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A 15-Year Multi-Temporal Assessment of Tropical Peatland Restoration in Sumatra: Landscape Dynamics, Hydrological Drivers, and Vegetation Outcomes at Kebun Raya Sriwijaya (2010-2025)

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ABSTRACT

Tropical peatland restoration has been dominated by rewetting approaches with narrow carbon-focused indicators. This 15-year multi-temporal assessment (2010–2025) of Kebun Raya Sriwijaya (KRS)—a converted oil palm plantation turned botanical garden on Sumatran peatland—examined three thematic domains: landscape-hydrological change, vegetation restoration and conservation outcomes, and botanical-garden governance, using satellite imagery, hydrological monitoring, survival assessments, and peat depth measurements. Continuous hydrological monitoring was limited to a 12-month post-intervention period (September 2020–August 2021); pre-intervention conditions were inferred from fire events, regional rainfall patterns, and satellite observations. Results showed that during this monitoring period, the post-intervention monitoring data suggested a reduced contemporaneous association between rainfall and water-table depth, although the 12-month dataset did not permit causal inference, an extreme wet event in early 2024 was associated with prolonged inundation and vegetation dieback. For the most recent cohorts (planted in 2022–2023), post-intervention two-year survival rates exceeded 95%, a substantial improvement over pre-intervention cohorts (40–64%) and the transitional 2021 cohort (50.4%). The garden conserves native species listed on the International Union for Conservation of Nature (IUCN) Red-listed species, with village cemeteries serving as unrecognized refugia. Institutional permanence enabled learning from fire and flood setbacks—a capacity absent from time-bound projects. However, external landscape drivers (adjacent drainage, fires) persistently affected site conditions, revealing fundamental limits to site-scale restoration. The botanical garden model sacrifices efficiency and scalability but offers complementary advantages for strategic locations near population centers where biodiversity, livelihoods, and long-term governance are priorities. Anomalous peat depth increases (up to +293 cm) were observed between 2023 and 2025, but these values far exceed plausible peat accumulation rates. Given that measurements were taken within 12 permanent 20 m x 20 m plots (distributed across KRS) but without permanent markers (e.g., subsidence poles) at each sampling point or bulk density measurement, these data are best interpreted as reflecting methodological uncertainty (spatial heterogeneity or measurement artifacts) rather than net peat growth or carbon sequestration.

1. INTRODUCTION

Tropical peatlands are among the earth's most carbon-dense ecosystems, yet they have been extensively degraded by land-use change, contributing significantly to global greenhouse gas emissions [1]. In response to the growing international focus on climate change, large-scale peatland restoration initiatives have been widely undertaken, especially across tropical regions. Indonesia, home to approximately 36% of the world's tropical peatlands, has committed to restoring over 1.2 million hectares of degraded peatlands through the establishment of the Peatland Restoration Agency (BRG) in 2016, later integrated into the National Peatland and Mangrove Restoration [2].

A substantial body of research has underscored the environmental [3, 4], biodiversity [5], economic [6], and social [7] aspects of peatland restoration efforts. However, to the best of our knowledge, no study has explicitly examined this topic through the lens of botanical garden functions. Globally, the role of botanical gardens has undergone significant evolution. Contemporary international frameworks recognize botanical gardens as multifunctional institutions whose contributions extend beyond traditional conservation and research [8] to encompass community engagement, sustainable resource utilization, and cultural heritage preservation [9]. Moreover, botanical gardens are increasingly positioned as key actors in climate change mitigation and adaptation strategies, including ecosystem restoration initiatives such as peatland rehabilitation [10].

These international developments have found concrete institutional expression in Indonesia, where botanical gardens (Kebun Raya) are formally mandated with five core functions: ex-situ conservation, research, environmental education, eco-tourism, and environmental services [11]. These functions reflect a holistic approach that integrates biodiversity preservation with public engagement and ecosystem-based benefits. Despite the growing recognition of botanical gardens as strategic platforms for integrated landscape management, their potential role in peatland restoration contexts remains unexplored. This gap is particularly salient given that peatland restoration projects often operate across multiple domains—ecological, social, and economic—that align closely with the multifunctional mandates of botanical gardens.

This study took Kebun Raya Sriwijaya (KRS) as a unique case to address this lacuna. Unlike typical peatland restoration sites that are often located within protected forest areas or remote coastal zones with limited accessibility, KRS presents a distinctive landscape context. The site was established from scratch through the conversion of an existing oil palm plantation—a land-use transition that represents a deliberate shift from extractive monoculture toward conservation-oriented multifunctionality. Furthermore, its location in proximity to residential areas, rather than in isolated peatland landscapes, enables the observation of simultaneous landscape transformation alongside dynamic human–environment interactions. This setting renders KRS an exemplary case for examining how botanical gardens function can be operationalized within a peatland restoration framework under conditions of high accessibility, active community engagement, and ongoing land-use change.

This study aims to address three interconnected thematic objectives based on 15 years of data from KRS: (1) Landscapehydrological change - to investigate how the landscape of KRS evolved from 2010 to 2025 using multi-temporal satellite imagery, and how peat hydrological dynamics responded to both climate variability and anthropogenic interventions (installation of water retention infrastructure in 2020); (2) Vegetation restoration and conservation outcomes - to quantify survival patterns of native peatland species planted across different cohorts (2011-2023) in relation to prevailing climate conditions, and to document the botanical garden's conservation function including International Union for Conservation of Nature (IUCN)-Red-listed species from culturally protected landscapes; and (3) Botanical-garden governance and multifunctionality - to analyze how institutional permanence enable learning from fire and flood setbacks, and to identify integrated lessons for peatland restoration design, particularly the trade-offs between site-scale interventions and external landscape drivers. Supporting analyses address socio-economic functions (natural dye utilization, eco-tourism, environmental education) and peat surface dynamics (repeat depth measurements 2023–2025), which inform the interpretation of governance outcomes and carbon implications, respectively.

2. METHODS

2.1 Study area

This study was conducted at KRS, a botanical garden located in Ogan Ilir Regency, South Sumatra Province, Indonesia (3°9'31.56"S; 104°32'44"E). The site covers approximately 100 ha and was situated within the Sungai Musi-Sungai Blidah Peat Hydrological Unit (KHG.16.03-07.01), which spans approximately 31,167 ha [12]. The broader hydrological unit represents a functionally connected peatland landscape where surface and subsurface water flows are hydrologically interdependent.

Land-use history and surrounding landscape. Before its designation as a botanical garden, the site was an active oil palm plantation from the 1990s until 2010. Between 2010 and 2015, the site remained in a transitional state with no active restoration, during which time existing drainage canals from the plantation period remained open. To the northeast of the KRS boundary, a coal haul road became visible in 2021 satellite imagery, indicating ongoing extractive land use adjacent to the site. Between 2021 and 2024, a parcel of land previously characterized by shrubland immediately northeast of the boundary was converted into an oil palm plantation. The drainage canals from this adjacent plantation are oriented perpendicular to the KRS boundary, directing water flow toward the botanical garden.

Hydrological infrastructure. In 2020, KRS management completed the installation of hydrological infrastructure designed to regulate water levels within the site. This infrastructure includes: (i) a perimeter canal surrounding the majority of the garden area, (ii) inlet and outlet gates that control water exchange between the perimeter canal and the external drainage network, and (iii) multiple retention ponds distributed across the site to store water during wet periods and release it gradually during dry periods.

2.2 Satellite imagery analysis

Landscape transformation at KRS was analyzed using multi-temporal satellite imagery sourced from Google Earth Pro. Four image acquisitions representing different stages of restoration were selected based on cloud-free conditions and temporal relevance: September 2014 (pre-restoration), February 2017 (early restoration period), July 2021 (mid-restoration), and November 2024 (late restoration). Pixel-based contrast analysis was performed to interpret land cover types, linear features (canals, roads, paleochannels), and boundary changes.

2.3 Peat depth measurement

Peat depth was measured in 2023 and 2025 at 12 sampling points (TS1 to TS14, excluding TS7 and TS11) distributed across KRS. Each sampling point was located within a separate permanent plot of 20 m × 20 m (Figure 1), with three replicates taken per point. Although no permanent above-ground markers (e.g., subsidence poles) were installed at the exact sampling points to enable precise repositioning, all measurements were consistently taken within the same 12 designated permanent plots during both surveys. A standard peat auger was used to extract peat cores vertically until resistance indicated the mineral substrate or dense root layer. Points were categorized based on management history:

- Undisturbed secondary forest (no clearing for ≥10 years): TS9 to TS10 and TS12 to TS14 (n = 5 points).
- Actively managed areas (regular weeding, planting, foot traffic): TS1 to TS6 and TS8 (n = 7 points).

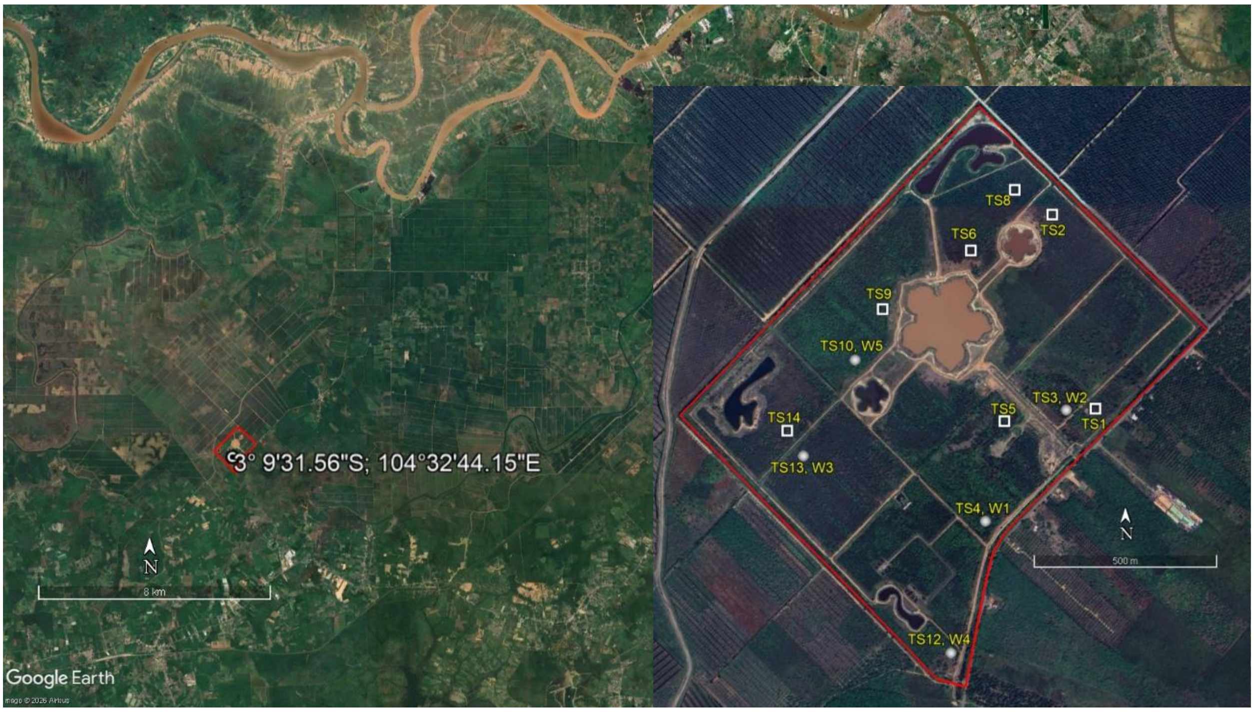

Figure 1. Relative position of Kebun Raya Sriwijaya (KRS) within land use (26 August 2025) and inset showing indicative peat depth sampling points (TS1 to TS14, excluding TS7 and TS11) and water table monitoring wells location (W1 to W5) (3 November 2024)

Note: The circle mark represents peat depth sampling points and the water table monitoring well. The squares represent peat depth sampling points.
Source: Google Earth Pro, accessed 7 May 2026.

Table 1. Vegetation type of permanent water table monitoring wells at Kebun Raya Sriwijaya (KRS)

Well No.	Vegetation Type	Management Status	Hydrological Zone	Distance from Ditch (m)
W1	*Shorea balangeran*	managed area	canal riparian	37
W2	*Fagraea fragrans*	managed area	flood-prone transitional	16
W3	mixed vegetation	secondary forest	flood-prone transitional	4
W4	mixed vegetation	secondary forest	canal riparian	2
W5	mixed vegetation	secondary forest	flood-prone transitional	4

2.4 Hydrological data collection

Post-intervention monitoring was conducted from September 2020 to August 2021 at five permanent wells (4-inch polyvinyl chloride, 200 cm depth, 50 cm above peat surface) distributed across different vegetation types. Each well was positioned at a minimum distance of 2 meters from the nearest drainage canal to avoid direct surface runoff effects. Table 1 describes the location, vegetation type, and represented zone for each well. The five wells were distributed across different vegetation types and hydrological zones within the 100-ha site: managed area, canal riparian (W1); managed area, flood-prone transitional area (W2); secondary forest, flood-prone transitional area (W3); secondary forest, canal riparian (W4); and secondary forest, flood-prone transitional area (W5). While the well locations cover the main vegetation types, we acknowledge that the 2-meter distance from canals may limit the representativeness of areas closer to drainage features. Well locations are shown in Figure 1.

Depth to water table was measured manually on the first Wednesday of each month. Monthly rainfall was recorded using a standard ombrometer. The 12-month monitoring period was limited by logistical constraints; this dataset represented the only continuous water table record available. For the pre-intervention period (2014–2019), hydrological conditions were inferred from fire events (2015, 2019), regional rainfall patterns, and satellite-observed surface wetness. Longer-term monitoring is ongoing and will be reported in future communications.

Pearson's correlation coefficient was used to examine the relationship between monthly rainfall and mean depth to water table (n = 12 months, September 2020 – August 2021). For each month, the water table depth was averaged across the five monitoring wells before correlation analysis. Statistical significance was set at $p < 0.05$. Analyses were conducted using SPSS version 23.

2.5 Vegetation survival assessment

Survival data were obtained from KRS's annual planting records from 2011 to 2023 and annual assessment records from 2013 to 2025. Each planted specimen was georeferenced and monitored annually. The two-year survival rate was defined as the proportion of planted specimens still alive 24 months after planting. A specimen was recorded as dead when no living above-ground tissue was observed during two consecutive annual surveys; specimens that could not be relocated due to missing or damaged tags were excluded from the analysis. For each cohort, the climate condition at maturation (24 months after planting) was assigned based on El Niño–Southern Oscillation (ENSO) phase records from the Indonesian Agency for Meteorology, Climatology and Geophysics [13].

2.6 Collection provenance and International Union for Conservation of Nature Red List status determination

Specimen origins were recorded in KRS's annual log reports (2013–2025). Sources included: (1) Bogor Botanic Gardens seed bank (2013); (2) plant exploration in South Sumatran secondary forests and peat swamps (2015–2025); and (3) village cemetery areas in three villages (2019–2021). Data recorded for each event included date, coordinates, collector name, local name (from local guides), and habitat. Voucher specimens were deposited at the KRS herbarium. The conservation status of each plant species was determined based on the IUCN Red List of Threatened Species. Status categories (Near Threatened (NT), Vulnerable (VU), Endangered (EN), and Critically Endangered (CR)) follow the IUCN Red List Categories and Criteria Version 3.1. Species statuses were verified through the IUCN Red List online database (https://www.iucnredlist.org) accessed in April 2026. For each species, the most recent assessment available was used.

2.7 Ethical procedures and data sources

A standardized ethical protocol was applied to all plant exploration activities, whether in village cemeteries or secondary forests. Formal written requests were submitted to village heads, followed by face-to-face meetings to obtain verbal informed consent. A resident served as a field guide for each collection event, assisting in site location and provision of local plant names. This ensured community participation and respect for local knowledge across all collection sites. To provide transparency regarding the multiple datasets used in this study, Table 2 summarizes the year, sample size, frequency, purpose, and limitations of each data type.

Table 2. Summary of data sources, sample characteristics, and analytical framework used in this study

Data Type	Year(s)	Sample Size (n)	Frequency	Purpose	Limitations
Satellite imagery (Google Earth Pro)	2014, 2017, 2021, 2024	4 images	Single acquisition per year	Landscape transformation analysis	Cloud cover in 2014 image; no species-level identification
Peat depth measurements	2023, 2025	36 measurements per survey years; 72 measurements across the two survey years.	Two repeated measurements	Spatial heterogeneity assessment	No fixed markers per point (plots are permanent); no bulk density; values may reflect artifacts
Water table depth	Sept 2020-Aug 2021	5 wells × 12 months = 60 measurements	Monthly	Hydrological dynamics assessment	Only 12-month post-intervention; no pre-intervention control
Monthly rainfall	Sept 2020-Aug 2021	12 monthly totals	Monthly	Correlation with water table	Limited to monitoring period
Planting and assessment records	2011-2025	11 cohorts (planting year: 2011-2023, no planting in 2012 and 2014; assessment year 2013-2025)	Annual monitoring	Survival rate calculation	Missing tags in early cohorts
IUCN Red List status	2026	12 native species	Single verification	Conservation status determination	Global assessment, not local population
Visitor records	2018-2025	8-year average	Annual	Eco-tourism function documentation	No formal economic impact assessment
Natural dye utilization	2023-2025	8 species	Ongoing	Socio-economic function documentation	Qualitative documentation only
Research activities	2015-2025	38 activities	Cumulative	Knowledge development documentation	Not all activities are published

3. RESULTS

3.1 Landscape-hydrological change

3.1.1 Landscape transformation (2014-2024)

Visual interpretation of satellite imagery from 2014 to 2024 suggested a trajectory of landscape transformation at KRS. In 2014, before the commencement of physical restoration activities, the site exhibited linear planting patterns and geometric blocks characteristic of active oil palm cultivation (Figure 2(A)). Due to cloud cover in the 2014 imagery, the 2017 imagery—which offered clear atmospheric conditions—

served as the baseline for identifying paleochannel traces. The 2017 imagery showed visible traces of a former river channel—a paleochannel—intersecting the site (Figure 2(B)). These features persisted despite the presence of drainage canals constructed during the oil palm plantation period.

A coal haul road was first visible in 2021 along the northeastern periphery (Figure 2(C)). By 2024, following a decade of restoration activities, the landscape had been substantially reconfigured. The former oil palm blocks had been replaced by a mosaic of land cover types: areas of secondary forest had emerged in previously cultivated zones, while designated planting blocks then contained native peatland species arranged in structured collections (Figure 2(D)). The paleochannel traces that remained visible in 2017 had become substantially less discernible by 2024, indicating the progressive obliteration of the natural watercourse that once traversed the site. This transformation was accompanied by the construction of water retention ponds designed to regulate water storage and modulate hydrological flows within the botanical garden.

Concurrent with these internal changes, landscape transformation was also observed in the area adjacent to the northeastern boundary of the site. During the same period, a parcel of land previously characterized by shrubland was converted into an oil palm plantation, with drainage canals oriented perpendicular to the KRS boundary, directing water flow toward the botanical garden (Figure 2(D)). The area highlighted by the red oval in Figure 2(D) corresponded to the flood-prone zone, where prolonged inundation was observed during the 2024 wet period.

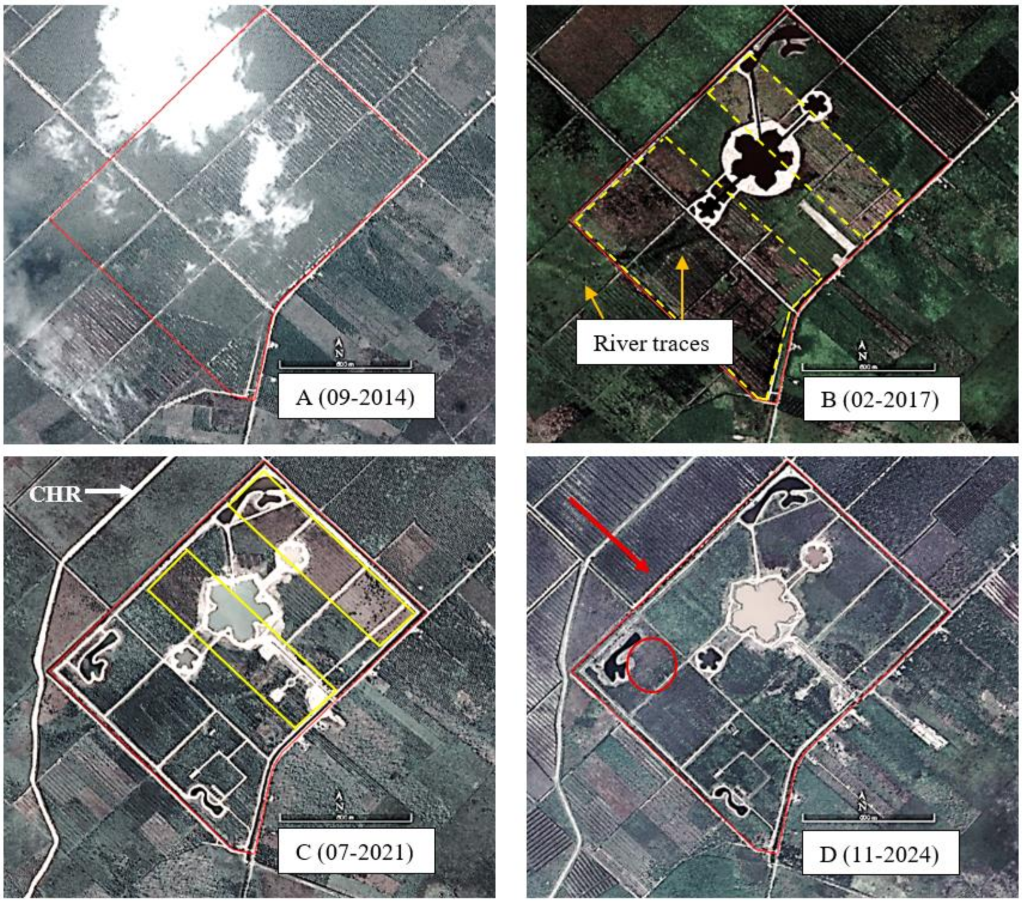

Figure 2. Multi-temporal satellite imagery of Kebun Raya Sriwijaya (KRS) showing landscape transformation from oil palm plantation to botanical garden (2014–2024). (A) 2014: Active oil palm plantation, (B) 2017: Paleochannel traces visible, (C) 2021: Coal haul road (CHR) appears; adjacent area (upper left) unchanged, (D) 2024: Botanical garden established Adjacent shrubland converted to an oil palm plantation with drainage canals oriented perpendicular to KRS boundary (red arrow). Red oval: flood-affected area. Source: Google Earth Pro imagery (accessed March 2026).

3.1.2 Hydrological dynamics and infrastructure effects

Direct water table measurements (Table 3) are available for a 12-month post-intervention period (September 2020–August 2021). For the pre-intervention period (2014–2019), hydrological conditions are inferred from fire events, regional rainfall patterns, and satellite-observed surface wetness. The hydrological dynamics of the study site over the 2014–2024 period were shaped by both climate variability and anthropogenic interventions. For the pre-intervention period (2014–2019), direct water table measurements were not available; however, the El Niño events of 2015 and 2019 resulted in prolonged regional drought conditions that led to fire events within the site on both occasions, indicating severe water table drawdown. The installation of water retention ponds and a perimeter canal with inlet and outlet gates, completed in 2020, fundamentally altered the site's hydrological regime. Continuous monitoring from September 2020 to August 2021 (Figure 3) showed that following this intervention, the water table declined only gradually during the dry period (February to August 2021). Rainfall reached a minimum of 60 mm in June 2021, while water-table depth gradually increased from 29.8 cm in February to 46.0 cm in August, preliminarily consistent with a buffering effect provided by the retention infrastructure.

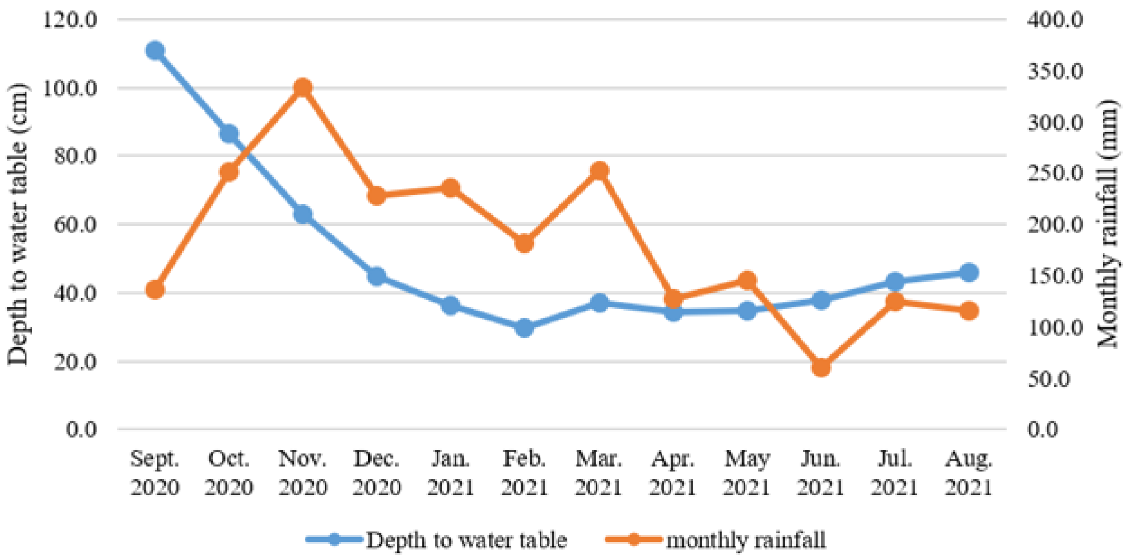

Figure 3. Post-intervention depth to water table and monthly rainfall (Sept. 2020 - Aug. 2021)
Source: in-situ monitoring.

Table 3. Individual depth to water table measurements

Well No.	Depth to Water Table (cm)											
	2020				2021							
	Sept.	Oct.	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	Jun.	Jul.	Aug.
1	115	78	60	40	33	27	37	28	33	32	40	42
2	103	79	58	42	33	27	35	30	32	35	40	44
3	112	90	67	44	38	33	35	33	35	40	43	44
4	110	93	65	46	38	30	39	38	35	44	47	50
5	114	92	64	51	40	32	39	43	38	38	47	50
Mean ± SD	110.8 ± 4.8	86.4 ± 7.3	62.8 ± 3.7	44.6 ± 4.2	36.4 ± 3.2	29.8 ± 2.8	37.0 ± 2.0	34.4 ± 6.1	34.6 ± 2.3	37.8 ± 4.6	43.4 ± 3.5	46.0 ± 3.7

Additional data from July 2025 further supported this interpretation: with monthly rainfall of 75 mm (lower than the 124.7 mm observed in 2021), the water table was recorded at 38 cm—within the same range as the 2021 dry period. This suggested that the buffering capacity had been maintained, and in relative terms, hydrological regulation appeared to have improved, as a lower rainfall input produced a comparable water table level. It is important to note, however, that this analysis was based on 12 months of post-intervention data supplemented by a single observation in 2025; longer-term monitoring would be required to confirm the persistence of this buffering capacity under varying climate extremes.

However, the same infrastructure, by impounding water, may have also contributed to prolonged inundation under extreme wet conditions. La Niña conditions in early 2024 sustained high water levels that exceeded the peat surface, leading to flood events with standing water persisting from the third week of January to the second week of March (7 weeks). Both extremes—drought-induced fire (pre-intervention) and prolonged inundation (post-intervention)—contributed to localized vegetation mortality at different temporal periods of the 15-year restoration timeline.

Over the 12-month monitoring period, the correlation between monthly rainfall and mean water table depth was weak and non-significant ($r = 0.159$, $p = 0.621$, $n = 12$). The remaining variation was likely attributable to the buffering effect of the retention infrastructure, although the limited sample size ($n = 12$) and lack of a pre-intervention control period preclude strong causal inference.

3.2 Vegetation restoration and conservation outcomes

3.2.1 Survival patterns across planting cohort (2011-2023)

The hydrological extremes described in Section 3.1.2—particularly the 2024 flood event—resulted in observable vegetation dieback (Figure 2(C)-2(D)). Pixel-based contrast analysis between the pre-flood (July 2021) and post-flood (November 2024) imagery showed visible changes in vegetation cover in the flood-affected zone. Field observation following the flood event confirmed mortality of several dominant taxa, particularly *Dicranopteris linearis* (fern), *Scleria sumatrensis* (graminoids), and *Melastoma malabathricum* (shrub). The satellite imagery alone, however, does not provide species-level identification; these interpretations are based on ground-truthing conducted after the flood. Vegetation dieback was observed across the site, not exclusively within the area shown in Figure 2(C)-2(D).

Beyond naturally occurring vegetation, the botanical garden had also established planted collections as part of its ex-situ conservation mandate. Survival rates of planted specimens varied considerably by planting year (Figure 4). During the pre-intervention period, two-year survival rates were consistently lower for cohorts that matured under La Niña-influenced (2017 and 2021 assessment years) than for those that matured under El Niño-influenced conditions (2015 and 2019 assessment years). However, we caution against direct attribution to ENSO alone, as other factors—including fire events (2015, 2019), species composition, planting number, and maintenance intensity—also varied across cohorts and could not be fully controlled in this long-term observational study. Under regulated conditions following the 2020 infrastructure completion, the influence of climate phases on survival weakened substantially, with the 2022 and 2023 cohorts exhibiting two-year survival rates exceeding 95.7%.

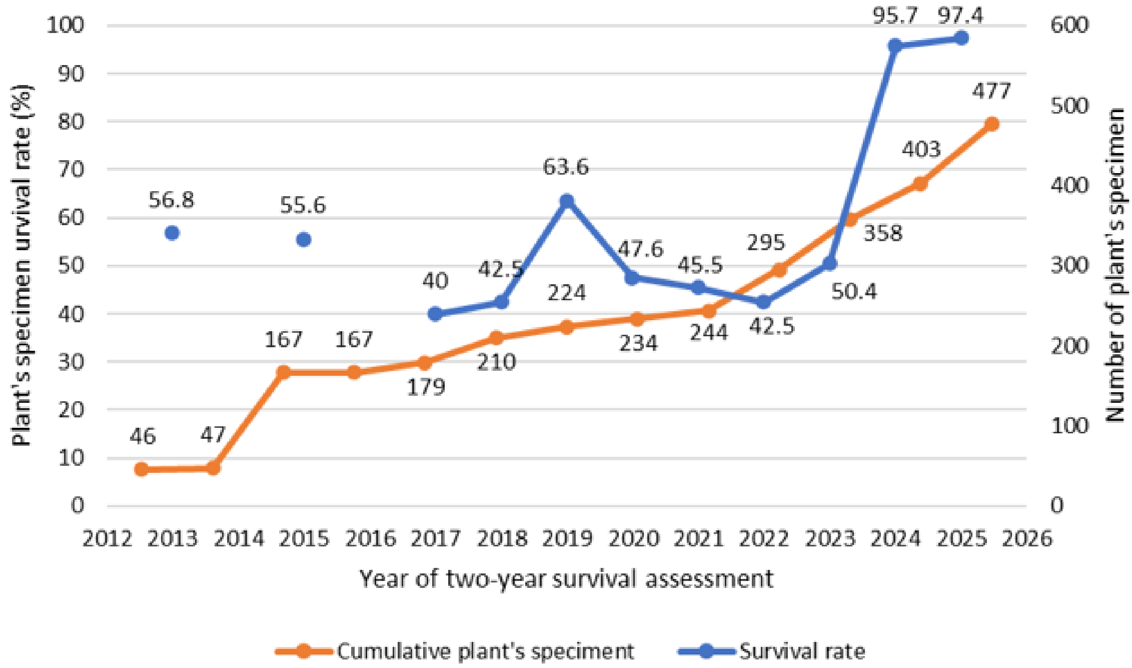

Figure 4. Cumulative living specimens and two-year survival rate by survival-assessment year
Survival was lower for cohorts that matured under La Niña conditions (2017-2018, 2020) compared to El Niño conditions (2015, 2019). Following the hydrological regulation (2020), survival exceeded 95.7% for the 2022 cohort onward. The 2024 and 2025 survival values represent the two-year outcomes of the 2022 and 2023 planting cohorts, respectively.
Source: in-situ observation.

Table 4. High conservation value species (IUCN Red List: NT to CR) managed by Kebun Raya Sriwijaya (KRS), all of which are native to Sumatran forests and peat swamps

No.	Plant's Species	IUCN Status	Collection Provenance
1	*Aquilaria malaccensis* Lam.	Critically Endangered	Secondary forest
2	*Shorea ovalis* (Korth.) Blume	Endangered	Secondary forest
3	*Syzygium zeylanicum* (L.) DC.	Endangered	Village cemetery
4	*Agathis dammara* (Lamb.) Rich. & A.Rich.	Vulnerable	Secondary forest
5	*Ctenolophon parvifolius* Oliv.	Vulnerable	Village cemetery
6	*Dyera polyphylla* (Miq.) Steenis	Vulnerable	Secondary forest
7	*Hopea mengarawan* Miq.	Vulnerable	Village cemetery
8	*Tetramerista glabra* Miq.	Vulnerable	Secondary forest
9	*Artocarpus kemando* Miq.	Near Threatened	Bogor seed bank
10	*Intsia bijuga* (Colebr.) Kuntze	Near Threatened	Village cemetery
11	*Nephelium juglandifolium* Blume	Near Threatened	Village cemetery
12	*Shorea pauciflora* King	Near Threatened	Secondary forest

Note: IUCN: International Union for Conservation of Nature; NT: Near Threatened; CR: Critically Endangered; IUCN status retrieved from the IUCN Red List of Threatened Species [14].

3.2.2 Conservation function: International Union for Conservation of Nature Red-listed species and culturally protected landscapes

As of December 2025, the living collection comprised 477 cultivated specimens, representing 117 species, 95 genera, and 49 families (the complete species list can be retrieved at www.makoyana.brin.go.id). Of these, 12 native Sumatran species are listed on the IUCN Red List (Table 4), ranging from Near Threatened to Critically Endangered. These 12 species represent 2.22% of the 540 forest and wetland plant species reported for Sumatra [14]. The strategic value of this collection is further elaborated in the Discussion.

Among these 12 native IUCN Red-listed species, five—*Syzygium zeylanicum*, *Ctenolophon parvifolius*, *Hopea mengarawan*, *Intsia bijuga*, and *Nephelium juglandifolium* (Species No. 3, 5, 7, 10, and 11 in Table 4)—were obtained from village cemetery areas across three different villages in South Sumatra, rather than from secondary forests. This finding illustrates that culturally protected landscapes outside conventional forest reserves can serve as refugia for threatened species, and that KRS has successfully translocated these genetic resources back onto peatland.

3.3 Supporting analysis

3.3.1 Peat surface dynamics (repeat depth measurements)

Peat depth measurements from 2023 and 2025 showed considerable inter-point variability (Figure 5). Among the five undisturbed secondary forest points, four (TS9, TS10, TS13, TS14) exhibited substantial increases in peat depth ranging from +83 cm to +293 cm over the two years, while one point (TS12) showed a slight decrease of -109 cm. Among the seven managed points, changes were also variable: TS1 and TS5 showed large increases (+192 cm and +134 cm, respectively), TS2 and TS4 showed moderate decreases (-22 cm and -16 cm), and the remaining points showed minor changes (ranging from -2 cm to +23 cm).

These results did not support a simple distinction between undisturbed and managed areas. Instead, they suggested that local factors—such as proximity to water sources, microtopography, vegetation composition, or past disturbance intensity—might exert stronger control on peat depth dynamics than broad management categories. The large increases in measured peat depth at several points (e.g., TS14: +293 cm over two years) exceeded expected accretion rates for tropical peatlands (typically <2 cm yr^{-1}). Accordingly, these data are not intended as quantitative estimates of net peat accumulation, but rather to highlight the spatial heterogeneity and methodological challenges inherent in repeat peat depth measurements within the study area.

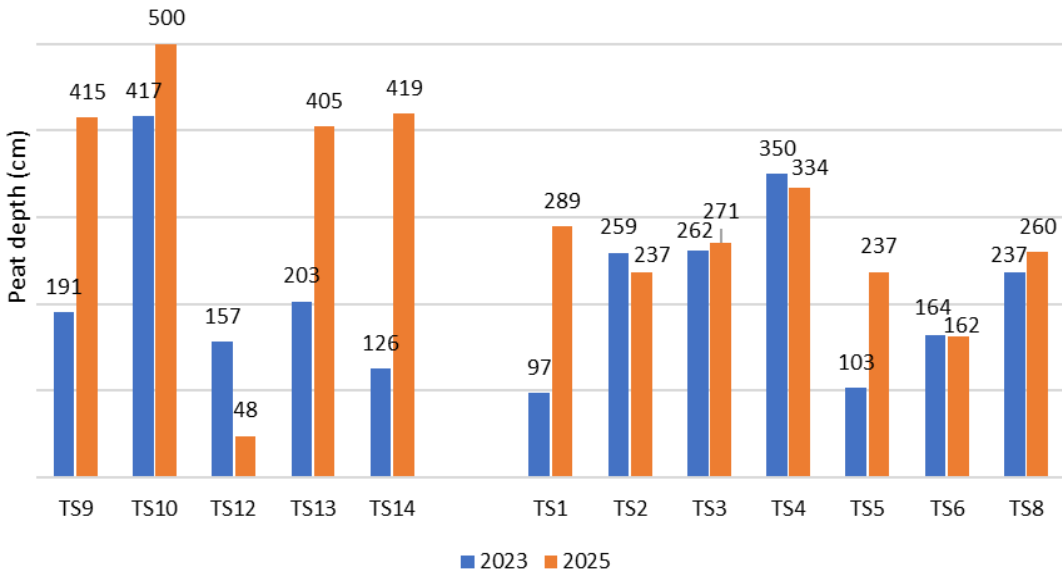

Figure 5. Peat depth in undisturbed secondary forest (TS9 to TS14, exclude TS11; n = 5) and managed areas (TS1 to TS8, exclude TS7; n = 7) at Kebun Raya Sriwijaya (KRS), 2023 vs 2025

Notes: Large increases at several points exceed plausible accretion rates for tropical peatlands and likely reflect spatial heterogeneity between sampling points or compaction artifacts in 2023 measurements. These data are presented to illustrate local variability, not as quantitative estimates of net peat accumulation.

3.3.2 Socio-economic and public engagement functions

As early as its initial full operation, KRS actively built public engagement in accordance with its mandate functions. Since 2018, KRS has received approximately 2,100 visitors annually (2018-2025 average), with children constituting 27.8% of total visitors. Annual visitation generated approximately 7-12 million rupiah in retribution income for the government. These data were derived from ticketing records and annual management archives maintained by KRS. KRS provided visitors with experiences in natural dyes and eco-printing as a form of environmental education, complementing the aesthetic value of its landscape. As of 2025, eight native peatland species had been investigated for their chromatic properties and utilized as natural dyes. Among them, *Shorea balangeran* and *Lagerstroemia speciosa* were two species that had been introduced to traditional textile (wastra) artisans in South Sumatra since 2023. The ability of KRS to preserve native species, undertake scientific investigation, and promote their utilization as natural dye sources was of great importance to the cultural conservation of wastra in South Sumatra.

In addition to its eco-tourism aspect, KRS also fostered public engagement through knowledge development. The complete transformation from an oil palm plantation into a botanical garden has become a subject of research for students specializing in land resource science. The radical transformation occurring at KRS has led to intriguing changes in the properties of the peatland within its area, making it worthy of further study. As of 2025, a total of 38 research activities—encompassing both internships and thesis research for degree completion—had been undertaken, and these endeavors would persist in the future. These research endeavors benefited not only the students but also aided the KRS management in comprehending its assets and facilitated informed decision-making for subsequent development. The mutualistic interaction between KRS management and students indirectly positioned KRS as a natural laboratory for the study and practice of peatland restoration.

4. DISCUSSION

4.1 Landscape-hydrological change: Beyond canal blocking and rewetting

Over fifteen years, KRS has developed a peatland restoration model distinct from the canal-blocking and rewetting approach dominant in Indonesia and other tropical peatlands. The conventional model, widely used by Indonesia's BRG since 2016, focuses on raising water tables through canal blocking, followed by native revegetation [15]. While effective for reducing carbon emissions, it relies on narrow ecological indicators and treats restoration as a time-bound project [3].

The KRS hydrological strategy diverged fundamentally from conventional peatland restoration practice. Most studies recommend canal blocking as the primary intervention to raise water tables and reduce carbon emissions in degraded tropical peatland [16]. In these approaches, hydrological management serves a single objective: carbon emission reduction. By contrast, KRS maintained the existing drainage network during Phase 1 (2010-2015) to prevent peat desiccation while site planning and nursery establishment proceeded. Over the 12-month monitoring period following infrastructure completion (September 2020–August 2021), the perimeter canal system with inlet-outlet gates and retention ponds appeared to reduce the sensitivity of site hydrology to rainfall variability [17], as indicated by the gradual water table decline despite sharp rainfall reduction (Figure 3). However, longer-term monitoring—particularly during extreme dry or wet periods beyond the 12-month window—remains necessary to confirm the persistence of this buffering capacity.

Unlike conventional canal blocking, which prioritizes carbon outcomes above all else, KRS suggests that hydrological infrastructure may serve multiple purposes [18] without compromising water table elevation. However, this multifunctionality comes at a cost: perimeter canal systems are more expensive and technically demanding than simple canal blocks [19]. The 2024 La Niña flood revealed an additional trade-off [20]: when restoration is located near communities, flood risk becomes a genuine management concern, not merely a carbon accounting footnote. This suggests that future restoration projects in inhabited peatlands must plan for both dry and wet extremes, not a simple return to "natural" water tables—a scenario planning gap evident in most existing hydrological guidelines.

4.2 Vegetation and conservation outcomes: From co-benefit to core mandate

In conventional peatland restoration, biodiversity outcomes are treated as co-benefits—secondary indicators measured if resources permit, but rarely central to project design [21]. Standard indicators focus on hydrology (water table depth) and carbon (emission factors) [22], while species recovery, when monitored at all, is limited to a handful of planted native trees. KRS inverts this priority entirely. As a botanical garden, conservation is a core mandate, not a co-benefit. The living collection of 477 cultivated specimens (117 species, 49 families) is maintained regardless of restoration outcomes, and 12 native Sumatran species with IUCN Red-listed status were deliberately collected, propagated, and planted as part of the garden's multifunctional mandate. Crucially, prioritizing ex-situ conservation increases the complexity of peatland restoration—shifting the goal from merely restoring degraded peat to actively building an ecosystem—a shift not addressed in conventional restoration frameworks [23].

The most distinctive finding—that five of these 12 species were obtained from village cemeteries, not forests—would not emerge from a conventional restoration monitoring framework. Conventional projects do not typically collect from cemeteries because their mandate is rewetting, not botanical exploration. This finding has practical implications beyond KRS: culturally protected landscapes across Sumatra (cemeteries, sacred groves, village forests) may harbor threatened species that formal conservation areas have failed to protect—a possibility that restoration guidelines have largely ignored. For restoration practitioners, this means community-engaged plant exploration can identify local genetic sources that are both culturally appropriate and ecologically valuable—an approach conventional canal-blocking projects rarely consider [24].

The presence of 12 IUCN Red-listed native species at KRS accounts for 2.22% of Sumatra's reported forest and wetland flora [14]. This percentage is not claimed as comprehensive ex-situ coverage of the region's threatened biodiversity. Rather, its significance lies elsewhere: these species now grow on a degraded peatland landscape where they were previously absent. Each individual, though currently maintained as an ex-situ living collection, occupies its native peatland habitat and will mature to produce seeds and other propagules. These propagules can be harvested, multiplied in nurseries, and deployed for large-scale reintroduction into surrounding degraded peatlands—a restoration supply function that conventional canal-blocking projects rarely provide. In this sense, the 2.22% figure is not an endpoint but a baseline: a nucleus of threatened species from which restoration material can be generated. Future work should prioritize seed orchard development and nursery capacity expansion at KRS to realize this potential.

The marked increase in survival rates following infrastructure completion confirms that hydrological regulation buffers planted collections against climate variability. During the pre-intervention period, survival rates were consistently lower for cohorts that matured under La Niña conditions—the 2015 cohort showed 40.0% survival when assessed in 2017, while the 2019 cohort showed 45.5% when assessed in 2021—compared to those that matured under El Niño conditions—the 2013 cohort showed 56.8% survival when assessed in 2015 and the 2017 cohort showed 63.6% when assessed in 2019. Under regulated conditions following the 2020 infrastructure completion, subsequent cohorts (planted in 2022 and 2023) exhibited two-year survival rates exceeding 95.7% (assessed in 2024 and 2025, respectively), indicating that hydrological regulation had effectively buffered the site against climate extremes.

4.3 Botanical-garden governance: Institutional permanence, failure tolerance, and landscape vulnerability

One of the most distinctive features of the KRS model is its ability to absorb and learn from failure. The 2015 and 2019 fires were not explained away as external anomalies but treated as operational setbacks that informed infrastructure investment. This capacity stands in sharp contrast to conventional project-based restoration, where donor reporting requirements incentivize low-risk site selection, proven methods, and selective reporting of success. A systematic review of peatland restoration literature found that failures—particularly fires and planting mortality—are substantially underreported, creating a biased evidence base that favors over-optimistic conclusions about restoration effectiveness [25].

KRS provides a rare longitudinal account of failure and learning over 15 years—something most restoration studies lack due to their project-based time horizons [21]. The five-year gap between the first fire and completion of water retention infrastructure represents real costs (lost specimens, carbon emissions, institutional credibility), but because KRS has no project closure deadline, it could absorb these costs and continue operating. This institutional feature—permanent mandate and patient capital—is not an accident; it is a deliberate design choice that project-based restoration cannot replicate. By contrast, a typical 3-5 year restoration project would have closed, reported partial success, and moved on. The case suggests a distinction that restoration policy may need to consider: project-based funding models incentivize failure avoidance, while institutionally permanent models may better incentivize learning from failure. However, this inference is drawn from a single case (KRS). Comparative studies across multiple restoration sites with different institutional arrangements would be needed to confirm this as a general mechanism.

However, conventional restoration projects often define their boundaries by administrative limits, implicitly assuming that interventions within those boundaries will produce measurable outcomes while external factors beyond the site can be safely ignored. The KRS case challenges this assumption. Two external landscape dynamics directly affected restoration outcomes at KRS despite optimal internal management. First, a coal haul road appeared along the northeastern boundary in 2021 (Figure 2(C)), signaling ongoing extractive land use adjacent to the site. Second, and more consequentially, a parcel of land previously characterized by shrubland was converted into an oil palm plantation, with drainage canals oriented perpendicular to the KRS boundary, directing water flow toward the botanical garden (Figure 2(D)). The 2015 and 2019 fires similarly originated outside KRS from adjacent land clearance and agricultural burning, not from within the garden itself.

These observations lead to an uncomfortable conclusion: even a well-designed, permanently institutionalized restoration site remains vulnerable to landscape-level drivers beyond its control—a finding that complicates the implicit assumption of most site-scale restoration studies that internal interventions are sufficient. The perimeter canal and water gates that successfully buffered KRS from rainfall variability (Section 3.1) cannot buffer drainage from adjacent oil palm plantations or fires from neighboring land. Landscape-scale restoration has been advocated for decades, yet most tropical peatland projects remain site-scale due to land tenure complexity and governance fragmentation. KRS demonstrates that without parallel investment in landscape governance—beyond the site boundary—the biophysical outcomes of site-scale restoration may be fundamentally capped, a limitation that most published restoration evaluations fail to acknowledge.

4.4 Supporting evidence and its implications

4.4.1 Interpreting anomalous peat depth increases

The peat depth measurements presented in Section 3.3.1 required cautious interpretation. Several points showed increases of +83 to +293 cm over two years—values far exceeding the maximum plausible vertical accretion rate for tropical peatlands (typically <2 cm yr^{-1}). These anomalies likely reflect spatial heterogeneity across the study area, compaction artifacts from repeated augering, and the inherent uncertainty associated with repeat measurements at non-fixed points within permanent plots. Therefore, these data should not be interpreted as evidence of net peat growth or long-term carbon accumulation at KRS. Instead, they serve two limited purposes: illustrating the high spatial heterogeneity of peat depth within a 100-ha site, and providing a methodological caution for restoration monitoring—without permanent markers and co-located elevation surveys, repeated peat depth measurements are unlikely to yield reliable estimates of change over short timeframes.

Beyond methodological artifacts, however, an alternative ecological interpretation deserves consideration. The 15-year restoration period at KRS was punctuated by alternating El Niño (drought) and La Niña (wet) events, which repeatedly stimulated boom-and-bust cycles in understory vegetation—particularly the dominant fern *Dicranopteris linearis*. This species produces rigid, hollow, highly lignified tissues that decompose slowly and may retain their physical form after death, creating a loosely packed, porous organic layer [26]. Under this hypothesis, the measured depth increases do not represent compacted, well-humified peat, but rather transient, low-density accumulation of undecomposed necromass—material that registers as "peat" in auger measurements but contributes little to long-term carbon storage. This interpretation would explain both the magnitude and reversibility of depth changes observed between 2023 and 2025. Confirmation requires additional data (bulk density, organic matter composition, permanent elevation markers), but if correct, this hypothesis carries an important implication: standard peat depth measurements alone cannot distinguish meaningful carbon accumulation from transient, low-density necromass buildup—a caution particularly relevant for restoration projects claiming rapid carbon recovery. Pending confirmation with permanent markers, elevation surveys, and bulk density measurements, the peat depth data presented here should be interpreted as a methodological caution rather than evidence of ecosystem change.

4.4.2 Socio-economic outcomes and political constituencies

Conventional restoration projects increasingly include "community engagement" components, but these typically take the form of consultation or compensation for lost access—rarely delivering direct economic benefits to communities [27]. Carbon credit revenues primarily flow to project developers and intermediaries, not to forest-adjacent communities, making the latter the least significant beneficiaries relative to their direct dependence on and interaction with forest ecosystems [28]. KRS operates on a different logic. Because the botanical garden has a permanent mandate beyond restoration, it can invest in livelihood linkages impractical for time-limited projects: (1) a natural dye program supplying local artisans, (2) eco-tourism generating local economic activity, and (3) environmental education reaching thousands of participants.

These outcomes suggest a hypothesis that warrants further investigation: that multifunctional restoration models may build local political constituencies for restoration [27]. At KRS, local artisans who benefit from dyes, tourism vendors who depend on visitors, and students who learn about peatlands all have a tangible stake in keeping the garden (and therefore the restored peatland) functional. However, this interpretation is based on garden records and artisan reports, not on formal interviews or surveys. We therefore present this as a case-based insight from KRS, not a generalizable mechanism. Dedicated social science research—including stakeholder interviews and economic valuation—would be required to test whether such political constituencies indeed emerge and whether they contribute to long-term sustainability.

This self-reinforcing political economy represents a mechanism that conventional restoration logic models have largely overlooked, and may prove critical for long-term sustainability—a hypothesis that warrants further investigation. However, this model sacrifices efficiency and scalability relative to conventional rewetting: building a botanical garden is more expensive and slower than blocking canals. The 15-year timeline from clearing to >95% survival is not replicable in a 3-5 year project—a temporal mismatch rarely discussed in restoration policy debates.

4.5 What Kebun Raya Sriwijaya sacrifices: Complementarity, not competition

Relative to conventional canal-blocking restoration, the KRS model sacrifices three things—trade-offs that multifunctional restoration advocates rarely quantify. First, efficiency. Building a botanical garden is more expensive and slower than blocking canals. The 15-year timeline to achieve >95% two-year survival for the most recent cohorts (planted in 2022–2023) is not replicable in a 3–5 year project, contrasting sharply with the rapid emission reduction targets assumed by carbon-focused restoration frameworks. Second, scalability. Indonesia needs to restore over 1.2 million hectares of degraded peatland. Developing botanical gardens across all of them is neither feasible nor desirable—a point often overlooked by proponents of multifunctional restoration who emphasize benefits without acknowledging scale constraints. Third, simplicity. The KRS model is harder to evaluate, harder to fund, and harder to replicate than canal blocking, requiring diverse expertise (botany, hydrology, education, tourism management, community engagement) and sustained political support—requirements that exceed the capacity of most conventional restoration projects.

The way forward is not to choose one model over the other—as much of the current restoration literature implicitly assumes by prioritizing carbon outcomes—but to recognize their complementarity. Conventional rewetting can be deployed at scale across remote peatlands where the primary objective is carbon emission reduction. The botanical garden model can be deployed at strategic nodes—near cities, on degraded plantation land, within existing institutional frameworks—to generate multifunctional outcomes that conventional restoration cannot. This complementary relationship has received limited attention in the peatland restoration literature and, to our knowledge, has not been explicitly articulated previously.

Synthesizing the findings across all domains, the KRS model offers three original contributions that distinguish it from conventional approaches. First, institutional permanence enables outcomes that project-based models cannot achieve. The capacity to survive the 2015 and 2019 fires, delay infrastructure investment, and continue operating after the 2024 flood—all without a project closure deadline—represents a category of restoration governance not captured by existing project-based logic models. Second, multifunctionality can be a design principle, not an afterthought. By embedding restoration within a botanical garden mandate, KRS generated outcomes—threatened species from village cemeteries, natural dye livelihoods, environmental education for thousands of participants—that are less likely to emerge from carbon- or hydrology-driven projects based on available evidence from this case. Third, even well-executed site-scale restoration remains vulnerable to external landscape drivers. Adjacent oil palm drainage, fires originating on neighboring land, and extractive land uses persistently affected site conditions despite optimal internal infrastructure, revealing fundamental limits that restoration science has under-theorized.

Thus, KRS does not replace conventional restoration but expands the available toolkit for peatland restoration practitioners and policymakers. The 15-year record provides preliminary empirical proof of concept for this expanded toolkit—a contribution that, to the best of our knowledge, has not been previously offered in tropical peatland restoration literature.

4.6 Limitations

Several limitations should be acknowledged. Hydrological data: continuous water table monitoring was available for only 12 months (September 2020–August 2021) due to COVID-19 disruptions; pre-intervention conditions (2014–2019) were therefore inferred from fire events, rainfall patterns, and satellite observations rather than direct measurement. Longer-term monitoring is ongoing. Peat depth measurements: large increases between 2023 and 2025 at several points exceed plausible accumulation rates, likely reflecting spatial heterogeneity or measurement artifacts, not net peat growth. Future studies should employ permanent marked points and co-located elevation surveys. Single-site case study: KRS is one botanical garden on one peatland in South Sumatra, so generalization requires caution. However, this study aims for analytical generalization—developing a model to be tested elsewhere. Replication at other Indonesian peatland botanical gardens would strengthen the evidence base. Socio-economic data: findings on natural dye, eco-tourism, and education are based on garden records and artisan reports, not independent economic evaluation. A formal economic impact assessment is a valuable next step. Carbon balance not directly measured: we did not measure greenhouse gas fluxes (CO_2, CH_4); carbon implications are inferential. Direct flux measurements using chamber or eddy covariance methods would strengthen future work.

5. CONCLUSION

Based on 15 years of data from KRS (2010–2025), the following findings are directly supported by the evidence: (1) Landscape-hydrological change. Hydrological infrastructure installed in 2020 reduced the sensitivity of water tables to rainfall variability over a 12-month monitoring period, although the rainfall-water table correlation was weak and non-significant ($r = 0.159$, $p = 0.621$). However, the same infrastructure prolonged inundation during the extreme wet event of early 2024, causing vegetation dieback. Anomalous peat depth increases (up to +293 cm) exceeded plausible accumulation rates and likely reflect methodological artifacts, not carbon sequestration; (2) Vegetation restoration outcomes. For the most recent cohorts (2022–2023), two-year survival rates exceeded 95%, a substantial improvement over pre-intervention cohorts (40–64%). Twelve native Sumatran species with IUCN Red List status (CR, EN, VU or NT) are conserved in the living collection, with five species obtained from village cemeteries; and (3) Governance and landscape vulnerability. The botanical garden's permanent mandate enabled learning from fire (2015, 2019) and flood (2024) setbacks. However, adjacent oil palm expansion and fires originating outside the boundary persistently affected site conditions, revealing fundamental limits to site-scale restoration.

The KRS model offers complementary advantages to conventional canal-blocking restoration but sacrifices efficiency, scalability, and simplicity. It is suitable for strategic nodes near population centers where biodiversity, livelihoods, and long-term governance are priorities. For remote peatlands where carbon reduction is the primary objective, conventional rewetting remains appropriate. Multifunctional restoration may build local political constituencies (artisans, visitors, students) whose stake in long-term sustainability warrants further investigation.

This study has several limitations: (i) water table data are limited to 12 months post-intervention without a pre-intervention control; (ii) peat depth measurements lack permanent markers and bulk density data; (iii) findings are from a single site, requiring replication elsewhere; (iv) carbon fluxes were not directly measured; (v) socio-economic data are derived from garden records without independent evaluation; (vi) ENSO-survival associations are correlational, not causal. Future work should include longer-term hydrological monitoring, permanent elevation markers for peat dynamics, controlled planting designs to isolate climate effects, direct greenhouse gas flux measurements, and formal socio-economic impact assessments. Replication at other peatland botanical gardens in Indonesia would strengthen the evidence base.

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AUTHOR CONTRIBUTIONS

All authors contributed to the writing of the manuscript, with the following descriptions: Dian Novriadhy and Sri Maryani played a role in concept development; Achmad Ubaidillah and Zepri Ariadi played a role in peat depth measurement and analysis; Tili Karenina and Oom Komalasari played a role in vegetation analysis; Wenni Tania Defriyanti, Desri Yesi, Hendrixon, and Popo Marinda played a role in socioeconomic analysis and political engagement.

STATEMENT ON THE USE OF GENERATIVE ARTIFICIAL INTELLIGENCE

During the writing process, only one AI tool, deepseek.com, was used to correct sentences and improve readability. The author provided input in the form of sentences containing analytical material, which the AI then refined. The author performed all data and analysis, and the AI generated no artificial data.

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