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Transnational agricultural land acquisitions threaten biodiversity in the Global South

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Abstract

Agricultural large-scale land acquisitions have been linked with enhanced deforestation and land use change. Yet the extent to which transnational agricultural large-scale land acquisitions (TALSLAs) contribute to—or merely correlate with—deforestation, and the expected biodiversity impacts of the intended land use changes across ecosystems, remains unclear. We examine 178 georeferenced TALSLA locations in 40 countries to address this gap. While forest cover within TALSLAs decreased by 17% between 2000 and 2018 and became more fragmented, the spatio-temporal patterns of deforestation varied substantially across regions. While deforestation rates within initially forested TALSLAs were 1.5 (Asia) to 2 times (Africa) higher than immediately surrounding areas, we detected no such difference in Europe and Latin America. Our findings suggest that, whereas TALSLAs may have accelerated forest loss in Asia, a different mechanism might emerge in Africa where TALSLAs target areas already experiencing elevated deforestation. Regarding biodiversity (here focused on vertebrate species), we find that nearly all (91%) studied deals will likely experience substantial losses in relative species richness (−14.1% on average within each deal)—with mixed outcomes for relative abundance—due to the intended land use transitions. We also find that 39% of TALSLAs fall at least partially within biodiversity hotspots, placing these areas at heightened risk of biodiversity loss. Taken together, these findings suggest distinct regional differences in the nature of the association between TALSLAs and forest loss and provide new evidence of TALSLAs as an emerging threat to biodiversity in the Global South.

1. Introduction

Large-scale land acquisitions, typically defined as land transfers of at least 200 hectares (ha), have surged since the start of the century, with 126 million hectares (Mha)—an area larger than South Africa—currently under contract for agriculture, logging, and mining globally [1]. Recent food and financial crises have been especially influential in this 'global land rush' [2–4], with the total area of acquired land increasing sharply between 2007 and 2014, and continuing, albeit at slower speed thereafter [5]. Transnational land investments by foreign companies for natural resources or agricultural production account for 76% of total global land acquired [1] and 64% of the total number of large scale land acquisitions. While extractive relationships between the 'Global North' and the 'Global South' have characterized both the colonial past as well as contemporary neoliberalism [6], the intensity of this recent wave of land acquisition is unprecedented. In particular, the agricultural component of these investments is becoming increasingly important in the exchange between the Global North and the Global South, leveraging long colonial and imperial histories to link rising global demands for agricultural products with land and natural resources in low and middle income countries, primarily in Latin America, Eastern Europe, sub-Saharan Africa, and Southeast Asia [2, 5, 7–11]. Because such investments are often executed with the goal of generating backflows of agricultural commodities to transnational investors, foreign governments, and global markets [2, 4, 12], 'transnational agricultural large-scale land acquisitions' (TALSLAs) represent a distinctive geographic separation of the potential effects of production from the non-local benefits of consumption (e.g. [13]).

Governments in targeted countries are encouraging such investments—viewing these deals as mechanisms for agricultural technology transfer, rural development, and local job creation [7]—and are promoting TALSLAs in lands often described as 'marginal' [14–17]. While some evidence suggests that TALSLAs can produce positive impacts on local employment (e.g. [5, 18–20]), the new agricultural use of acquired land tends to have a lower labor intensity and crowd out smallholders [5, 18], which may minimize these benefits. As such, despite employment benefits observed in certain instances, the rationale of expanding large-scale commercial farming in the Global South is debated [21, 22] in both academic and political circles, as this form of development may overlook traditional land use of potential 'marginal lands' and agricultural modes of subsistence such as pastoralism, agro-ecological practices, and resource use by local communities [10, 23–27] to support livelihoods [28] and food security [13, 29–31]. In addition to the livelihood implications of LSLAs, lands acquired through such deals also provide a range of local (e.g. biodiversity) to global (e.g. carbon storage) ecosystem services [32]. Conserving such ecological services is necessary to enable 'nature based solutions' to key environmental and development challenges (e.g. [33]). Yet, the agricultural production transformation propelled by TALSLAs typically involves land use conversions associated with large-scale commercial farming systems, which may involve a range of environmental impacts [31, 32, 34]. Depending on the previous land use, the rapid expansion of TALSLAs may entail widespread impacts to natural systems and to biodiversity—the loss of which carries intrinsic cultural and existential value as well as palpable monetary costs for targeted areas [35]. In this vein of potentially problematic socio-environmental implications of TALSLAs, much of the 'global land rush' literature has originated from critical social science—particularly political ecology, agrarian studies, and critical political economy—to take normative positions fundamentally questioning TALSLAs [12, 36]. To complement this body of work—which has hypothesized many of the socio-environmental impacts of TALSLAs based on outcomes of site-specific case studies, we approach the issue from a quantitative environmental modeling perspective, adopting an integrated analytical strategy to provide novel empirical insights on the global ecological effects of TALSLAs.

While there has been a growing number of case studies examining selected environmental impacts of large-scale land acquisitions, it remains unclear whether the outcomes that they observe are representative across different contexts [37]. In selected countries, the recent availability of detailed geospatial information has enabled global assessments of the environmental effects of different types of land acquisitions intended for agriculture, logging, or mining. Recent work in Indonesia [38, 39] and Cambodia [40–42] has demonstrated that land investments for oil palm and rubber plantations have led to increased rates of forest removal. A similar global study of different land investment types found that oil palm, wood fiber, and tree plantations are associated with enhanced forest loss while forests within logging and mining concessions have experienced mixed outcomes [43]. Other work has evaluated land use changes associated with land investments (within and outside concession areas) in Mozambique and Kenya and found that land deals have driven deforestation and excluded smallholders from accessing water and other agricultural resources [44, 45]. Given the pronounced impacts to diverse ecosystems and their critical role in supporting global biodiversity [46], this handful of studies highlights the changes in land use and tenure that can be associated with land investments and points to the emerging risk that the rapid expansion of

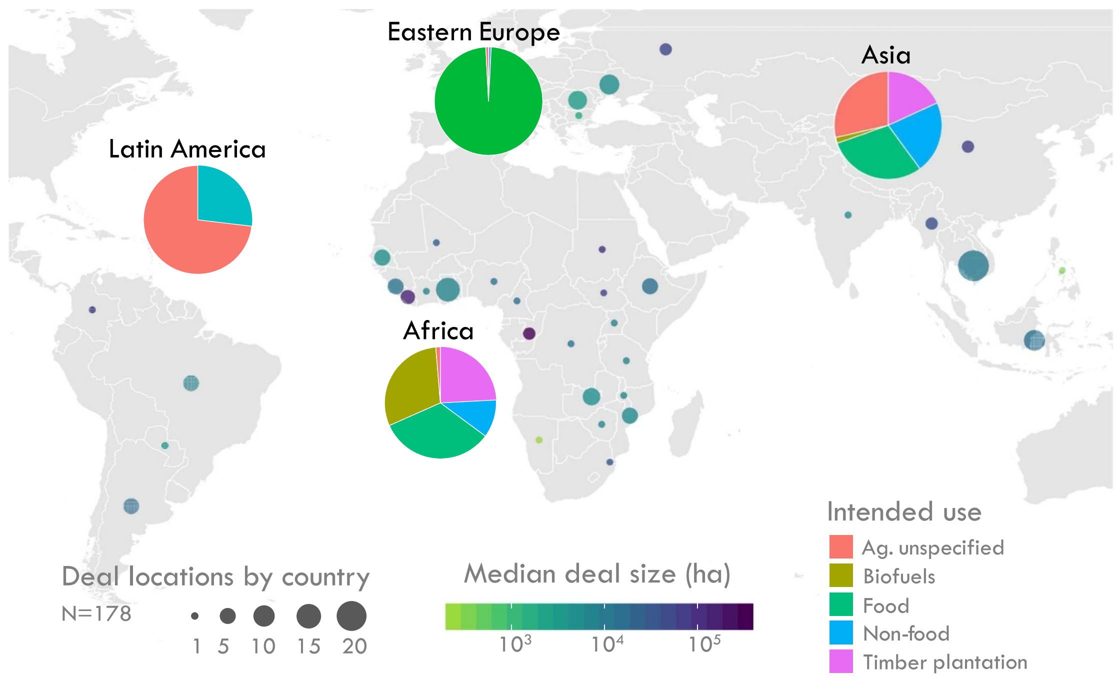

Figure 1. Location, size, and intended use of sampled TALSLAs.

TALSLAs poses to ecosystem functions, which are directly linked to human needs (e.g. livelihoods, food security, cultural identity) [30, 47]. Yet the spatial and temporal dynamics and magnitude of deforestation in and around TALSLAs has yet to been systematically evaluated at the global scale. Doing so can provide new insights on whether TALSLAs in fact contribute to forest loss and other land use changes, or if these transnational land acquisitions are merely established in places where enhanced deforestation and land use change are already occurring. In addition, the implications of TALSLAs for potential biodiversity losses in the future remain unquantified across a variety of agro-ecological contexts and intended uses.

To bridge these gaps, we perform a multi-country assessment of the ecological implications of TALSLAs intended for crops, livestock, or tree plantations. We examine 178 TALSLA locations (153 distinct land acquisition contracts)—covering 4.0 Mha across 40 countries—for which exact centroid locations are known [1] (figure 1). These TALSLAs comprise the largest available consistent global dataset with georeferenced locations and contract dates. We integrate this contract-specific data on geographic coordinates, size, contract year, and intended use [1] with annual maps of forest cover and satellite image time series to isolate the potential impacts of TALSLAs on forest loss and fragmentation between 2000 and 2018. We complement this historical analysis with an estimation of the future potential biodiversity impacts should the areas within TALSLAs be converted to agriculture. For that purpose, we use existing land system maps and associated biodiversity scores for relative species richness (i.e. number of vertebrate species relative to those supported by native vegetation) and abundance (i.e. number of individuals per vertebrate species relative to native vegetation). In parallel, we assess potential impacts on ecologically important areas by intersecting TALSLA locations with zones designated as either biodiversity hotspots, characterized by high numbers of endemic species facing substantial habitat loss, or as critical habitats deemed suitable for supporting threatened or endangered species. Taken together, these multiple lines of evidence provide a quantitative picture of the recent ecological effects associated with TALSLAs in targeted areas as well as their future implications for biodiversity in targeted countries. To the best of our knowledge, this study represents the most comprehensive global systematic evaluation of these environmental impacts for large (>200 ha) transnational agricultural land deals to date.

2. Results

2.1. Association between TALSLAs and forest loss

Relative forest cover, that is the fraction of land covered by forest, declined by an average of 6.6 percentage points (95% confidence interval: [3.8,10.0]) between the years 2000 and 2018 across the $N = 178$ TALSLAs of our sample (table 1). Considering that an average of 39% (95% CI: [24, 53]) of the land was forested per TALSLA ($N = 178$) in the year 2000, this corresponds to an average of 17% loss in forest cover (from 39% to 32.4%) per deal (table 1). Regionally, losses in relative forest cover are largest in

Table 1. Forest cover and loss within selected TALSLAs. Estimates were obtained from the full sample of TALSLAs ($N = 178$). 'Fraction of Land Matrix deals sampled' refers to the proportion of comparable TALSLAs in the Land Matrix that are georeferenced and included in the sample. 'Year 2000 forest cover' refers to the percentage of total acquired land covered by forest in the year 2000. '2000–'18 Losses' refers to the reduction in the fraction of total acquired land covered by forest between 2000 and 2018. To evaluate forest cover, TALSLA areas were pixelized and each 30 m pixel was defined as initially forested if it had a year 2000 tree cover of 50% or greater. Brackets provide bootstrapped 95% confidence intervals (1000 repetitions).

Region (# of deal locations)	Fraction of Land Matrix deals sampled	Total TALSLA size in sample (Mha)	Average year 2000 forest cover per TALSLA location (% of land)	Average 2000-'18 Losses per TALSLA location (%-points of land)
Africa (79)	16.7%	2.39	44% [20, 67]	5.0% [1.3, 10.3]
Asia (44)	10.7%	0.57	51% [28, 78]	15.5% [8.4, 27.0]
L. Amer. (12)	3.8%	0.55	31% [4, 38]	9.4% [0.8, 12.5]
Europe (43)	6.4%	0.51	10% [6, 14]	1.2% [0.2, 2.6]
All (178)	8.0%	4.02	39% [24, 53]	6.6% [3.8, 10.0]

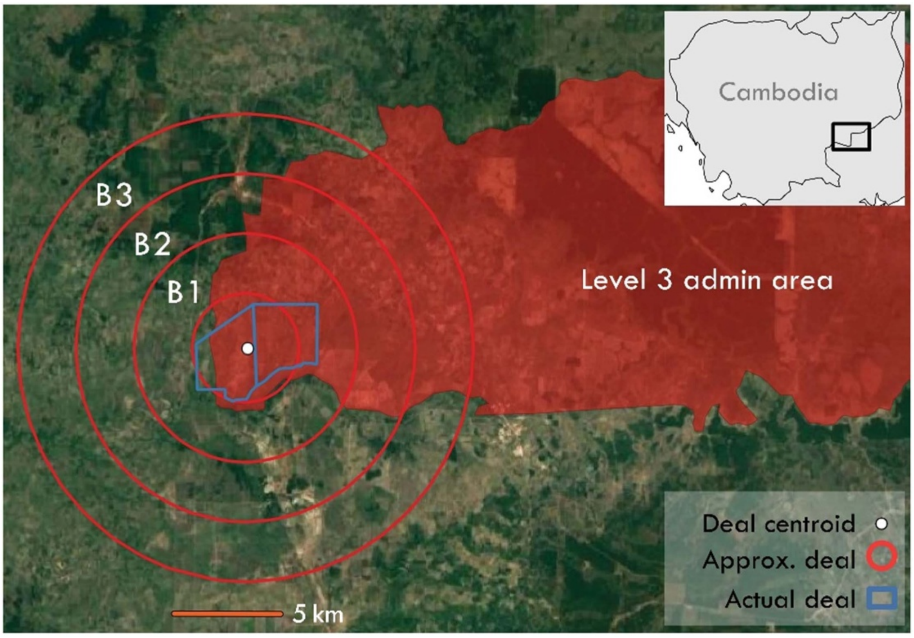

Figure 2. Example of approximated deal and control areas. Figure shows an approximated deal (central red circle) and spatial buffers (B1–B3, red lines) for a specific deal in eastern Cambodia. The actual deal extent is represented in blue. The red-filled area represents the level 3 administrative area (Commune) that contains the deal centroid (white dot). B2 and B3 are utilized as buffer control areas in the deforestation analysis, while B1 is not a valid control due to its substantial overlap with the actual deal (see also validation analysis in figure S5).

Asia (15.5%-points [8.4,27.0], from 51% to 35.5%) and Latin America (9.4%-point [0.8,12.5], from 31% to 21.6%), noting that the large confidence intervals in Latin America might arise from the region being under-sampled in the available data (see table 1 and Methods).

Forest loss within TALSLAs might be affected by confounding regional factors (e.g. favorable policy or economic conditions) affecting deforestation rates in their vicinity. To control for these regional fixed effects, we compared forest loss within TALSLAs to corresponding losses observed within a series of surrounding context areas (i.e. 'buffer' control areas B1, B2, B3 at increasing distances from each considered deal) (see figure 2 and Methods) at increasing distances from the area covered by each deal. For this analysis, we focus on the subset of $N = 131$ TALSLAs with initially forested locations where forest losses in relation to the land deal could be properly estimated (see Methods). Results reveal substantial regional differences. In Africa and Asia, losses in relative forest cover within a TALSLA between 2000 and 2018 were approximately 1.5–2 times higher than in surrounding buffer control areas. Within the considered $N = 131$ TALSLAs, the proportion of land covered by forest decreased by an average of 8.1%-points (95% C.I.: [2.5, 15.4]) and 15.6%-points (95% C.I.: [8.9, 26.4]) respectively in Africa and Asia, against a decrease of 4.5%-points (95% C.I.: [1.8, 7.6]) and 10.4%-points

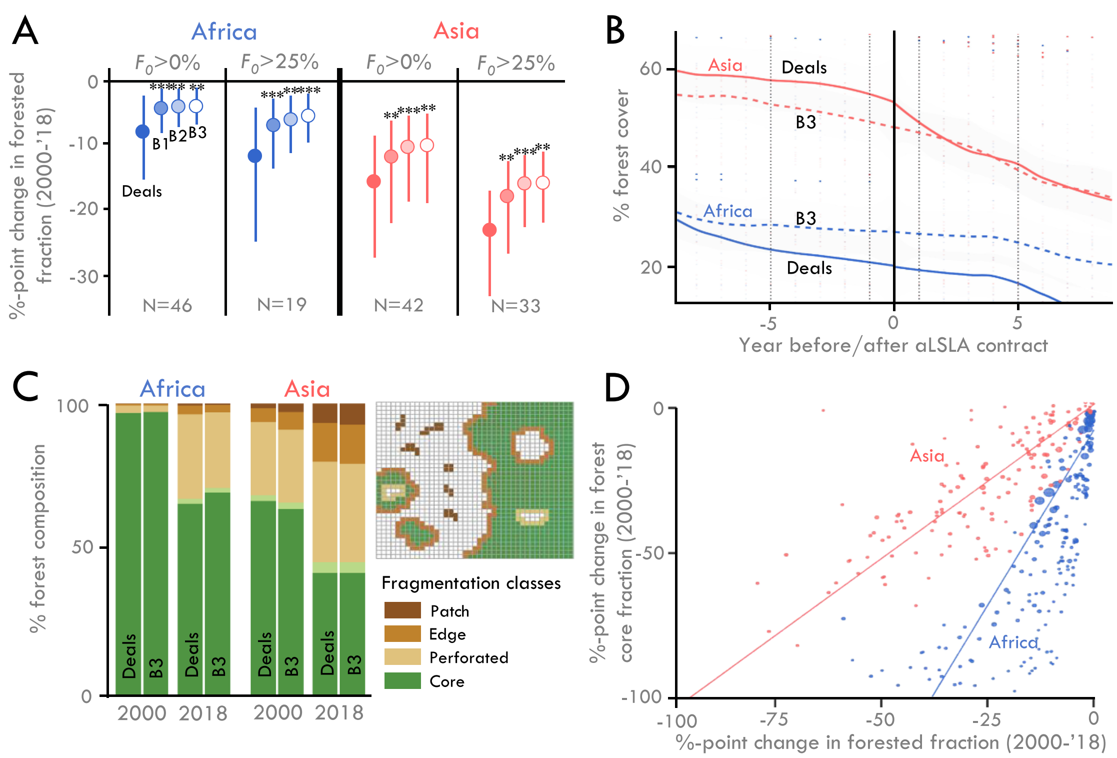

Figure 3. TALSLA-associated forest loss and fragmentation. (A) Relative change in forest cover (2000–2018) in deal and control regions in Africa and Asia using different thresholds of initial forest cover in the year 2000 (F_0). Deals with initial forest cover lower than that threshold or with zero initial forest cover in any of the corresponding zones (Deals, B1 B2, B3) were excluded. Error bars represent area-weighted bootstrapped ($R = 1000$) 95% confidence intervals around the mean. Asterisks represent bootstrapped p-values of pair-wise comparisons between relative losses in the deals and each of the control regions (*: $p < 0.1$; **: $p < 0.05$; ***: $p < 0.01$). For TALSLAs in Europe and Latin America, we found that forest losses within TALSLAs were not statistically significantly different from losses in control regions. (B) Annual forest cover estimates before and after the contract year of TALSLA deals in the acquired land (*solid line*) and control region B3 (*dashed line*) for Africa ($N = 40$) and Asia ($N = 39$). Lines represent forest cover estimates from LOESS regressions fitted independently to observations taken before (x axis negative) and after (x axis positive) the deal contract year, with 95% confidence intervals in light grey. (C) Changes in forest composition, in terms of fragmentation classes, between 2000 and 2018 for the deal regions and control region B3 for Africa and Asia. Forest fragmentation classes are given as percentages of the total forested area across the 46 African deals and 42 Asian deals in our sample with non-zero initial forest cover in the year 2000. The pixel map on the right of the panel illustrate the different fragmentation classes for a fictitious forest cover raster. (D) Relation between changes in land fraction occupied by forest (x-axis) and changes in the fraction of forest occupied by the core fragmentation class (y-axis) for Africa ($N = 46$) and Asia ($N = 42$). Deals are colored by continent and their symbol size is proportional to their (absolute) initial forested area. Linear regression trends are represented as colored lines.

(95% C.I.:[5.6, 19.1]) in the corresponding B2 buffer control regions (figure 3(A)). These differences increased substantially when further focusing on the subset of deals with a higher initial tree cover (figure 3(A), $F_0 > 25\%$). This suggests that the association between land deals and deforestation is higher for more densely forested deals, where the potential implications for biodiversity and carbon reduction may be most pronounced [48]. In Latin America and Europe, in contrast, we found that deforestation within the TALSLAs locations does not occur at a significantly (95% CI) higher rate than in the surrounding buffer control areas. This suggests that forest loss within TALSLAs in these regions might be a manifestation of broader regional deforestation trends rather than the effect of the land deals themselves.

Importantly, a higher deforestation rate within land deals in Asia and Africa is not sufficient to establish that a causal relationship between TALSLAs and forest losses exists in these regions. Indeed, temporal trends of deforestation using non-parametric regression (LOESS) methods applied to our reduced sample of $N = 131$ forested deals suggest distinct mechanisms across regions (figure 3(B)). In Africa, higher deforestation rates appear to have taken place in the deal regions prior to the TALSLA contract year. While it is possible that some deals require the land to be cleared before the transfer of ownership, a covariate matching analysis (see figures S1–S4, table S1) shows that factors typically associated with deforestation (low topographic slope, high soil suitability for agriculture and proximity to roads, trains, rivers, and cities) [40] are more prevalent within land deals than in the control regions. This pattern in Africa suggests a more complex relationship between TALSLAs and deforestation, whereby pre-existing active forest clearing (or the factors that facilitated it) may have made the acquired land more attractive to foreign investors in the first place. In Asia, on the other hand,

Table 2. Potential changes in mean relative species richness (RSR) and relative abundance (RA) within TALSLAs. Results for individual TALSLAs ($N = 178$) are shown in figure 4.

Region (# of deals)	Year 2005 RSR (%)	Potential RSR change (%-points)	Deals experiencing reductions in RSR (%)	Year 2005 RA (%)	Potential RA change (%-points)	Deals experiencing reductions in RA (%)
Africa (79)	77.5	−14.4	89	80.5	2.9	51
Asia (44)	82.1	−22.3	91	83.3	16.2	21
L. Amer. (12)	72.7	−8.8	83	72.5	6.2	33
Europe (43)	71.0	−7.2	98	74.8	−8.1	95
All (178)	76.7	−14.1	91	79.3	3.5	53

forest clearing tends to happen after the land acquisition within the area of investment, but not in the surrounding areas. The covariate matching analysis also suggests that, in Asia, the surrounding buffer control areas (B2) and (B3) are sufficiently comparable to the TALSLAs to be used as counterfactuals of forest loss within TALSLAs had the deals not been concluded. Under these conditions, differences in forest loss between TALSLAs and the surrounding areas can potentially be interpreted as the causal effect of land deals on deforestation (figures 3(B) and S2).

The ecological implications of deforestation are determined not only by how much forest is removed but also by the patterns by which forest loss occurs [49–51]. Continuous canopy cover affects key determinants of biodiversity, such as home ranges and migration corridors, and may be compromised by TALSLA-related forest degradation [52–55]. To evaluate this effect, we characterized the spatial fragmentation of forest cover by distinguishing the spatially continuous forest core from discontinuous tree cover classes (i.e. forest patches, edges, and clearings) that are less ecologically favorable. Focusing on our reduced sample of $N = 131$ forested deals, we find that the fraction of forest area occupied by continuous core forest in TALSLAs decreased from 69% (Africa) and 97% (Asia) in 2000%–46% (Africa) and 67% (Asia) in 2018 (figure 3(C)). However, the analysis also shows no discernible difference in the magnitude of this change between the deals and the buffer control regions. This suggests that forest fragmentation might not be driven by the land deals themselves, but rather by confounding factors affecting both the deals and the control regions. We do however find a strong relation between deforestation rates and the fragmentation of the remaining forest cover, indicating that the fraction of remaining forest occupied by the core decreases more in those deals where deforestation rates are the highest (figure 3(D)).

2.2. Estimated effects of *TALSLAs* on biodiversity

While forests are of vital importance for biodiversity conservation [56], we find that the locations of the study TALSLAs span a diversity of land use classes and ecosystems—in contrast to often being promoted as occurring in 'marginal land'—and point to varied implications for biodiversity effects if these deals are fully put into production. To estimate this potential, we quantify the expected effect of converting each of the 178 TALSLAs of our original sample (i.e. now including land deals in Europe and Latin America, and deals with no forest cover in 2000) to its intended agricultural use. Focusing on vertebrate species (i.e. birds, mammals, amphibians, reptiles), we use two key biodiversity indicators: relative species richness (i.e. number of species in an area) and relative species abundance (i.e. number of individuals per species). Under a conversion to their intended use from their original (year 2005) land system [57], we estimate that nearly all 178 TALSLAs locations (91%) will experience substantial losses in relative species richness (−14.1% reduction on average within each deal) (table 2, figures 4(A) and (B)). However, for relative species abundance, roughly half (53%) of all studied TALSLAs might see reductions of up to −30%, while the rest (47%)—mostly those intended as tree plantations—could experience substantial increases of up to +90% (table 2, figures 4(C) and (D)). While potential relative species richness loss is especially pronounced within deals that would transition from natural forest to tree plantations, potential relative abundance loss is expected to be highest for TALSLAs that intend to convert extensively used mosaic cropland (or idle and marginal land) into intensively cultivated cropland. Relative abundance may increase for land that is converted from mosaic cropland (or idle and marginal land) to tree plantations. Most deals experiencing high relative species richness loss are located in Africa (−14.4% loss on average) and Asia (−22.3% loss on average), while the majority of deals experiencing high relative abundance loss are located in Eastern Europe (−8% loss on average). With increasing distance from the deal area, we observed similar mean values for relative species richness loss (changes ranging between −0.4 and 0.3%) as well as abundance loss (changes ranging between—0.3 and 0.7%). In all, this suggests that TALSLAs will have the greatest biodiversity impact on species richness, while species abundance may be either negatively or

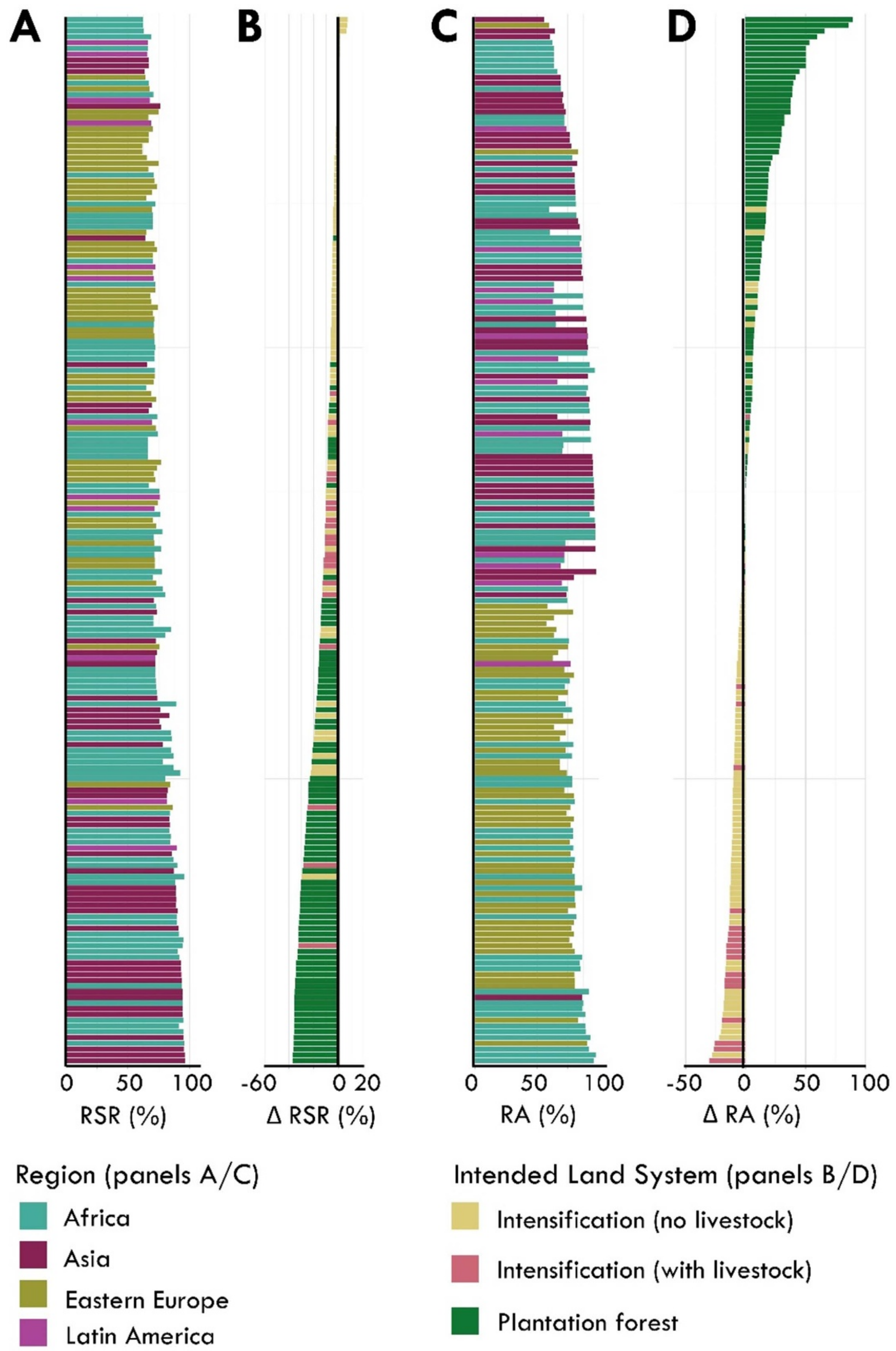

Figure 4. Estimated changes in species richness and abundance. (A) Year 2005 relative species richness (RSR), (B) potential RSR change (%) after TALSLA implementation, (C) year 2005 relative abundance (RA), and (D) potential RA change after TALSLA implementation. Bars in panels A and B represent the same deal, and bars in panels C and D represent the same deal. Colors indicate the region of implementation (A) and (C) and the pathways of expansion and intensification (B) and (D).

positively impacted in the land area, depending on its original land use and conversion type.

To further understand the potential biodiversity impact implications of TALSLAs, we also examined the extent to which contracted production areas of TALSLAs overlap with areas defined as biodiversity hotspots or critical habitat. We find that 39% of TALSLAs fall at least partially within biodiversity hotspot areas; the majority of these TALSLAs (35% of all deals) have >80% of their area within biodiversity hotspots (figure 5(A)), placing them at high risk of losing biodiversity. A smaller percentage (13%) overlap at least 40% of their contracted size with likely critical habitat (figure 5(B)). In addition, should indirect land use change occur (i.e. change caused by the TALSLAs but occurring in the control regions B1–B3) as a result of TALSLAs establishment, the overlap with and potential impact on biodiversity

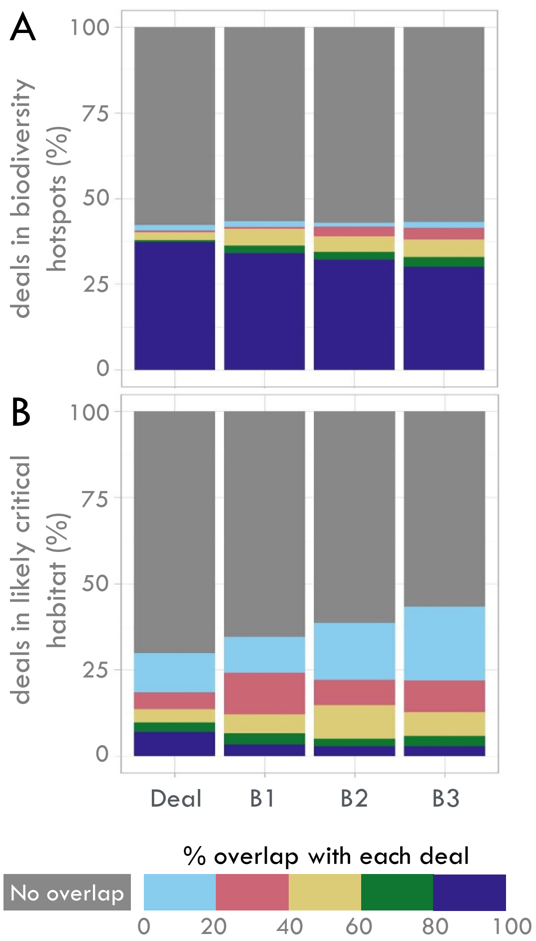

Figure 5. Percent overlap of TALSLA areas with biodiversity hotspots and critical habitat. Categories show the percentage of each deal's area overlapping with (A) biodiversity hotspots and (B) critical habitat. Biodiversity hotspots are characterized by high numbers of endemic species as well as substantial habitat loss, and critical habitats are areas deemed suitable for supporting threatened or endangered species. Both are considered highly important, containing unique ecosystems that are under threat from human pressure [97] but do not necessarily lie within protected areas.

hotspots and likely critical habitat would remain stable or increase modestly (up to 42% and 14%, respectively).

3. Discussion

Our analysis provides new evidence quantifying the observed and potential transformations of landscapes affected by TALSLAs across different targeted regions in the Global South. Our findings indicate that forest cover declined significantly within TALSLAs, especially in Africa and Asia, over the last two decades and became more fragmented, thus increasingly threatening ecologically important landscapes and vertebrate biodiversity. However, the mechanisms relating TALSLAs to forest loss vary substantially across regions. In Europe and Latin America, TALSLAs were often granted in areas with existing cropland and mainly served as a mechanism for farm consolidation [13], which explains the comparatively low initial forest covers (table 1). While we did not detect significantly different forest losses within TALSLAs compared to surrounding areas in these regions, this does not necessarily mean that such differences do not exist, as small sample sizes (particularly in Latin America) might have prevented us from detecting them. However, it does suggest that deforestation within TALSLAs might be, to a large extent, explained by the interaction with regional characteristics rather than the land deals alone.

In Asia, TALSLAs have been granted in areas with relatively high forest cover and may have contributed to deforestation. Deforestation is significantly higher within the deals than in comparable surrounding areas and accelerates shortly after the land acquisition. This suggests that, in Asia, accelerated forest loss within TALSLAs is directly related to agricultural expansion (e.g. palm oil and rubber) in remote and densely forested areas. The temporal dynamics that we observe also suggest that transnational land investments in this region are typically put to productive use relatively quickly (as opposed to remaining as fallow, speculative, or failed investments) (figure 3(C)). In Africa, the temporal patterns of forest loss suggest a different mechanism of transnational land deal establishment. Deforestation rates within African TALSLAs are larger than in the surrounding areas and appear to frequently increase several years before the deal contract year. This suggests that land deals in this region may not have significantly increased deforestation, but rather benefitted from previous land clearing (e.g. for pasture or smallholder farming) and/or their spatial covariates (e.g. proximity to roads and rivers).

Regardless of its cause, the rapidly decreasing—and increasingly fragmented—forest cover within TALSLAs has strong ecological implications, particularly if the acquisition overlaps with an ecologically sensitive zone. Our biodiversity analysis increases our understanding of these ecological implications, showing that the complete development of the considered TALSLAs can have dramatic consequences for biodiversity within targeted countries. Our findings suggest that TALSLAs can pose a serious threat to relative species richness and that the productive use of a large fraction of deals directly impact areas deemed important for supporting endemic, endangered, or threatened species. We found similar potential biodiversity losses in areas surrounding the deals, suggesting that TALSLAs may also give rise to indirect ecological impacts beyond the deal boundaries if the previous land users are 'pushed out' to surrounding land [44]. However, our results related to species abundance demonstrate nearly equal number of deals expected to see increases in abundance as compared to loss of abundance. This demonstrates that the previous land use and the conversion type of a given TALSLA is critical for fully understanding the biodiversity impacts.

The implications of such substantial losses of forest cover, especially in Africa and Asia, are noteworthy for their potential livelihood impacts. Forest cover is positively associated with desirable health outcomes—including greater diet diversity and reduced micronutrient deficiencies in children [58], improved water quality [59], and potential reduction in malaria [60]. As such, the immediate ecological impacts of TALSLAs could cascade into longer-term human health and livelihood consequences, further undermining the achievement of global development goals. Further, the combined effects of habitat reduction and human proximity to previously remote species that TALSLAs enable may increase the incidence of zoonotic diseases and the probability of future outbreaks (e.g. Ebola, COVID-19) [51, 61–66]. Such human livelihood impacts should be carefully weighed against the potential livelihood gains in employment or income that are often discussed related to TALSLAs. Future research could more completely estimate the system tradeoffs of LSLAs, including the potential livelihood gains or losses resulting from ecological damage.

Together, these findings provide evidence that current TALSLA practices will have deep ecological impacts on biodiversity and ecosystem functioning. Given the vast extent over which TALSLAs are granted globally (+399% in the last decade according to the entire Land Matrix Database) and the continued increase in areas affected by TALSLAs [5], there are multiple strategies that could help reduce potential environmental impacts. First, instituting ecological services criteria in concession procedures and acknowledging the importance of traditional agricultural practices when authorizing, implementing, and monitoring such deals can play a key role in local-to-global scale protection of biodiversity and promotion of sustainable development [67]. Such efforts can utilize 'no net-loss' approaches—in terms of biodiversity and livelihood impacts—to establish a minimum threshold for the implementation of transnational land deals, and alternative programs such as community conservation areas [68] can be implemented to recognize the fundamental role of local communities in protecting biodiversity [69, 70]. However, any efforts to do so must involve international and multi-lateral agreements across countries, which further points towards the necessity of greater policy and governance attention to TALSLAs. In particular, as the international community considers science-based targets for biodiversity conservation through mechanisms such as The Earth Commission [71] and the next round of Program of Work for the Convention on Biological Diversity [72, 73], policies and agreements across countries to implement ecological criteria into approval processes or avoidance of TALSLAs altogether in highly vulnerable or sensitive areas may be critical. For instance, mismanagement of both the granting and regulation of TALSLAs could work at counter purposes to national conservation commitments under the recently agreed global target of 30% protected area by 2030—and the associated historic establishment of the Global Biodiversity Fund. Simultaneously, ensuring that TALSLAs are governed appropriately and monitored after a deal should be prioritized on the agenda of governments and donor institutions. Kenya's National Land Commission is an example of on organization which manages public land on behalf of the national and county governments, initiates investigations into present or historical land injustices and recommend appropriate redress, and monitors and has oversight responsibilities over land use planning throughout the country. However, given that national governments operate under very different institutional and governance structures, further research could better explore the socio-political contexts in which TALSLAs occur and their implications for effective governance.

Multiple existing instruments all provide opportunity to govern under such principles including the UN Guiding Principles on Business and Human Rights [74], the voluntary Guidelines on Responsible Governance of Tenure of Land, Fisheries and Forests [75], and the Principles for Responsible Investments in Agriculture and Food Systems [76]. Finally, as many TALSLAs are associated with products for major multinational companies, such entities can be better held to account for the impacts of their acquisitions through mechanisms such as the United Nations Global Compact [77]. While many major global food companies entered into framework agreements in 2015 to comply with guidelines on buying land in areas where formal documentation and legal systems over ownership are questionable or contested, critics have argued these efforts are not enough and furthermore usually focus on land rights rather than other impacts [78]. As such, there remains continued opportunities for improving both our understanding of the longer term ecological impacts of TALSLAs through research as well as monitoring of shifting governance, institutional, and corporate commitments.

4. Methods

4.1. TALSLA data

Information on individual TALSLAs came from Land Matrix [1], a joint international initiative of several research and development organizations which has been collecting data on transnational and domestic land deals since 2009. The Land Matrix initiative acknowledges several possible data biases: from the beginning of the initiative the main focus has been on agricultural as well as transnational deals, whereas domestic deals are being collected in a systematic way only in five selected countries so far. Furthermore, different levels of transparency with regard to official data on land deals, degrees of freedom of press, and strength of reporting networks can bias the data [5]. To minimize this bias, we limit our dataset to transnational agricultural deals, noting that insofar as domestic deals are based on similar technologies, modes of production, and pathways of similar land use change, some of the findings of our analysis might also be relevant. We focus on land deals that (a) entail a transfer of user rights from smallholders and communities to commercial users through sale, lease, or concession; (b) cover an area greater than 200 ha; (c) focus on agricultural production (e.g. crops, livestock, tree plantation); and (d) have a concluded or confirmed contract between 2000 and 2018 [17]. These steps identified an initial population of 2141 Land deals, which was reduced to an initial sample of 208 deal locations (supplementary data 1) by selecting deals that contained details on their exact geospatial position or location coordinates. To our knowledge, this sample is the largest existing global dataset of georeferenced TALSLAs. We further excluded deals that were obviously centered on a town (based on Google Earth Pro imagery), shared the same geospatial position with another deal, or fell with its contracted area completely within the contracted area of another nearby deal (either within sample or georeferenced for another intended use (e.g. mining)). These also helped to ensure that our buffer areas were valid to serve as counterfactuals. After filtering based on these criteria, the final TALSLA sample for the study contained 178 deal locations (representing 153 distinct land acquisition contracts) across 40 countries and four continents (here referred to as 'regions'). Our sample is, however, representative of the wider 2141 Land Matrix deals (figure S12), that reveal no substantial difference in terms or deal size, contract year, regional location, and intention of investment, which builds confidence in our assumption that our sample is representative of the full population of Land Matrix deals (but may still contain any bias inherent in the original Land Matrix dataset). Note that our results might be sensitive to whether speculative land deals are simultaneously less likely to be reported as speculative (and georeferenced) and to have a substantial effect on forest cover or ecological outcome. This caveat should be considered when interpreting our results.

The spatial extent of each deal was approximated as a disc (i.e. deal area) with surface area and center coordinates equivalent to the values reported by Land Matrix [1] (figure 2). We note that these deal areas approximate the actual spatial extents of the deals, as the exact delineation of each deal's boundary was not reported in Land Matrix. A validation against a sample of TALSLAs with known boundaries suggests an overlap of approximately 75% between TALSLAs and their approximation (figure S5); this locational error is not accounted for in the bootstrapped confidence intervals in figure 2(A) and table 1. Land acquisition contracts containing $k > 1$ deal locations—which may be geographically distant and managed separately—were split into k deal areas, each with an area $1/k$ of the total deal area and centered on each reported location. Overlapping areas were associated with all of the corresponding deals (i.e. were double-counted) in the final analysis; because these overlapping areas account for only 1.1% of the total deal coverage, we do not expect them to affect the final results. However, deal areas overlapping with Land Matrix deals that were not included in the analysis (e.g. mining deals) were spatially subtracted (see e.g. figure S5). Lastly, we generated three control areas (B1, B2, and B3) outside each deal area, each with a width equal to the radius of the corresponding circular approximation of the TALSLA extent (figure 2). Validation against TALSLAs with known boundaries suggest that approximately 21%, 0.06% and <0.01% of control areas B1, B2 and B3 (respectively) overlap with the true TALSLA (figure S5). For this reason, we consider control areas B2 and B3 (but not B1) as unpolluted representations of the areas surrounding the deals. Once determined, the deal disk and control areas B1, B2 and B3 were rasterized to match the resolution of the considered forest cover dataset (see below).

For the analyses of the spatio-temporal patterns of forest loss and fragmentation, we reduced the initial sample of 178 deal locations to remove deals where the relative change in forest cover could not be suitably estimated. Specifically, we removed deals with insufficient initial forest cover (32 locations); a deal location was removed if less than 1% of the 30 m pixels that are contained in either the deal area or any of the control areas are more than 50% covered by canopy. We also removed deal locations with no reported contract year, or a contract year outside of the 2002–2016 period because no reliable average forest cover before or after the deal could be estimated (15 locations). These pre-processing steps left a final dataset of 131 deal locations (46 in Africa, 42 in Asia, 34 in Europe, and 9 in South America), which were used for the forest loss and fragmentation analysis. Note that the biodiversity analyses (i.e. changes in relative species richness and abundance; overlap with biodiversity hotspots and critical habitat areas) were performed on the original sample of 178 deal locations.

4.2. Forest cover

We used the Global Forest Change forest cover data [79] to estimate initial forest cover in 2000 as a fraction of the surface areas of each TALSLA and their corresponding control areas. The Global Forest Change database provides initial forest cover as the fractional tree cover (i.e. vegetation taller than 5 m) of a 30 m pixel. For the main analysis we define a pixel as being initially forested if its year 2000 tree cover is 50% or more, following [79]. Using information on the year of forest loss within each pixel (where it occurred), we then estimated annual forest cover as a fraction of the area of the deal or control region.

We performed three robustness checks to alleviate potential concerns about forest cover detection. First, to ensure that results are not driven by the arbitrary choice of initial tree cover threshold, we replicated the analysis using the same forest cover product (Global Forest Change), but defining a pixel as being initially forest if its year 2000 tree cover is >0%. Second, to address concerns that tree cover may be challenging to accurately estimate in sparse tropical forests [80], we also replicated the analysis using an alternative Landsat-based product of humid tropical forests [81]. The dataset includes annual data on different forest classifications including 'undisturbed tropical moist forest' (Class 1), 'degraded tropical moist forest' (Class 2), and 'deforested land' (Class 3) and covers 81 of the studied locations. Third, to ensure that results are not driven by tree plantations misclassified as forest in the Global Forest Change dataset, we replicated our analysis on a subset of 87 locations where overlaps with established tree plantations (obtained from the Global Forest Watch [82]) were identified and masked out. For all three replication analyses, we computed the difference between the 2000–2018 decrease in relative forest cover within the deals and the corresponding decreases in each of the three control areas. Across all of these robustness checks, we found no statistically significant difference with the corresponding results obtained from the main analysis using the Global Forest Change dataset (figures S6–S8).

4.3. Forest fragmentation

Following the approach by Vogt *et al* [83], we performed a fragmentation analysis which defined four classes of forest cover (i.e. cores, patches, edges, perforated) (figure 3(C)). Using the Global Forest Change forest cover dataset for the year 2000 and cumulative gain and loss for the year 2018 [79], we first defined each pixel as forested (i.e. having forest cover greater than 50%) or non-forested (i.e. having forest cover smaller than 50%). Forest *cores* were defined as forested pixels that are not adjacent to non-wooded pixels. Forest *patches* were forested pixels that are not adjacent to forest core pixels. Forest *edges* were forested pixels at the forest–nonforest boundary and were classified as *perforated* if they were adjacent to smaller non-forested areas. A threshold of 100 m across a non-forested area was used to distinguish between perforated and edge pixels. It is possible that fragmentation estimates are affected by uncertainties associated with the boundaries of TALSLAs, which are approximated as disc-shaped in our analysis. To alleviate this concern, we replicated the fragmentation analysis for a subset of 13 TALSLA locations in Cambodia with known boundaries [84]. The resulting fraction of forested area occupied by core fragmentation classes is not significantly different from that obtained with the disc-shape approximation (figure S9).

4.4. Excess forest loss within TALSLAs and potential causal interpretation

We evaluated forest loss within each deal and control region as the decrease in the fraction of land area covered by forest between 2000 and 2018:

$$\Delta FC_{ig} = FC_{ig,2018} - FC_{ig,2000} \qquad (1)$$

where $FC_{ig,2018}$ and $FC_{ig,2000}$ are the area fraction of the region type g (deal or controls) associated with deal i and covered by forest at the beginning (2000) and end (2018) of the considered period. The expectation of ΔFC_{ig} for each region type g, and the associated confidence interval, were then obtained through non-parametric bootstrap (1000 repetitions), weighing each observation by the area of the corresponding TALSLA. The excess forest loss in each deal, compared to the corresponding control area was then estimated as:

$$\delta_i = \Delta FC_{i,\mathrm{DEAL}} - \Delta FC_{i,\mathrm{CONTROL}} \qquad (2)$$

where $\Delta \mathrm{FC}_{i,\mathrm{CONTROL}}$ designates $\Delta \mathrm{FC}_{ig}$ for each of the two considered control area (B2 and B3). Estimates reported in the Results section consider control area B2. A robustness check using TALSLAs with known boundaries shows that the overlap between B2 and the true TALSLA is likely minimal (figure S5). The expectation of δ_i was estimated through parametric bootstrap (1000 repetitions, weighted by the area of each TALSLA). Accordingly, $E[\delta]$ can be interpreted as the expected excess change (in percentage-point) associated with TALSLAs in the deal region, compared to the baseline change in the control region. For $E[\delta]$ to be interpreted as the average effect of TALSLA on forest loss, two conditions must be simultaneously satisfied [85]: (a) the land deal should not affect forest loss in the control areas (exclusion restriction), and (b) control areas should be comparable to land deals in all aspects affecting forest loss, except for the deal itself (comparable controls). We note however that—while satisfying these two conditions means that there is no evidence to the contrary of a causal interpretation—they do not necessarily imply a strong causal relationship.

4.4.1. Exclusion restriction

Although challenging to test formally, we investigated the exclusion restriction assumption by estimating forest loss (ΔFC) for the smallest administrative area that contains the deal centroid, as determined by the Level 2 or Level 3 (depending on the country) administrative sub-division from the GADM database [86].

The considered administrative subdivisions are typically many times larger than the corresponding TALSLA, and so the average forest losses within them are unlikely to be affected by the land deals. We found no significant difference with the corresponding forest losses in the control area B2 (figure S10), which builds confidence in the exclusion restriction in the context of this analysis.

4.4.2. Comparable controls

The second condition necessary for a causal interpretation of $E[\delta]$ is that factors influencing forest losses within land deals should be comparable to those in the control areas, except for the effect of the TALSLAs themselves. We investigated this condition by testing whether a set of known covariates of forest loss were significantly different in the deal and corresponding control regions—a significant difference would imply that controls are not comparable to deal areas. To proceed, for each TALSLA we randomly selected approximately 3000 points in the deal and control areas—approximately 6% of these points fell within the deal areas. For each random point, we estimated the distance from the nearest road, distance from the nearest waterway, distance from the nearest railway, distance from the nearest urban area (i.e. population density greater than 300 people km^{-2}), slope class, and agro-ecological suitability, all of which are known covariates of forest loss (see [40, 43]). Distance from the nearest urban area was calculated using a year 2005 population density dataset from the Center for International Earth Science Information Network [87]. Classes for median-terrain slope and agro-ecological suitability for rain-fed high-input cereals were assigned using data from the Food and Agriculture Organization/International Institute for Applied Systems Analysis Global Agro-Ecological Zones [88]. For TALSLAs in Asia and Africa (where $E[\delta]$ is significant), we found that the expected value of two of the six covariate was significantly different ($p < 0.05$) in the deal and the control areas: the distance to waterway and the distance to road (figure S1). In Latin America, only the distance to road was found to be significantly different between the treatment and control regions, although this effect is dominated by one large outlier land deal in Brazil. No covariate is significantly different between the treatment and control areas when we remove the outlier. Similarly, no significant differences were found in Europe for any of the covariates. As such, performing covariate matching described in the following paragraphs for Latin America and Europe would produce similar rates of forest loss as reported in figure 3.

4.4.3. Covariate matching and temporal dynamics

The above results suggest that higher deforestation rates within TALSLAs—as compared to the immediately surrounding regions—do not necessarily imply that TALSLAs have *caused* increased forest losses. To investigate the hypothesized causal relationship between TALSLAs and forest loss, we carried out two complementary analyses.

First, covariate matching was performed in *R* using the 'Matching' package [89] to identify a set of random points in each control region (i.e. B2, B3) with covariate distributions that are nearly identical to those of the random points within the deal area (table S1). We then estimated the average change in forest cover $\Delta FC_{\mathrm{CONTROL}}$ for the control areas corresponding to each TALSLA, but only using the set of identified matching points (as opposed to the entire control area). Estimates of $E[\delta]$ using the new $\Delta FC_{\mathrm{CONTROL}}$ values are reliable estimates of the causal effect of TALSLAs on forest loss. Our approach effectively combines spatial controls [13] and covariate matching [40, 43, 90] to construct deal-specific counterfactuals. It adequately controls for factors known to influence forest loss (i.e. covariates) and isolates the potential association with enhanced forest loss for each individual TALSLA.

Second, we also performed non-parametric regressions (LOESS [91]) against time for forest cover from within TALSLA and control areas. The analysis was performed on a reduced sample ($N = 40$ in Africa and $N = 39$ in Asia), where deals concluded before 2003 or after 2015 were removed, in order to ensure a sufficient number of years within the Global Forest Change dataset [79] both before and after the contract year. Similar to regression discontinuity designs (e.g. [92]), separate non-parametric regressions were executed before and after the contract year in order to estimate the level and trend of forest cover immediately before and immediately after the establishment of the deal. In standard regression discontinuity analysis, the difference between the two is used to estimate the causal effect at the discontinuity (i.e. the establishment of the TALSLA). Here the temporal resolution of the data (annual observation) and possible lags between the establishment of TALSLAs and their effect on forest cover prevented using a regression discontinuity within a quantitative estimation. Instead, we leverage the framework to qualitatively visualize the relative timing of forest loss and TALSLA in the deal and control areas. In particular, whereas a discontinuity at the deal year denotes an immediate short term effect, LOESS allow for a visualization of whether selection (change in forest losses in anticipation of the deal) or lagged effect (delayed changes in forest losses due to the deal taking time to be implemented) may be taking place (see figure 3(B)).

Outcomes from the covariate matching analysis for Asia are indistinguishable from our original spatial control method (figure S2). This suggests that the spatial controls B2 and B3 in Asia represent valid counterfactuals for deforestation within TALSLAs in the absence of the land deal, and that the relationship between TALSLAs and enhanced deforestation can be interpreted causally. This is not the case in Africa, however, where the covariate matching analysis yields significantly different results from the spatial controls approach (figure S3). Specifically, conditions in B2 and B3 are less favorable for clearing than in the TALSLAs themselves, suggesting that the spatial control areas B2 and B3 are not valid counterfactuals for TALSLAs in the region. These results are consistent with the LOESS results showing enhanced deforestation in TALSLAs shortly after the deals in Asia, but *prior* to the deals in Africa. The analysis was not applied to Europe and Latin America, where deforestation within TALSLAs are not significantly different from the surrounding areas (i.e. $E[\delta]$ is not statistically significant).

4.5. Potential impacts to biodiversity

4.5.1. Relative species richness and abundance

We assessed species richness and abundance change—for vertebrate species (i.e. birds, mammals, amphibians, reptiles—along three pathways of land use change (originally developed by Kehoe *et al* [57]) using a global land systems map and well-established statistical relationships between land use and biodiversity. The land systems map of Kehoe *et al* [57] is approximately representative of the year 2005, which is close to the beginning of our study period (i.e. 2000) and before most of the reported contract dates of the study deals (see e.g. figure S11). The data that allow for estimates of changes in species richness and abundance under land use change originate from the PREDICTS database, a collation of 11 525 local-scale studies with the goal of quantifying species-level and community-level responses to a range of human activities including agriculture, deforestation, introduction of invasive species, and urbanization [93]. Using a space for time approach, generalized linear mixed effects models analyzed 320 924 records of species richness and 1130 251 records of abundance to predict percentage changes in species richness, rarefied species richness, and abundance relative to a natural baseline as a result of various levels of agricultural land-use intensity [94]. Our model considers factors such as human pressure and invasive species while climate change effects are excluded because spatial comparisons do not capture climate change effects well [94].

The results from these earlier analyses provide estimates of vertebrate biodiversity loss per land-use intensity class relative to a natural unimpacted baseline. We estimated the potential spatial patterns of biodiversity loss for each agricultural development pathway in accordance with the intended use for each TALSLA following two main steps. (a) *Calculation of average biodiversity loss per land system*: We first matched our land-systems classes to levels of high, medium, and low intensity for each land-use type (for detailed conversion table, see [94]). We were then able to calculate the average biodiversity loss per land system—relative to an unimpacted baseline—by taking the mean model estimates of biodiversity loss per land-use intensity class from this existing work [94]. The result gave average relative biodiversity gain or loss per land system. (b) *Calculation of the biodiversity percentage change*: While the original PREDICTS estimates are from an unimpacted baseline, our estimates represent the biodiversity loss associated with shifting from one land system to another. In other words, our estimates of biodiversity loss are the difference between 1. the loss that has already taken place in converting from natural land to low or medium intensity (given in step a above) and 2. the biodiversity loss associated with high-intensity land use under the intended use of a TALSLA. This was calculated according to the land-system conversion in each of three considered pathways: (i) *Intensification* (with few to no livestock) in the case of an intended use for crop production, (ii) *Intensification with livestock* in the case of intended livestock production, or (iii) *Plantation forest* (intense use) in the case of an intended use for any type of tree plantation. To estimate the biodiversity loss associated with land-system conversions, we took the difference between the loss associated with the original land system and the loss associated with the new land system to which our pathway converted. We then divided this by the loss estimate for the original land system. This gave the relative biodiversity change per conversion pathway (table S2). These calculations were done for each TALSLA (i.e. its approximate circular shaped area) as well as for the corresponding three control areas (B1, B2, B3).

4.5.2. Biodiversity hotspots and critical habitats

Using the circular deal area and the control areas, we calculated the percentage of each deal overlapping with two spatial datasets representing biodiversity hotspots [95] and likely critical habitat areas [96].

Data availability statement

The data used, generated, and analyzed in the current study are publicly available or are available from the corresponding author upon reasonable request. Publicly available data can be accessed through the sources provided in the associated references.

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