems.

Overall, the results demonstrate that SWBE constitutes a multifunctional nature-based solution capable of integrating structural stabilization with ecological recovery processes. By linking vegetation establishment, successional recruitment, and soil development, these interventions contribute to the restoration of ecosystem structure, functioning, and resilience in degraded tropical riparian environments.

4.4. Implications for Tropical Soil and Water Bioengineering Practice

The results provide important practical guidance for species selection and the design of bioengineering interventions in tropical riparian environments. Among the tested species, *C. urucurana* and *S. virgata* demonstrated favorable performance in terms of early establishment, growth, and structural development, indicating strong suitability for SWBE applications under dynamic hydrological conditions. In particular, *C. urucurana* exhibited the highest survival and sustained structural development throughout the monitoring period, supporting its prioritization as a primary species for live staking in tropical SWBE interventions, especially in zones exposed to frequent hydrological disturbance. *S. virgata* showed strong early establishment and rapid growth, highlighting its value as a complementary species capable of promoting rapid initial stabilization, early substrate reinforcement, and facilitation of subsequent successional processes. In contrast, *G. schottiana* showed poor establishment success when installed as live cuttings, suggesting limited suitability for direct live staking under extreme hydrological disturbance. However, its use may still be viable through the planting of nursery-grown seedlings with pre-established root systems, which may enhance anchorage capacity and increase resistance to hydraulic disturbance during the critical early establishment phase.

These findings emphasize the importance of selecting species with functional traits associated with rapid anchorage, structural reinforcement, and resilience to prolonged flooding [23,49]. Previous applications of SWBE in Central and South America have similarly demonstrated that successful implementation depends strongly on appropriate species selection, local environmental adaptation, and understanding of site-specific climatic and hydrological conditions [12,69]. Incorporating species zonation along hydrological gradients and combining early-establishing pioneer species with longer-lived riparian species may enhance long-term restoration success [21].

The study site represents a highly disturbed post-mining disaster environment, which may differ from natural riparian systems in substrate composition,

geomorphology, and disturbance intensity. Therefore, while the findings provide valuable insights, extrapolation to other sites should consider local environmental conditions and disturbance regimes.

Seasonal timing of installation represents another critical factor influencing establishment success in tropical riparian environments characterized by distinct wet and dry seasons [12,69]. In regions with pronounced seasonal precipitation patterns, such as southeastern Brazil, where intense rainfall and flooding are concentrated within a defined wet season, live staking should be scheduled to maximize the period available for root system development prior to the onset of high-flow conditions. In the present study, planting was conducted in July, during the dry season, when precipitation levels were already substantially reduced and flood risk was minimal, allowing approximately three to four months of initial growth before the onset of severe flooding. However, installation during the dry season also requires supplemental irrigation during the first months after planting to compensate for limited rainfall and prevent desiccation of cuttings before root systems become fully functional. In this study, supplemental irrigation was applied during the early establishment phase, likely contributing to initial survival and shoot development.

An alternative and potentially optimal strategy is installation near the end of the rainy season, typically between April and early May, when rainfall remains sufficient to support establishment but occurs at lower intensity and with reduced flood risk compared to peak rainy-season months. This timing may provide adequate soil moisture while maximizing the duration available for root development prior to the next period of intense hydrological disturbance. These findings highlight that aligning installation timing with seasonal hydrological cycles, combined with supplemental irrigation when necessary, represents a critical design consideration for optimizing survival, anchorage development, and long-term stabilization effectiveness in tropical soil and water bioengineering interventions. In tropical regions with predictable seasonal rainfall regimes, installation near the end of the rainy season may provide the most favorable balance between moisture availability and reduced hydraulic disturbance risk.

Under projected climate change scenarios, which are expected to intensify hydrological extremes in tropical regions, bioengineering interventions that incorporate climate-resilient species and adaptive design principles will become increasingly important. By integrating structural stabilization with ecological recovery processes, soil and water bioengineering represents a promising nature-based solution for enhancing the resilience of riparian systems exposed to intensified disturbance regimes. Nevertheless, the underlying ecological mechanisms identified in this study are broadly applicable to riparian bioengineering interventions in other tropical systems with similar hydrological regimes.

4.5. Study Limitations and Implications for Long-Term Riparian Restoration

This study was conducted at a single experimental site without replicated control plots, which limits the ability to attribute observed vegetation and soil changes exclusively to the soil and water bioengineering (SWBE) intervention. Although adjacent non-planted riprap sections provided qualitative comparison, they do not constitute fully replicated experimental controls and therefore do not allow formal statistical inference regarding treatment effects.

Additionally, soil samples were collected as composite samples representing the experimental area rather than stratified by species or spatial position. This sampling approach precludes statistical analysis and prevents attribution of soil changes to specific plant species or localized vegetation development patterns. Furthermore, sediment deposition during flooding events may have contributed to increases in soil organic carbon and nutrient levels through allochthonous inputs, indicating that soil improvements likely reflect the combined influence of vegetation establishment and fluvial sediment dynamics.

The monitoring period of 33 months represents an establishment-phase assessment rather than long-term ecosystem development. Early establishing species such as *C. urucurana* and *S. virgata* may function as ecological facilitators, enhancing substrate stabilization, promoting sediment retention, and creating environmental conditions that facilitate the recruitment and establishment of additional riparian species. The spontaneous recruitment of native species observed during the study, including *Inga vera* and *Salix humboldtiana*, suggests that successional processes have already been initiated.

However, long-term vegetation trajectories remain uncertain and may be influenced by future hydrological disturbance regimes, species interactions, and ongoing geomorphological adjustments. Early dominance of rapidly establishing species does not necessarily imply long-term persistence or community dominance, as later-successional species may progressively increase in abundance as ecosystem structure and stability develop. Continued monitoring is therefore necessary to evaluate long-term community dynamics, vegetation succession, and ecosystem stabilization processes following SWBE implementation.

Despite these limitations, the study provides strong field-based evidence of species-specific performance under realistic hydrological disturbance conditions and highlights key functional traits associated with successful establishment. These findings contribute valuable empirical guidance for species selection and the design of SWBE interventions in tropical river systems.

5. Conclusions

This study demonstrates that the effectiveness of soil and water bioengineering interventions in tropical riparian environments is strongly influenced by species functional traits and their interaction with hydrological disturbance during the critical establishment phase. Prolonged flooding acted as the dominant ecological filter, selectively favoring species capable of rapid root development, effective anchorage, and structural resistance to hydrodynamic stress.

Among the evaluated species, *Croton urucurana* exhibited the highest survival and sustained structural development, supporting its prioritization for live staking in SWBE interventions under dynamic hydrological conditions. *Sesbania virgata* showed rapid early growth and establishment but reduced long-term persistence, while *Gymnanthes schottiana* failed to establish successfully as live cuttings, suggesting that alternative approaches, such as planting nursery-grown seedlings, may be required for its use.

Vegetation establishment promoted spontaneous recruitment and soil development, indicating that SWBE interventions can initiate ecological recovery processes beyond structural stabilization. These findings provide practical guidance for species selection and implementation strategies and support the use of soil and water bioengineering as an effective nature-based solution for stabilizing riverbanks and restoring tropical riparian ecosystems.

Author Contributions: Conceptualization, P.L.W.K. and S.V.M.; methodology, P.L.W.K. and S.V.M.; software, P.L.W.K.; validation, P.L.W.K., F.D.D.C., M.d.C.C. and E.M.F.; formal analysis, P.L.W.K., F.D.D.C. and M.d.C.C.; investigation, P.L.W.K., F.D.D.C., M.d.C.C. and E.M.F.; resources, D.A.d.S.O.; data curation, P.L.W.K.; writing—original draft preparation, P.L.W.K., F.D.D.C. and M.d.C.C.; writing—review and editing, P.L.W.K., S.V.M., F.D.D.C., M.d.C.C., E.M.F. and D.A.d.S.O.; visualization, P.L.W.K., F.D.D.C. and M.d.C.C.; supervision, S.V.M.; project administration, S.V.M.; funding acquisition, S.V.M. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by VALE S.A., Grant No. 1539 UFV/VALE/SIF. The APC was funded by VALE S.A.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: The original contributions presented in this study are included in the article. Further inquiries can be directed to the corresponding author.

Acknowledgments: The authors acknowledge Vale S.A. for funding the project and CNPq for providing a Research Productivity Fellowship to the second author.

Conflicts of Interest: Author Diego Aniceto dos Santos Oliveira was employed by the company Vale S.A. The remaining authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest. This study received funding from Vale S.A. The funder was not involved in the study design, data collection, analysis, interpretation of data, writing of the manuscript, or the decision to submit the article for publication.

Abbreviations

The following abbreviations are used in this manuscript:

SWBE	Soil and Water Bioengineering
SOC	Soil Organic Carbon
CO ₂ -eq	Carbon Dioxide Equivalent
CEC	Cation Exchange Capacity
V (%)	Base Saturation
FT	Flood Tolerance
BT	Burial Tolerance
DR	Drought Resistance
WF	Wildlife Food Provision
NF	Biological Nitrogen Fixation

Appendix A

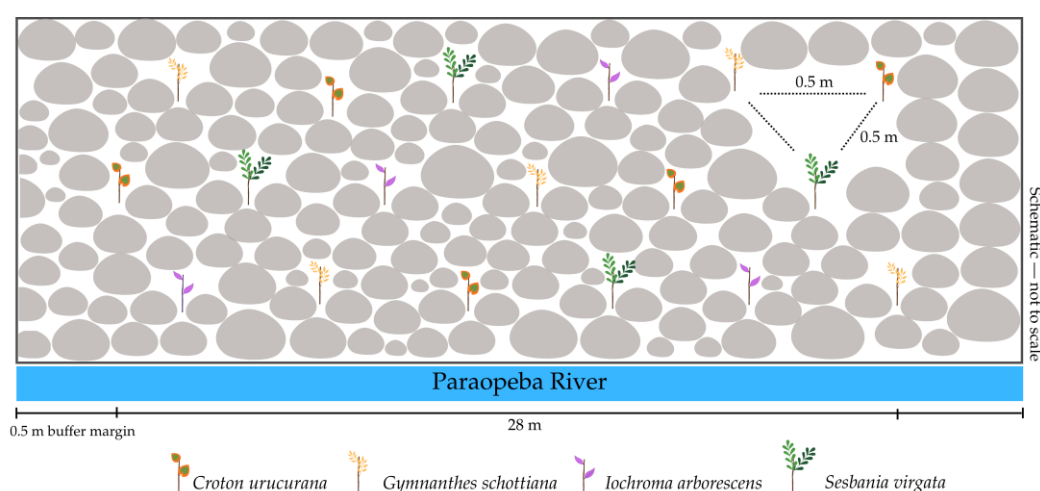


Figure A1. Schematic layout of the live-stake planting design on the riprap bank of the Paraopeba River, showing three parallel planting lines with alternating species, 0.5 m spacing, and buffer margins. Not to scale.

Table A1. Median values and 95% bootstrap confidence intervals (in parentheses) for height, cumulative shoot diameter, shoot number, and cumulative shoot length of live cuttings at each monitoring period. Confidence intervals were estimated using 5000 bootstrap resamples with replacement. Dashes (–) indicate the absence of surviving individuals. Confidence intervals were not calculated when sample size was lower than two ($n < 2$).

		<i>C. urucurana</i>	<i>G. schottiana</i>	<i>I. arborescens</i>	<i>S. virgata</i>
Height (cm)	3 months	56.15 (38.5–64.9)	11.25 (5.9–20.35)	40.25 (29.45–58)	74.8 (63–86)
	9 months	110 (95–135)	—	137.5 (107–232.5)	149 (114–168)
	15 months	270 (265–310)	—	290 (183.5–335)	220 (214–250)
	33 months	505.85 (397.5–610)	—	442.5 (327.5–518.35)	252
Cumulative shoot diameter (mm)	3 months	1.6 (1.35–2.1)	0.8 (0.65–1)	1.7 (1.1–2.4)	1.5 (1.1–1.8)
	9 months	20.87 (16.73–23.11)	—	22.87 (19.64–36.33)	20.1 (16.85–22.9)
	15 months	46.5 (43.69–49.56)	—	50.94 (25.73–63.52)	30.48 (23.41–33.71)
	33 months	106.47 (83.08–115.55)	—	54.8 (38.36–132.97)	56.34
Shoot number	3 months	3 (2–3)	5 (4.5–7)	2 (2–3)	2 (2–2)
	9 months	2 (2–3)	—	2 (2–3)	2 (2–2)
	15 months	2 (2–3)	—	2 (2–3.5)	2 (1–2)
	33 months	2 (1–3)	—	1.5 (1–2.5)	1
Cumulative shoot length (cm)	3 months	112.75 (71.8–149.85)	23.75 (13.05–74.15)	72.95 (39.08–111.8)	125.1 (87.8–148.2)
	9 months	321 (190–380)	—	275 (214–697.5)	215 (156–350)
	15 months	650 (530–810)	—	640 (367–1055)	428 (255–500)
	33 months	1085 (543–1520.1)	—	578.35 (375–1220)	252

References

- Davis, S.; Grainger, M.; Pfeifer, M.; Pattison, Z.; Stephens, P.; Sanderson, R. Restoring Riparian Habitats for Benefits to Biodiversity and Human Livelihoods: A Systematic Map Protocol for Riparian Restoration Approaches in the Tropics. *Environ. Evid.* **2025**, *14*, 2. <https://doi.org/10.1186/s13750-025-00355-8>.
- Vagheei, H.; Boano, F. Freshwater Riparian Zones in a Changing Climate: A Comprehensive Review. *Ecol. Indic.* **2025**, *176*, 113600. <https://doi.org/10.1016/j.ecolind.2025.113600>.
- Camarero, J.J.; Colangelo, M.; Rodríguez-Gonzalez, P.M. Historical Disconnection from Floodplain Alters Riparian Forest Composition, Tree Growth and Deadwood Amount. *Sci. Total Environ.* **2023**, *896*, 165266. <https://doi.org/10.1016/j.scitotenv.2023.165266>.
- Okeke, C.A.U.; Uno, J.; Academe, S.; Emenike, P.G.C.; Abam, T.K.S.; Omole, D.O. An Integrated Assessment of Land Use Impact, Riparian Vegetation and Lithologic Variation on Streambank Stability in a Peri-Urban Watershed (Nigeria). *Sci. Rep.* **2022**, *12*, 10989. <https://doi.org/10.1038/s41598-022-15008-w>.
- IPCC. *Climate Change 2023: Synthesis Report. Contribution of Working Groups I, II and III to the Sixth Assessment*; Intergovernmental Panel on Climate Change (IPCC): Geneva, Switzerland, 2023.
- Adeyeri, O.E.; Folorunsho, A.H.; Adeliyi, T.E.; Ayegbusi, K.I.; Akinsanola, A.A.; Ndehedehe, C.E.; Ahmed, N.; Babalola, T.E. Climate Change Is Intensifying Rainfall Erosivity and Soil Erosion in West Africa. *Sci. Total Environ.* **2024**, *955*, 177174. <https://doi.org/10.1016/j.scitotenv.2024.177174>.
- Rey, F. Harmonizing Erosion Control and Flood Prevention with Restoration of Biodiversity through Ecological Engineering Used for Co-Benefits Nature-Based Solutions. *Sustainability* **2021**, *13*, 1150. <https://doi.org/10.3390/su132011150>.
- Preti, F.; Capobianco, V.; Sangalli, P. Soil and Water Bioengineering (SWB) Is and Has Always Been a Nature-Based Solution (NBS): A Reasoned Comparison of Terms and Definitions. *Ecol. Eng.* **2022**, *181*, 106687. <https://doi.org/10.1016/j.ecoleng.2022.106687>.
- dos Santos Sousa, R.; Dewes, J.J.; Rauch, H.P.; Sutili, F.J.; Hörbinger, S. First Results of Soil and Water Bioengineering Interventions to Stabilise and Control Erosion Processes in Hydroelectric Power Plant Reservoirs in Brazil. *Ecol. Eng.* **2025**, *211*, 107458. <https://doi.org/10.1016/j.ecoleng.2024.107458>.

10. von der Thannen, M.; Hoerbinger, S.; Muellebner, C.; Biber, H.; Rauch, H.P. Case Study of a Water Bioengineering Construction Site in Austria. Ecological Aspects and Application of an Environmental Life Cycle Assessment Model. *Int. J. Energy Environ. Eng.* **2021**, *12*, 599–609. <https://doi.org/10.1007/s40095-021-00419-8>.
11. Kettenhuber, P.L.W.; Sousa, R.S.; Dewes, J.J.; Rauch, H.P.; Sutili, F.J.; Hörbinger, S. Performance Assessment of a Soil and Water Bioengineering Work on the Basis of the Flora Development and Its Associated Ecosystem Processes. *Ecol. Eng.* **2023**, *186*, 106840. <https://doi.org/10.1016/j.ecoleng.2022.106840>.
12. Maxwald, M.; Crocetti, C.; Ferrari, R.; Petrone, A.; Rauch, H.P.; Preti, F. Soil and Water Bioengineering Applications in Central and South America: A Transferability Analysis. *Sustainability* **2020**, *12*, 10505. <https://doi.org/10.3390/su122410505>.
13. Zhang, H.; Zhao, Z.; Ma, G.; Sun, L. Quantitative Evaluation of Soil Anti-Erodibility in Riverbank Slope Remediated with Nature-Based Soil Bioengineering in Liaohe River, Northeast China. *Ecol. Eng.* **2020**, *151*, 105840. <https://doi.org/10.1016/j.ecoleng.2020.105840>.
14. Petrone, A.; Preti, F. Soil Bioengineering for Risk Mitigation and Environmental Restoration in a Humid Tropical Area. *Hydrol. Earth Syst. Sci.* **2010**, *14*, 239–250. <https://doi.org/10.5194/hess-14-239-2010>.
15. Anstead, L.; Boar, R.R.; Tovey, N.K. The Effectiveness of a Soil Bioengineering Solution for River Bank Stabilisation during Flood and Drought Conditions: Two Case Studies from East Anglia. *Area* **2012**, *44*, 479–488. <https://doi.org/10.1111/j.1475-4762.2012.01132.x>.
16. Giupponi, L.; Bischetti, G.B.; Giorgi, A. A Proposal for Assessing the Success of Soil Bioengineering Work by Analysing Vegetation: Results of Two Case Studies in the Italian Alps. *Landsc. Ecol. Eng.* **2017**, *13*, 305–318. <https://doi.org/10.1007/s11355-016-0323-5>.
17. Preti, F.; Crocetti, C.; Nijimbere, S.; Nsabiymva, J.M.V.; Hicintuka, C.; Castelli, G. Soil and Water Bioengineering (SWBE) in Africa: First Experience and Lessons-Learned in Burundi. *Ecol. Eng.* **2025**, *212*, 107510. <https://doi.org/10.1016/j.ecoleng.2024.107510>.
18. Tisserant, M.; Bourgeois, B.; González, E.; Evette, A.; Poulin, M. Controlling Erosion While Fostering Plant Biodiversity: A Comparison of Riverbank Stabilization Techniques. *Ecol. Eng.* **2021**, *172*, 106387. <https://doi.org/10.1016/j.ecoleng.2021.106387>.
19. Schmitt, K.; Schäffer, M.; Koop, J.; Symmank, L. River Bank Stabilisation by Bioengineering: Potentials for Ecological Diversity. *J. Appl. Water Eng. Res.* **2018**, *6*, 262–273. <https://doi.org/10.1080/23249676.2018.1466735>.
20. Rey, F.; Bifulco, C.; Bischetti, G.B.; Bourrier, F.; De Cesare, G.; Florineth, F.; Graf, F.; Marden, M.; Mickovski, S.B.; Phillips, C.; et al. Soil and Water Bioengineering: Practice and Research Needs for Reconciling Natural Hazard Control and Ecological Restoration. *Sci. Total Environ.* **2019**, *648*, 1210–1218. <https://doi.org/10.1016/j.scitotenv.2018.08.217>.
21. Gurnell, A.M.; Bertoldi, W.; Corenblit, D. Changing River Channels: The Roles of Hydrological Processes, Plants and Pioneer Fluvial Landforms in Humid Temperate, Mixed Load, Gravel Bed Rivers. *Earth. Sci. Rev.* **2012**, *111*, 129–141. <https://doi.org/10.1016/j.earscirev.2011.11.005>.
22. Corenblit, D.; Piégay, H.; Arrignon, F.; González-Sargas, E.; Bonis, A.; Davies, N.S.; Ebengo, D.M.; Garófano-Gómez, V.; Gurnell, A.M.; Henry, A.L.; et al. Interactions between Vegetation and River Morphodynamics. Part I: Research Clarifications and Challenges. *Earth. Sci. Rev.* **2024**, *253*.
23. Ghestem, M.; Cao, K.; Ma, W.; Rowe, N.; Leclerc, R.; Gadenne, C.; Stokes, A. A Framework for Identifying Plant Species to Be Used as “ecological Engineers” for Fixing Soil on Unstable Slopes. *PLoS ONE* **2014**, *9*, e95876. <https://doi.org/10.1371/journal.pone.0095876>.
24. Stokes, A.; Douglas, G.B.; Fourcaud, T.; Giadrossich, F.; Gillies, C.; Hubble, T.; Kim, J.H.; Loades, K.W.; Mao, Z.; McIvor, I.R.; et al. Ecological Mitigation of Hillslope Instability: Ten Key Issues Facing Researchers and Practitioners. *Plant Soil* **2014**, *377*, 1–23. <https://doi.org/10.1007/s11104-014-2044-6>.
25. Postma, J.A.; Schurr, U.; Fiorani, F. Dynamic Root Growth and Architecture Responses to Limiting Nutrient Availability: Linking Physiological Models and Experimentation. *Biotechnol. Adv.* **2014**, *32*, 53–65. <https://doi.org/10.1016/j.biotechadv.2013.08.019>.
26. Ghestem, M.; Sidle, R.C.; Stokes, A. The Influence of Plant Root Systems on Subsurface Flow: Implications for Slope Stability. *Bioscience* **2011**, *61*, 869–879. <https://doi.org/10.1525/bio.2011.61.11.6>.
27. Keita, N.; Bourgeois, B.; Evette, A.; Tisserant, M.; González, E.; Breton, V.; Goulet, C.; Poulin, M. Growth Response of Cuttings to Drought and Intermittent Flooding for Three Salix Species and Implications for Riverbank Soil Bioengineering. *Environ. Manag.* **2021**, *67*, 1137–1144. <https://doi.org/10.1007/s00267-021-01444-3>.
28. Kettenhuber, P.L.W.; Sousa, R.; Sutili, F. Vegetative Propagation of Brazilian Native Species for Restoration of Degraded Areas. *Floresta E Ambiente* **2019**, *26*, e20170956. <https://doi.org/10.1590/2179-8087.095617>.

29. Kettenhuber, P.L.W.; Oliveira, D.A.S.; Martins, S.V. Selecting Native Species for Soil and Water Bioengineering Techniques: Alternative to Restore Areas in Brumadinho, MG, Brazil. *Floresta E Ambiente* **2023**, *30*, e20220088. <https://doi.org/10.1590/2179-8087-FLORAM-2022-0088>.
30. Corenblit, D.; Steiger, J.; Gurnell, A.M.; Tabacchi, E.; Roques, L. Control of Sediment Dynamics by Vegetation as a Key Function Driving Biogeomorphic Succession within Fluvial Corridors. *Earth Surf. Process. Landf.* **2009**, *34*, 1790–1810. <https://doi.org/10.1002/esp.1876>.
31. Tisserant, M.; Janssen, P.; Evette, A.; González, E.; Cavaillé, P.; Poulin, M. Diversity and Succession of Riparian Plant Communities along Riverbanks Bioengineered for Erosion Control: A Case Study in the Foothills of the Alps and the Jura Mountains. *Ecol. Eng.* **2020**, *152*, 105880. <https://doi.org/10.1016/j.ecoleng.2020.105880>.
32. Janssen, P.; Dommanget, F.; Cavaillé, P.; Evette, A. Does Soil Bioengineering Benefits Aquatic Biodiversity? An Empirical Study of the Relative Influence of Local and Regional Drivers on Benthic Macroinvertebrate Assemblages. *Ecol. Eng.* **2021**, *168*, 106287. <https://doi.org/10.1016/j.ecoleng.2021.106287>.
33. Symmank, L.; Natho, S.; Scholz, M.; Schröder, U.; Raupach, K.; Schulz-Zunkel, C. The Impact of Bioengineering Techniques for Riverbank Protection on Ecosystem Services of Riparian Zones. *Ecol. Eng.* **2020**, *158*, 106040. <https://doi.org/10.1016/j.ecoleng.2020.106040>.
34. Prugne, M.; Corenblit, D.; Boivin, M.; Evette, A.; Buffin-Bélanger, T. Soil and Water Bioengineering in Cold Rivers: A Biogeomorphological Perspective. *Ecol. Eng.* **2024**, *204*, 107261. <https://doi.org/10.1016/j.ecoleng.2024.107261>.
35. Alvares, C.A.; Stape, J.L.; Sentelhas, P.C.; De Moraes Gonçalves, J.L.; Sparovek, G. Köppen's Climate Classification Map for Brazil. *Meteorol. Z.* **2013**, *22*, 711–728. <https://doi.org/10.1127/0941-2948/2013/0507>.
36. BDMET/INMET Banco de Dados Meteorológicos. Instituto Nacional de Meteorologia. Available online: <https://mapas.inmet.gov.br/> (accessed on: 15 December 2025).
37. Jardim Botânico do Rio de Janeiro—JBRJ Lista de Espécies Da Flora Do Brasil. Available online: <http://floradobrasil.jbrj.gov.br/> (accessed on: 20 December 2025).
38. Deng, L.; Zhu, G.Y.; Tang, Z.S.; Shangguan, Z.P. Global Patterns of the Effects of Land-Use Changes on Soil Carbon Stocks. *Glob. Ecol. Conserv.* **2016**, *5*, 127–138. <https://doi.org/10.1016/j.gecco.2015.12.004>.
39. Recking, A.; Piton, G.; Montabonnet, L.; Posi, S.; Evette, A. Design of Fascines for Riverbank Protection in Alpine Rivers: Insight from Flume Experiments. *Ecol. Eng.* **2019**, *138*, 323–333. <https://doi.org/10.1016/j.ecoleng.2019.07.019>.
40. Miele, P.; Di Martire, D.; Di Napoli, M.; Guerriero, L.; Calcaterra, D. Temporal Efficiencies of Soil Bioengineering Techniques to Mitigate Geo-Hydrological Risks. *Ecol. Eng.* **2021**, *170*, 106338. <https://doi.org/10.1016/j.ecoleng.2021.106338>.
41. Tardio, G.; Mickovski, S.B. A Novel Integrated Design Methodology for Nature-Based Solutions and Soil and Water Bioengineering Interventions: The Tardio&Mickovski Methodology. *Sustainability* **2023**, *15*, 3044. <https://doi.org/10.3390/su15043044>.
42. Valizade, N.; Tabarsa, A. Laboratory Investigation of Plant Root Reinforcement on the Mechanical Behaviour and Collapse Potential of Loess Soil. *Eur. J. Environ. Civ. Eng.* **2020**, *26*, 1475–1491. <https://doi.org/10.1080/19648189.2020.1715848>.
43. Weissteiner, C.; Schenkenbach, N.; Lammeranner, W.; Kalny, G.; Rauch, H.P. Cutting Diameter on Early Growth Performance of Purple Willow (*Salix purpurea* L.). *J. Soil Water Conserv.* **2019**, *74*, 380–388. <https://doi.org/10.2489/jswc.74.4.380>.
44. Weissteiner, C.; Rauch, H.P. The Architecture and Dynamic Properties of Young *Salix Purpurea* Trees in the Context of Riparian Engineering Systems. *Ecol. Eng.* **2018**, *123*, 161–167. <https://doi.org/10.1016/j.ecoleng.2018.09.014>.
45. Maffra, C.R.B.; Sousa, R.D.S.; Sutili, F.J.; Pinheiro, R.J.B. Evaluation of Live Cuttings Effect on Slope Stability. *Floresta e Ambiente* **2019**, *26*, e20170738. <https://doi.org/10.1590/2179-8087.073817>.
46. Bordoni, M.; Vivaldi, V.; Giarola, A.; Valentino, R.; Bittelli, M.; Meisina, C. Comparison between Mechanical and Hydrological Reinforcement Effects of Cultivated Plants on Shallow Slope Stability. *Sci. Total Environ.* **2024**, *912*, 168999. <https://doi.org/10.1016/j.scitotenv.2023.168999>.
47. Yazdani, F.; Sadeghi, H.; AliPanahi, P.; Gholami, M.; Leung, A.K. Evaluation of Plant Growth and Spacing Effects on Bioengineered Slopes Subjected to Rainfall. *Biogeotechnics* **2024**, *2*, 100080. <https://doi.org/10.1016/j.bgtech.2024.100080>.
48. Maffra, C.R.B.; Sousa, R.D.S.; Pinheiro, R.J.B.; Sutili, F.J. Evaluation of the Relationship between Morphological Characteristics and Pullout Resistance of Live Cuttings of *Phyllanthus Sellowianus* (Klotzsch) Müll.Arg. *Floresta* **2021**, *51*, 329. <https://doi.org/10.5380/rf.v51i2.65159>.
49. Liu, Y.; Rauch, H.P.; Zhang, J.; Yang, X.; Gao, J. Development and Soil Reinforcement Characteristics of Five Native Species Planted as Cuttings in Local Area of Beijing. *Ecol. Eng.* **2014**, *71*, 190–196. <https://doi.org/10.1016/j.ecoleng.2014.07.017>.
50. Fukao, T.; Barrera-Figueroa, B.E.; Juntawong, P.; Peña-Castro, J.M. Submergence and Waterlogging Stress in Plants: A Review Highlighting Research Opportunities and Understudied Aspects. *Front. Plant Sci.* **2019**, *10*, 340.

51. Markus-Michalczyk, H.; Hanelt, D. Willow Species Vary in Elevational Occurrence and Morphological Characteristics on the Tidal Freshwater Section of the Elbe Estuary. *Ecohydrol. Hydrobiol.* **2019**, *19*, 14–23. <https://doi.org/10.1016/J.ECOHYD.2018.08.006>.
52. Silva, A.C.; Higuchi, P.; Berg, E.V.D.; Nunes, M.H.; Carvalho, D.A.D. *Florestas Inundáveis: Ecologia, Florística e Adaptações Das Espécies*; Editora UFLA: Lavras, 2012; ISBN 9788581270128.
53. Zanandrea, I.; Alves, J.D.; Deuner, S.; Goulart, P.D.F.P.; Henrique, P.D.C.; Silveira, N.M. Tolerance of *Sesbania virgata* Plants to Flooding. *Aust. J. Bot.* **2009**, *57*, 661–669. <https://doi.org/10.1071/BT09144>.
54. Alves, J.D.; Zanandrea, I.; Deuner, S.; de F.P. Goulart, P.; de Souza, K.R.D.; de O. Santos, M. Antioxidative Responses and Morpho-Anatomical Adaptations to Waterlogging in *Sesbania virgata*. *Trees-Struct. Funct.* **2013**, *27*, 717–728. <https://doi.org/10.1007/s00468-012-0827-z>.
55. Köhler, M.; Bernardes, M.G.; Brack, P. Espécies Reófitas. In *Flora da Bacia do Rio Pelotas: Uso e Conservação de Espécies RS-Brasil*; UFRGS: Porto Alegre, Brazil, 2016; pp. 145–170.
56. Aximoff, I.A.; Soares, H.M.; Bernadello, G. *Acnistus Arborescens* (Solanaceae): An Important Food Resource for Birds in an Atlantic Forest Site, Southeastern Brazil. *Rodriguésia* **2020**, *71*, e02232018. <https://doi.org/10.1590/2175-7860202071030>.
57. Delarmelina, W.M.; Caldeira, M.V.W.; Faria, J.C.T.; De Oliveira Gonçalves, E.; Rocha, R.L.F. Diferentes Substratos Para a Produção de Mudanças de *Sesbania virgata*. *Floresta E Ambiente* **2014**, *21*, 224–233. <https://doi.org/10.4322/floram.2014.027>.
58. Ferreira, L.V.M.; Nóbrega, R.S.A.; Silva, G.C.; Nóbrega, J.C.A.; Moreira, F.M.S. Crescimento e Nodulação de *Sesbania virgata* Com Estirpes Nativas e Introduzidas. *Rev. De Ciências Agrar.-Amaz. J. Agric. Environ. Sci.* **2015**, *58*, 327–334. <https://doi.org/10.4322/rca.1881>.
59. Cortezi, D.G.; Colli, S. Efeito de Alagamento e de Aplicação de Fitorreguladores Na Brotação de Guazuma Ulmifolia (Malvaceae) e de *Sesbania virgata* (Fabaceae). *Rev. Bras. De Botânica* **2011**, *34*, 423–430. <https://doi.org/10.1590/s0100-84042011000300015>.
60. Sánchez-Soto, S.; Guedes, J.C.; Nakano, O. *Neodiplogrammus Quadrivittatus* (Olivier) (Coleoptera: Curculionidae) No Estado de São Paulo. *Neotrop. Entomol.* **2003**, *32*, 511–512. <https://doi.org/10.1590/S1519-566X2003000300022>.
61. Suttili, F.J.; Denardi, L.; Durlo, M.A.; Rauch, H.P.; Weissteiner, C. Flexural Behaviour of Selected Riparian Plants under Static Load. *Ecol. Eng.* **2012**, *43*, 85–90. <https://doi.org/10.1016/j.ecoleng.2012.02.012>.
62. Suttili, F.J.; Dorneles, R.D.S.; Vargas, C.O.; Kettenhuber, P.L.W. Avaliação Da Propagação Vegetativa de Espécies Na Estabilização de Obras de Terra Com Técnicas de Engenharia Natural. *Ciência Florest.* **2018**, *28*, 1–12.
63. Rivaes, R.; Pinheiro, A.N.; Egger, G.; Ferreira, T. The Role of River Morphodynamic Disturbance and Groundwater Hydrology As Driving Factors of Riparian Landscape Patterns in Mediterranean Rivers. *Front. Plant Sci.* **2017**, *8*, 1612. <https://doi.org/10.3389/fpls.2017.01612>.
64. Corenblit, D.; Vidal, V.; Cabanis, M.; Steiger, J.; Garófano-Gómez, V.; Garreau, A.; Hortobágyi, B.; Otto, T.; Roussel, E.; Voldoire, O. Seed Retention by Pioneer Trees Enhances Plant Diversity Resilience on Gravel Bars: Observations from the River Allier, France. *Adv. Water Resour.* **2016**, *93*, 182–192. <https://doi.org/10.1016/j.advwatres.2016.02.015>.
65. Xu, S.; Eisenhauer, N.; Zeng, Z.; Mo, X.; Ding, Y.; Lai, D.Y.F.; Wang, J. Drivers of Soil Organic Carbon Recovery under Forest Restoration: A Global Meta-Analysis. *Carbon Res.* **2024**, *3*, 80. <https://doi.org/10.1007/s44246-024-00165-6>.
66. Wang, K.; Ma, Z.; Qin, W.; Li, X.; Shi, H.; Hasi, B.; Liu, X. Soil Nutrients and PH Modulate Carbon Dynamics in Particulate and Mineral-Associated Organic Matter during Restoration of a Tibetan Alpine Grassland. *Ecol. Eng.* **2025**, *212*, 107522. <https://doi.org/10.1016/j.ecoleng.2025.107522>.
67. Filipović, A.; Perčin, A.; Hadžibulić, A.; Mandić, A. Transformation of Organic Matter and Impact on the Ecosystem. In *Agroforestry for Carbon and Ecosystem Management*; Elsevier: Amsterdam, The Netherlands, 2024; pp. 311–329. <https://doi.org/10.1016/B978-0-323-95393-1.00018-X>.
68. Bieluczyk, W.; Merloti, L.F.; Cherubin, M.R.; Mendes, L.W.; Bendassolli, J.A.; Rodrigues, R.R.; de Camargo, P.B.; van der Putten, W.H.; Tsai, S.M. Forest Restoration Rehabilitates Soil Multifunctionality in Riparian Zones of Sugarcane Production Landscapes. *Sci. Total Environ.* **2023**, *888*, 164175. <https://doi.org/10.1016/j.scitotenv.2023.164175>.
69. Petrone, A.; Preti, F. Suitability of Soil Bioengineering Techniques in Central America: A Case Study in Nicaragua. *Hydrol. Earth Syst. Sci.* **2008**, *12*, 1241–1248. <https://doi.org/10.5194/hess-12-1241-2008>.

Disclaimer/Publisher's Note: The statements, opinions and data contained in all publications are solely those of the individual author(s) and contributor(s) and not of MDPI and/or the editor(s). MDPI and/or the editor(s) disclaim responsibility for any injury to people or property resulting from any ideas, methods, instructions or products referred to in the content.