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Tropical Peatland Restoration Reduces Fire Occurrence

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Key Points:

- Hotspots density in restored peatlands reduced significantly by up to 2.6 km^{-2} compared to matched unrestored peatlands in 2019
- Deep wells show greater hotspot reduction and cover a larger area, while effect of canal dams is constrained by distance and their durability
- Peatland restoration contributed 38% of the total observed hotspot reduction between 2015 and 2019 across the restored peatlands

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract Extensive tropical peatlands in Indonesia have been drained and deforested, leaving them highly susceptible to fire. In response to widespread fires in 2015 El Niño, large-scale restoration schemes were implemented. While fire occurrence was dramatically reduced during 2019 and 2023 El Niño, it is not known to what extent restoration contributed. Here we examined the impacts of the two key restoration methods, canal blockings using dams and pumped groundwater from deep wells, on fire occurrence across Indonesian peatlands. We found restoration significantly reduced fire hotspot density in 2019 by up to 2.6 km^{-2}, compared to matched controls. However, the impact was not evident during 2023 when the fire hotspots were much lower. The effectiveness of restoration varies according to climate, ecosystem, and restoration type. Canal dams and deep wells together contributed 38% of the total fire hotspot reduction observed between 2015 and 2019 in 3,920 km^2 restored peatlands.

Plain Language Summary Fire hotspots and burnt area in Indonesian peatland have decreased since 2015, but the relative role of climate variability and peatland restoration is unclear. Our study estimates the effectiveness of the key peat restoration strategies (pumped groundwater and canal blocking) in reducing fire occurrence. Our analysis applies a matching methodology to separate the impacts of restoration and climate variability. We find that restoration has reduced fire occurrence by up to 2.6 hotspots km^{-2} (compared to unrestored matched controls). The effectiveness of restoration varies according to climate, ecosystem condition (land cover and region) and restoration type (method of restoration, age, and density of installation) up to 5.3 hotspots km^{-2}. Restoration reduced fire occurrence in 2019 by 22% in restored peatlands and accounted for 38% of the observed fire reduction between the 2015 and 2019 El Niño across 392,000 ha of restored peatlands. Restoration efforts have provided positive outcomes; hence maintaining and optimizing restoration practices remains a priority.

1. Introduction

Indonesia has about 15 Mha of tropical peatlands (Ritung et al., 2011), one of the largest carbon pools globally, but drainage and recurring fires result in large carbon emissions (Hooijer et al., 2010; Lu et al., 2022; Page et al., 2002; Prosperi et al., 2020; Ribeiro et al., 2021). The vulnerability of Indonesian peatlands to fires is exacerbated during El Niño years (Albar et al., 2018). In 2015, peatlands accounted for around 32% of 2.6 Mha burned scars despite these areas representing less than 8% of Indonesian land (BPS, 2023; MoF, 2023, 2025; Ritung et al., 2011); releasing up to 857 Tg CO_2 (Huijnen et al., 2016; MoEF, 2021). These fires resulted in environmental degradation, biodiversity loss, substantial health and socio-economic problems (Agus et al., 2018; Harrison et al., 2024; Hein et al., 2022; Kiely et al., 2021). The loss from 2004 to 2015 Indonesian peat fires reached US$93.9 billion (Kiely et al., 2021).

Intensive peatland restoration efforts commenced following the 2015 severe wildfires, building upon restoration techniques that had previously been undertaken at smaller scales (e.g., Ansori & Dohong, 2005; Applegate et al., 2012; Limin et al., 2008; Wibisono et al., 2005). The Peat Restoration Agency of Indonesia (BRG/BRGM) was established in 2016 to implement rewetting, revegetation, and revitalization activities (MoEF, 2017). Restoration aims to revive peat ecosystem functions, reduce carbon emissions, prevent fire, and improve the local economy (BRGM, 2021) targeting 1.2–2 Mha of degraded peatland (Presidential Regulation, 2016b, 2020). This represents one of the largest national efforts to restore degraded peatlands. Restoration projects still continue via initiatives like GREEN for Riau targeting conservation and emissions reductions.

Rewetting aims to reduce the water-table depth (WTD) to within 40-cm of the surface (Presidential Regulation, 2016a), typically through canal and ditch dams, groundwater pumping, and canal infilling. Several studies have explored the impact of canal dams on peatland WTD. In Riau, shallower WTDs were observed in areas close to dams (Safitri et al., 2024; Sutikno et al., 2020). In Central Kalimantan, the WTDs became shallower after dam construction (Limin et al., 2008). However, during El Niño years, WTD dropped to >40 cm reported in several restored areas (Lestari et al., 2022; Lu et al., 2021; Putra et al., 2021; Ritzema et al., 2014). Thus, during El Niño, restoration may not prevent the WTD from exceeding the fire-prone level defined by the government (>40 cm).

While several studies have examined the effects of canal blocking on peatland hydrology, there is still insufficient understanding of how rewetting contributes to mitigating fire occurrence. There have been decreases in both fire hotspot density and fire-vulnerable areas after restoration in Riau and Central Kalimantan (Hapsari et al., 2020; Taufik et al., 2023). However, no studies provide a spatial distinction between restored and control (e.g., unrestored) area under comparable pre-restoration conditions. In addition, burnt scars and hotspot count in Indonesia generally declined after 2015, including El Niño post-restoration years, 2019 and 2023 (MoF, 2025). Hence, isolating the role of climate variability and different restoration approaches is needed. Notably, despite being the most widely installed infrastructure, the effects of deep wells and groundwater pumping remains understudied.

It is currently unclear how peatland restoration activities impact fire occurrence relative to areas without restoration. This insight is essential for optimizing and sustaining restoration efforts through a wider understanding of their impacts, particularly when facing local livelihood challenges. Here, we make the first national-scale estimate of peat restoration (rewetting: canal dams and deep wells) effectiveness on fire occurrence. To account for climate variability, we assess fire occurrences on restored areas in comparison to matched unrestored areas. We focus our analysis on El Niño years, when fire activity is greatest. Given potential land cover effects, land cover was incorporated in our analysis (Salmayenti et al., 2025).

2. Methods

2.1. Study Area and Data set

Our study focuses on Indonesian peatlands in Sumatra and Kalimantan islands, based on a peat map accessible via the global PeatMap (https://archive.researchdata.leeds.ac.uk/251; accessed in July 2023) (Ritung et al., 2011; Xu et al., 2018). We analyzed peatland restoration activities implemented by BRGM and their partners (https://prims.brgm.go.id/, accessed in April 2024). We gathered data including information on restoration type, location, and construction year (2016–2023) in Sumatra (Figure 1a) and Kalimantan (Figure 1b). Two types of rewetting infrastructure were analyzed: canal dams and deep wells (details are provided in Description S1 of Supporting Information S1). Canal-infilling omitted due to small installations. About 86% of such infrastructure is located inside peatlands, with the remainder in nearby hydrological systems.

Fire is represented by an active fire (hotspot) product of the Visible Infrared Imaging Radiometer Suite (S-NPP) from FIRMS NASA (https://firms.modaps.eosdis.nasa.gov/download/, accessed in May 2024) (Giglio et al., 2024; NASA, 2024). Daily hotspot counts include day and nighttime detections, noting diurnal differences in minimum detectable fire size (Johnston et al., 2018). The land cover types are based on land cover maps from ESA and C3S (www.arcgis.com/home/item.html?id=1453082255024699af55c960bc3dc1fe, accessed in September 2023) and a plantation map from Transparent World accessed via Global Forest Watch (https://data.globalforestwatch.org/datasets/gfw::tree-plantations/explore, accessed in September 2023) (Copernicus Climate Change Service, Climate Data Store, 2019; Defourny et al., 2017; ESA, 2017; Global Forest Watch, 2016). A drainage map (https://purl.stanford.edu/yj761xk5815, accessed in July 2023) is used to identify drained peatlands (Dadap et al., 2021).

2.2. Matching Analysis

Our study compared fire occurrences between three El Niño years (NOAA, 2024): 2015 (pre-restoration), 2019 and 2023 (post-restoration). We divided the study area into restored and unrestored areas. We assumed restored areas are peatlands within 250 m of restoration installation. Unrestored peatlands are drained peatlands outside the restored areas. We divided studied areas into six categories of land cover: forests, plantations, crops, crop-dominated mixed natural vegetation (NV) (CNV), NV-dominated mixed crop (NVC), and NV-dominated mixed other vegetation (NVO). Due to variations in map production years, we restricted study areas to those with no land cover change during this period, accounting for 91% of restoration sites on Indonesian peatlands.

Cells with 0.5×0.5 km size in restoration affected and non-affected areas were generated separately for further analysis. The total area of each land cover type and the number of hotspots were calculated in each cell. The

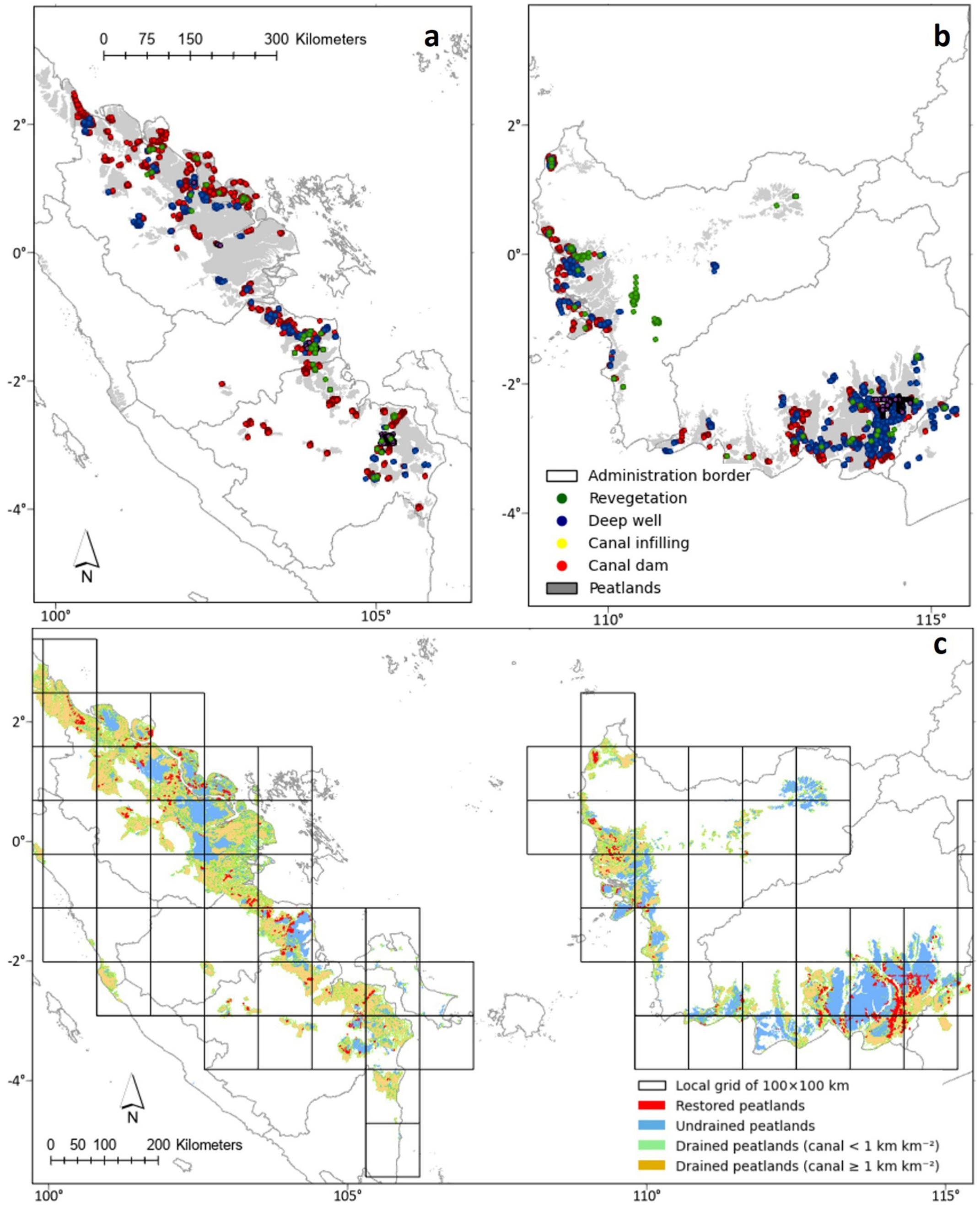

Figure 1. Study areas. Types and location of restoration activities in Indonesian peatlands of Sumatra (a) and Kalimantan (b) based on PRIMS data set (prims.brgm.go.id). Peatland conditions including undrained, restored and drained (c).

number of fire hotspots per unit area was calculated as hotspot density. Restored cells contained only restored peatland, and non-restored cells only drained-non-restored peatland.

We compared the hotspot density between restored and unrestored areas. To account for potential pre-restoration differences between restored and unrestored areas (Table 1), we applied a matching analysis (Figure S3 in Supporting Information S1). Firstly, we divided the area into 100 × 100 km grids, each assumed to experience the same local meteorological conditions. Second, we matched restored cells with potential unrestored cells within the same 100-km grid that experienced a similar 2015 hotspot density (±10%) and land cover type. Restored sites have a mean canal density >1 km km^{-2} (Table 1); hence, we filtered the matched unrestored cells accordingly.

Table 1
Mean Canal Density and Number of Fire Hotspots During 2015 Within 0.25 km² Cells (0.5 × 0.5 km) Surrounding Peatland Restoration

Areas	Density of installations	% of total installation	Total area (km^{-2})[a]	Canal density (km km^{-2})	2015 hotspot density (km^{-2})
Restored with canal dams	1	25%		2.5	8.0
	2	24%		2.4	8.7
	3	16%		2.5	9.4
	≥4	35%		2.5	10.5
	All		1,113	2.5	9.3
Restored with deep wells	1–2	13%		2.1	6.5
	3–5	20%		1.6	10.2
	6–10	30%		1.6	12.6
	≥10	37%		1.2	14.5
	All		861	1.5	12.6
Drained				2.6	3.2
Undrained				0.0	1.1
All peatlands				1.2	2.1

Note. Restoration was completed by BRGM and partners between 2016 and 2023, across peatlands of Sumatra and Kalimantan, Indonesia. [a]Is based on 0.5 × 0.5 km cell surrounding the restoration in peatlands.

We then selected matched cells at random from the array of potential matches, then compared the hotspot density after restoration. We undertook the matching and comparison separately for canal dams and deep wells.

2.3. Difference-in-Difference Model

To estimate the impact of restoration, we applied the difference-in-difference (DiD) method, allowing us to examine the change in hotspot density between restored and unrestored areas over time. This method allows us to eliminate any persistent differences between the two groups across the study period (Gertler et al., 2016). We used DiD with regression analysis for canal dams and deep wells between two different El Niño periods, 2015–2019 and 2015–2023, expressed as:

$$Y_{\text{it}} = \alpha + \beta_1 R_i + \beta_2 Yr_t + \beta_3 R_i Yr_t + \varepsilon_{\text{it}}$$

where Y_{it} represents the hotspot density of restored peatlands i in year t. R_i is a dummy variable equal to one in restored areas and zero otherwise. Yr_t, represents the time dummy, equal to one post-restoration and zero pre-restoration. R_iYr_t is is their interaction, equal to one if restored and post-restoration, otherwise zero. B_3 is the coefficient representing the restoration effect in restored areas relative to unrestored areas. ε_{it} is the random error term. We estimated the restoration effects based on land cover types, restoration density, age, and regions.

2.4. Sensitivity Test

A sensitivity analysis was conducted to assess how distance from restoration infrastructure affected results. Additional matching and DiD analyses were performed based on larger 1 × 1 and 2 × 2 km cells. We then estimated the total effect of canal dams and deep wells in the affected areas by multiplying the mean restoration effect of each restoration types with their total affected areas (km^2).

3. Results

3.1. Distribution of Installed Dams and Wells

There are at least 8,014 canal dams and 14,321 deep wells were installed (BRGM, 2024). Most restoration (82% of dams and 98% of wells) was completed before 2020 (Table S1 in Supporting Information S1). The installation exhibited distinct spatial distributions, linked with environmental factors such as the drainage network and historical fire (Table 1). Canal dams were typically widely dispersed across the landscape, whereas deep wells tended to be more clustered (Table 1). About 25% of dams were isolated, with no other dam present within the same 0.5 × 0.5 km cell, while 40% and 35% of dams were in clusters of 2–3 and ≥4 dams. In contrast, 67% of wells were located close to ≥6 other wells. Because of the clustering of wells, if we assume a 0.25 km^2 area of restored peatland around each dam or well, the area restored with dams was 1,113 km^2, 29% larger than the restored area around wells (861 km^2) despite around a third fewer dams being installed compared to wells (Table 1).

Restoration effort was targeted in areas of denser drainage that had experienced high hotspot densities in 2015 (Table 1). The mean canal density of peatlands with dams was 2.5 km km^{-2}, around twice that of all peatlands (1.2 km km^{-2}) and two-thirds greater than near wells (1.5 km km^{-2}). Areas selected for restoration with wells and dams (hotspots of 12.6 km^{-2} and 9.3 km^{-2}, respectively) experienced a 3–4 times greater hotspot density in 2015 than drained peatland and a factor 9–12 greater than undrained peatlands.

3.2. Hotspot Density in Matched Restored and Unrestored Peatlands

Matched restored and unrestored cells experienced a similar number of fire hotspots in 2015, with mean hotspot of 8.7 km^{-2} (canal dams) and 11.8 km^{-2} (deep wells), confirming that our matching method was successful (Figure 2a). In 2019, the hotspot density in peatlands restored with dams (5 km^{-2}) was significantly lower than in matched unrestored areas (6 km^{-2}, $p < 0.05$). The hotspot density for areas restored with deep wells (6.4 km^{-2}) was also significantly lower than in matched unrestored areas (8.8 km^{-2}, $p < 0.05$). In 2023, both restored and unrestored areas with dams and wells experienced fewer hotspots but with no significant differences between restored areas and matched controls.

3.3. Restoration Effectiveness in Reducing Fire Hotspots

Restored peatlands experienced significantly fewer 2019 fire hotspots compared to the matched controls. In 2019, deep wells and canal dams affected hotspot density by −2.6 km^{-2} ($p < 0.001$) and −1.1 km^{-2} ($p < 0.05$), relative to the unrestored control (Figures 2b–2e). In 2023, there was no significant effect of either restoration type (Figure S2 in Supporting Information S1).

The restoration effect on fire hotspot reduction, relative to the control group, varied across land cover type, infrastructure density, age of infrastructure, and regions. In 2019, deep wells reduced fire hotspots more than controls in most land covers, with significant effects seen in mixed natural vegetation lands (−4.2 to −2.2 km^{-2}, $p < 0.05$) (Figure 2b). There were also reduced hotspots associated with canal dams in most land covers (−1.9 to −0.2 km^{-2}), but the effects were insignificant except in the NVO (−5.3 km^{-2}, $p < 0.001$).

The effect on hotspot reduction also varied with the density of restoration infrastructure (Figure 2c). Areas with 1–3 dams showed insignificant effect, while areas restored by ≥4 dams exhibited a significant reduction in hotspots (−1.7 km^{-2}, $p < 0.05$). The same pattern also occurred for deep wells, with significant hotspot reductions in areas restored by ≥3 wells (−2.3 to −4.3 km^{-2}, $p < 0.05$).

The impacts of restoration differed between southern and northern Sumatra and Kalimantan (Figure 2d). In 2019, dams had a greater effect in the south with −1.3 km^{-2} (Sumatra) and −1.6 km^{-2} (Kalimantan), while the overall mean effect was −1.1 km^{-2}; the effect in northern Sumatra and Kalimantan was only −0.4 and −0.1 km^{-2}. Deep wells had a significant impact in southern Kalimantan with −3.5 km^{-2} ($p < 0.001$), while there was no significant effect in other regions.

In 2019, 1-year old canal dams significantly reduced hotspots by −2.2 km^{-2} ($p = 0.002$) while there was a lack of significant control for 2-year-old dams (Figure 2e). In comparison, both 1-year and 2-year-old deep wells significantly reduced hotspots, and the effect was greater for 2-year-old wells (−5.2 km^{-2}, $p < 0.001$). No significant effects were recorded for dams and wells constructed in the same year as the fire event.

We explored the spatial extent of restoration through comparing the impact of restoration with distance from the restoration infrastructure (Figure 3). In 2019, consistent significant effects of −2.6 km^{-2}, −1.4 and −1 km^{-2} ($p < 0.05$) were observed in 0.25, 1, and 4 km^2 cells centered on wells respectively. However, significantly reduced hotspots were only found for the 0.25 km^2 cells centered on dams (−1.1 km^{-2}) with no significant effect in larger cells.

We combined information on the spatial effect (Figure 3a) with the spatial distribution of restoration infrastructures (Figure 1) to estimate the extent of restoration and the overall reduction in fire. We estimated that the

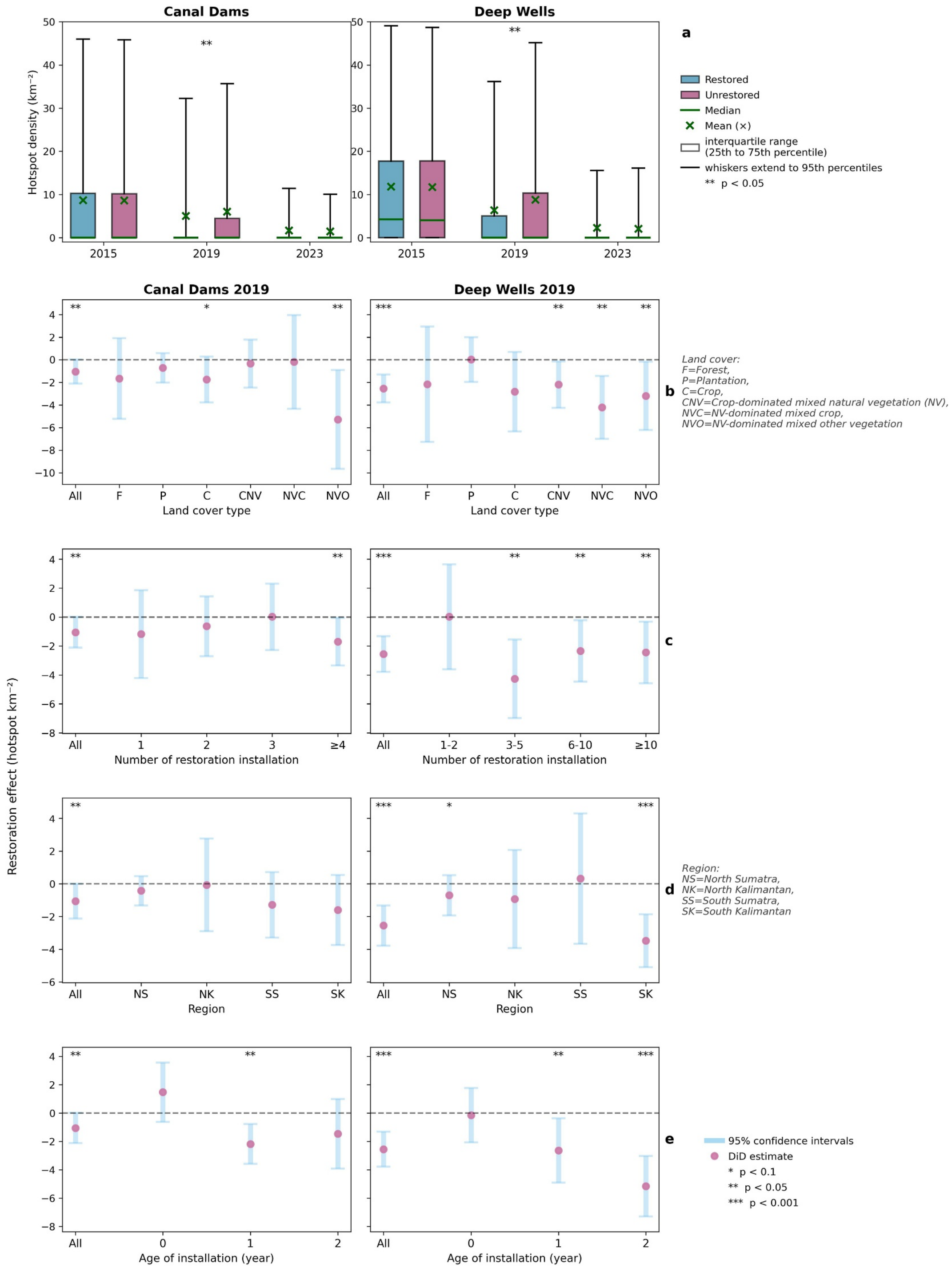

Figure 2. Restoration effectiveness in reducing fire hotpots. (a) The hotspot density of matched restored and unrestored peatlands in Sumatra and Kalimantan, Indonesia before (2015) and after (2019 and 2023) restoration, 0.25 km^2 cells are paired based on fire hotpot density in 2015. (b–e) Restoration impact on hotspot density (km^{-2}) in areas with canal dams and deep wells based on Difference-in-differences (DiD) model in 2019, showing the coefficient of the treatment (restoration) effect relative the control (unrestored) categorized by (b) land cover classes; (c) number of restoration installations; (d) regions; and (e) age of restoration in year.

peatlands that experienced significant hotspot reduction from canal dams and deep wells in 2019 was 3,920 km^2, where 2,788 km^2 are affected by wells and 1,113 km^2 are affected by dams (Figure 3b). This represents about 33% of the national restoration target (12,000 km^2) based on Presidential Regulation (2020). We estimated that dams

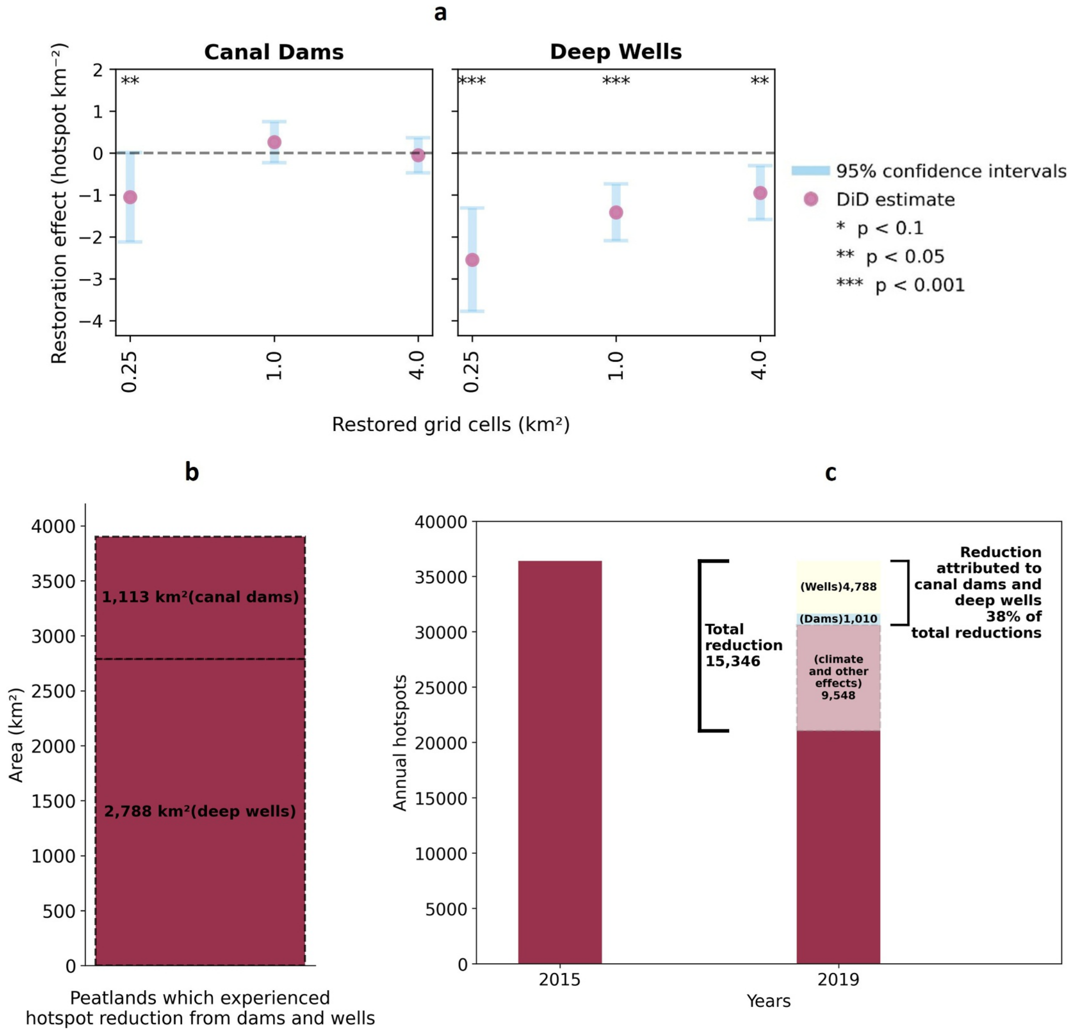

Figure 3. Restoration impact on hotspot density (km^{-2}) in areas with canal dams and deep wells between 2015 and 2019. (a) The restoration impact based on cells of 0.25, 1, and 4 km^2 centered on restoration infrastructure, based on Difference-in-differences (DiD) method, showing the coefficient of the treatment (restoration) effect relative the control (unrestored). (b) The total areas with significant hotspot reduction from restoration based on (a). (c) Total hotspot reduction.

and wells reduced fire hotspots across this area by 22%, contributing 38% of the total reduction in fire hotspots observed between 2015 and 2019 (Figure 3c).

4. Discussion

4.1. Effectiveness Comparison

We found canal dams delivered a smaller hotspot reduction compared to deep wells (Figure 2), potentially due to several factors. First, many dams were installed widely across the landscape at low density per 0.25-km^2 (typically ≤3 dams) (Table 1), while the significant effect was only found in areas that were densely restored (≥4 dams) (Figure 2d).

Second, our analysis suggests effective fire reduction at distances up to 2 km from the well, whereas canal dams were effective at reducing fire at distances only up to 250 m (Figure 3a). This suggests that the effect of canal blocking is constrained by distance (Hapsari et al., 2020; Sutikno et al., 2020), likely influenced by design, position, hydrological conditions, peat physical properties, and topography (Joosten, 2021; Suwito & Poedjirahajoe, 2022; Urzainki et al., 2020; Yakubson et al., 2023). Meanwhile, the operation of deep wells strongly depends on the efficiency of the associated task force, who not only aims to rewet dry peatlands, but also to extinguish burning peat when fire occurs (MoEF, 2018). This operation can reach wider areas using mobile infrastructure, like firefighting vehicles (MoEF, 2018).

Third, dams are likely less effective in strong El Niño years when both drained and undrained peatlands become drier and fire is less heavily controlled by drainage networks (Lu et al., 2021; Salmayenti et al., 2025). Therefore, canal dams may have had a limited impact during extreme dry periods, especially in areas of low-density dams. This finding is consistent with research showing that WTDs in peatlands with intact and blocked canals both fell >40 cm during El Niño years (e.g., Putra et al., 2021; Ritzema et al., 2014).

Furthermore, the functional lifespan of canal dams may influence their efficiency. Some dams were made from temporary materials, typically lasting only 1–5 years (Dohong et al., 2017; MoEF, 2023). Our results show that 1-year old dams have a significant and greater impact on fire compared to 2-year-old dams (Figure 2e). This suggests that older dams may be less effective; 23% and 88% of dams were ≥2 years old in 2019 and 2023. In addition to the materials, faster degradation of dams could also be influenced by flood damage or linked to agricultural activities and transportation access (Jaya et al., 2022; Suwito & Poedjirahajoe, 2022; Ward et al., 2021). The pronounced effect of dams in areas of natural vegetation with no crops (NVO) is likely due to reduced human disturbance. In contrast, our results suggest that deep wells maintain efficiency for at least 2 years, with 2-year-old wells resulting in the greatest fire reduction (Figure 2e).

4.2. Restoration Effect in Relation to Fire Management and Climate

The impact of deep wells varied across land cover classes (Figure 2b), which may be due to differences in fire management practices. Degraded NV areas experienced very dense hotspots (≥8 km^{-2}) in 2015 and showed a significant fire reduction after installation of deep wells. This reduction can be explained by rewetting operations targeting fire-prone areas with wells and clustered hotspot areas (BRG, 2019; MoEF, 2018). Meanwhile, areas with agricultural activities, especially plantations, had fewer 2015 hotspots. Restored and unrestored plantations showed no significant difference after restoration. In concession areas, landowners are required to prevent and suppress fires on their land, possibly explaining why the plantations were not affected by external rewetting operations (MoA, 2018). For cropland, the restored areas still exhibited a reduction in hotspots, although it was less pronounced than in non-agricultural land. In contrast, unrestored cropland showed an increase in hotspot occurrences, which may arise from regular agriculture activities (Hergoualch et al., 2018).

We find a greater effect of restoration in the southern peatlands, especially in Kalimantan (Figure 2d). Southern regions are more prone to ENSO conditions (Fanin & Van Der Werf, 2017), making them a priority area for restoration. This difference in effect could also be related to local variability in restoration implementation. Regional autonomy in Indonesia may cause restoration management to vary at local scale, reflecting differences in social, economic, and political dynamics across regions (Budiningsih et al., 2024; Harrison et al., 2020).

The impact of restoration was not strong in 2023, when hotspot densities were 85% lower compared to 2015, likely due to the climate variability linked to ENSO. Compared to previous El Niño periods, 2023 had a shorter transition into El Niño after a three-year La Niña, which is a rare case (Jia et al., 2025). In comparison, the 2015 El Niño was a peak of a prolonged extreme drought, which started in 2014 after an ENSO-neutral year; the 2019 El Niño started during the second half of 2018 after a two-year La Niña (NOAA, 2024). During La Niña, affected regions in Indonesia received more rainfall, which contributed to shallower WTDs on peatlands (Ismail et al., 2021; Kurniadi et al., 2021). Prolonged rainy periods can cause ponding on peatlands with flooding/inundation often widespread, especially in areas severely damaged by fires (Wösten et al., 2006). The length of transition between rainy to dry periods will affect the duration of the water level recession, also, varies according to the initial WTD (Ismail et al., 2021). The shorter dry-day period after a prolonged wet season in 2023, potentially caused the peatlands, especially in early 2023 El Niño period, to be wetter than in the previous El Niño periods. Furthermore, since hotspot occurrence was low in 2023, there were fewer targeted areas for the fire extinguishing operations to cover which likely enabled operations to cover both restored and unrestored areas (MoEF, 2016, 2018).

4.3. Implications for Peatland Restoration

Indonesia has implemented one of the largest peat restoration programs globally (Andersen et al., 2017; Vasander et al., 2003), targeting priority areas that are heavily drained and fire-impacted in Sumatra and Kalimantan (Figure 1 and Table 1). Our analysis shows that restoration with canal dams and deep wells reduced the fire hotspots in 2019 by 22% across 3,920 km^2 of degraded peatlands (Figure 3b). We estimated that restoration was responsible for 38% of the total observed hotspot reduction between 2015 and 2019, with the remaining effect due to climate variability (Figure 3c). Our work shows the success of a national scale peatland restoration scheme, providing evidence to support ongoing restoration. Indeed, there are still large areas of degraded peatlands that require restoration (Table S2 in Supporting Information S1).

Our analysis shows that fire still occurs in degraded peatlands even after restoration, especially in restored areas of degraded NV, where high hotspot density ($\geq$5 km^{-2}, Figure S1 in Supporting Information S1) was double the mean of all peatlands (2.1 km^{-2}, Table 1) and five times greater than undrained peatlands (1 km^{-2}). Our analysis identifies the challenge of reducing fire frequency to that experienced in pristine, undrained forested peatlands. This highlights a critical need to prevent deforestation and drainage expansion into undrained peatlands across Indonesia, particularly to mitigate fire-prone conditions associated with a warming climate and the potential for strong El Niño events in the future (Sasmito et al., 2025). Reductions in deforestation rate in Indonesia between 2015/2016 and 2024, particularly in primary forest loss, are therefore encouraging indicators (Angelsen et al., 2025).

We have shown that pumping groundwater using deep wells is effective in reducing fire; however, their use serves fire management more than ecosystem restoration. Wider ecosystem benefits from well operation in tropical peatlands requires urgent study. The well operations are labor-intensive (e.g., moving pumps from well to well), and requires coordination involving multilevel stakeholders, from community groups to central government, each with distinct roles and decision-making processes, which may involve administrative complexity (Waluyo et al., 2020). While canal dams need maintenance, they may be less labor and administratively intensive, and, when canals are fully blocked, they potentially reduce fires by up to 70% (Horton et al., 2022). When functioning, canal dams provide rewetting and ecosystem benefits via shallower and more stable WTDs especially during non-El Niño years, bringing wider ecological and greenhouse gas emissions reduction benefits (Hooijer et al., 2024; Lestari et al., 2022; Limin et al., 2008; Novita et al., 2022; Ritzema et al., 2014; Susilo et al., 2013; Tan et al., 2022). The relative economic and environmental benefits of dams and well operation in Indonesian peatlands require further assessment. Additionally, reduced fires would imply lower emissions, future analyses using Fire Radiative Power data would useful.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Availability Statement

The data used in this study were sourced from the following institutions and available data sets: BRGM (2024), ESA CCI Land Cover project (2017), and Copernicus Climate Change Service, Climate Data Store (2019), Transparent World-Global Forest Watch (2016), FIRMS NASA (2024), Dadap et al., (2021), and Xu et al. (2017). Detailed information on data access can be found in the Methods section of this paper.

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